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A natural capital approach to agroforestry decision-making

by

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Statement of Ethical Conduct

The research associated with this thesis abides by the international and Australian codes on human and animal experimentation, the guidelines by the Australian Government's Office of the Gene Technology Regulator and the rulings of the Safety, Ethics, and Institutional Biosafety Committees of the University.

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Abstract

Agroforestry systems are well known to enhance provision of multiple ecosystem services (e.g. wind speed reduction, habitat provision, carbon sequestration). These ecosystem services deliver both public and private benefits at a range of scales. For example, well-designed agroforestry systems enhance livestock productivity at the paddock scale by providing shelter from extreme temperatures, while also contributing to biodiversity conservation and climate change mitigation at broader scales. Despite growing global awareness of these benefits, adoption, particularly in some temperate industrialised agricultural landscapes including those in Australia, remains constrained. To encourage adoption of agroforestry and inform optimal design of agroforestry systems, new approaches are required to measure and communicate the broad range of benefits provided by agroforestry. Communication of farm-scale benefits is particularly important, as decisions relating to adoption and design of agroforestry in these landscapes generally fall to individual landowners. Natural capital accounting (NCA), a process of accounting for natural resources underpinned by ecosystem service valuation, could provide the framework for such an approach.

While NCA is currently effectively applied at regional and national scales, this thesis examined whether NCA concepts can be usefully and practically applied at finer scales, such as individual farms or paddocks, to improve farm-scale decision-making and encourage adoption of agroforestry.

The first of three research questions asked: can measurement and valuation of multiple ecosystem services, within the NCA framework, be usefully and practically applied for the purpose of encouraging adoption of agroforestry? To answer this question, existing concepts and methods for ecosystem service measurement and valuation were reviewed to assess their suitability for application in this context (Chapter 2). The review found that while some existing concepts (e.g. ecosystem service classification systems) are readily transferrable to agroforestry, established methods for ecosystem service measurement and valuation require adaptation (namely in how they deal with scale and choice of beneficiary) in order to be usefully applied to encourage adoption of agroforestry and inform decision-making. A discrete choice experiment (DCE) survey was also conducted with farmers from areas of Australia that hold potential for expansion of temperate agroforestry (Chapter 6). Results of this survey highlighted the importance of demonstrating 'value for money', or a return on investment in agroforestry, and revealed that farmers value a wide range of ecosystem services in agroforestry systems. These findings confirm the need for agroforestry valuation methods that incorporate multiple ecosystem services and acknowledge different forms of 'value'. Ecosystem service valuation within the NCA framework provides this flexibility and is therefore useful in an agroforestry context.

The second research question explored the issue of scale: which scale is most appropriate when applying NCA concepts for the purpose of encouraging agroforestry adoption and informing decision-making, and what implications does this choice of scale have for ecosystem service measurement and valuation? A review of the literature (Chapter 2) showed that because farmers are the key decision makers, NCA concepts should be applied at the farm or paddock scale and focus on farmers (or their investors) as the key beneficiaries. Chapter 2 also identified that there are significant gaps in the evidence base for measurement of ecosystem services by agroforestry assets at fine scales, particularly in the case of biodiversity-related services. Field experiments conducted in shelterbelt systems in the Midlands region of Tasmania, Australia (Chapters 3, 4, and 5), addressed several of these gaps. Chapter 3 showed that paddock-scale provision of a range of ecosystem services (wind speed reduction, wood production, carbon sequestration, and habitat provision) can be predicted based on measurement of vegetation structural characteristics. Chapters 4 and 5 found that

shelterbelts support invertebrate communities that differ significantly from those in open pasture, and that they improve potential for pollination in adjacent paddocks. These findings contribute to the expanding evidence base for ecosystem service provision in agroforestry, at scales that are useful for informing decision-making by farmers.

The third research question focused on the critical NCA concept of 'condition', asking: is vegetation structure a suitable and practical metric for assessing the condition of agroforestry assets? Results from the DCE (Chapter 6) showed that farmers had strong preferences for particular agroforestry design attributes (e.g. tree species selection) and further, that farmers are motivated to make design choices that will maximise benefits in alignment with their objectives. Many of the ecosystem services identified as being important to farmers (e.g. wind speed reduction) are influenced by vegetation structure (e.g. shelterbelt porosity). Chapter 3 examined whether vegetation structure offers a practical solution to the issue of condition assessment, by quantifying the impact of shelterbelt tree species selection (*Pinus radiata*, *Eucalyptus nitens*, and mixed native species) on specific structural attributes (e.g. height, porosity, and floral diversity) that determine fine-scale provision of key ecosystem services. Species selection, in addition to shelterbelt age, was found to significantly affect structure, and therefore delivery of services and benefits at fine scales. However, Chapters 4 and 5 found that shelterbelt tree species selection, and therefore structure, had relatively low influence on composition of invertebrate communities within and adjacent to shelterbelts. Findings from Chapters 3-5 show that while structure serves as a practical condition metric that is useful for informing agroforestry design, tracking flows of some services (e.g. biodiversity) may require consideration of broader landscape-scale condition metrics such as connectivity and complexity.

Overall, this thesis demonstrates that with some adaptation, NCA concepts (e.g. condition assessment, ecosystem service measurement, and valuation) can be practically and usefully applied to agroforestry systems at the farm scale. The thesis shows that NCA concepts can be used to highlight the wide range of values provided in agroforestry systems, thereby building a more holistic business case for agroforestry that will appeal to a wider range of farmers and investors. Ultimately, this will assist in increasing adoption of agroforestry by farmers. The thesis also shows that application of NCA concepts to agroforestry will assist in informing design decisions (e.g. tree species selection), enabling farmers to design systems which maximise benefits that are important to them. Expanding this approach to consider both private and public values could also be useful in guiding development of policies aimed at enhancing broader benefits of agroforestry, such as biodiversity conservation and climate change mitigation.

Chapter 1: General introduction

1.1 Background

Measurement and valuation of ecosystem services in agricultural systems and landscapes can assist in encouraging appropriate adoption of management practices. This is increasingly important as we seek to balance the needs of production and sustainability into the future.

Projected continued global population growth (United Nations, 2017b), coupled with a rise in per capita consumption of goods and energy across many regions (FAO 2021), will lead to an increase in global demand for agricultural commodities (i.e. food, fibre, fuel) over coming decades. With competing land uses and limited suitable land available for agricultural expansion, it is expected that this demand will be mainly met through the intensification of agricultural production (Smith et al., 2010).

Agricultural production gains to date, whilst impressive, have come at a significant environmental cost. Globally, agriculture is estimated to cause 60 to 70 percent of terrestrial biodiversity loss (CBD, 2014), and accounts for 22 percent of total greenhouse gas emissions (Smith et al., 2014). In Australia, agriculture remains the greatest consumer of inland water and continues to threaten the condition of waterways through nutrient pollution and sedimentation (Jackson et al., 2016). The environmental impacts of agriculture are not only external. For example, agricultural intensification has been linked to decline of invertebrate groups such as pollinators, generating concern around continued provision of ecosystem services that are essential to production (González-Varo et al., 2013). These issues call for greater consideration of the environmental costs and benefits incurred through the supply of agricultural products.

Meeting future agricultural demand without further degradation of the natural resources and processes upon which we depend requires the capacity to assess and value the broad range of services that these resources provide. The natural capital accounting framework, underpinned by concepts of ecosystem service measurement and valuation, provides a foundation for these efforts. Ongoing refinement and adaptation of this framework holds promise for its useful application in a broad range of agricultural contexts (e.g. FAO and UN 2020). The overall aim of this thesis is to explore how this framework can be adapted to support decision-making relating to a particular agricultural land management strategy, agroforestry: the integration of trees into farming systems.

1.2 Landscape change and future thinking

Globally, land scarcity is becoming an increasingly significant issue as agriculture, forestry, conservation, and urban expansion continue to compete for available land (Lambin & Meyfroidt, 2011). Academic discourse on this issue is dominated by a debate between two strategies: ‘land sparing’, which involves intensifying production on existing agricultural land and protecting natural habitats from further conversion to agriculture, and ‘land sharing’, which involves integrating both objectives on the same land (Fischer et al., 2014). The conceptualisation of production and conservation as competing drivers within this debate has shaped much of the recent thinking around the role of trees in agricultural landscapes.

Viewing the landscape through a biodiversity conservation lens, advocates of land sparing argue that we should remove habitat resources from active farming areas to maximise their productive potential and focus on protecting larger areas of undisturbed habitat (e.g. Green et al., 2005). Advocates of land sharing suggest that integrating habitat into farms would provide greater ecological benefits, even if this requires us to increase our overall agricultural footprint to account for reduced productivity (e.g. Perfecto & Vandermeer, 2010). From the perspective of those focused on agricultural productivity, land sparing may also appear to be the best option, as trees in the landscape can sometimes restrict technology that improves farm profitability (e.g. large seeding rigs and centre pivot irrigation). However, if the goal is to continue producing agricultural commodities over the long term, it is necessary to consider the sustainability and risks associated with removing or adding trees in agricultural landscapes.

In addition to providing and supporting biodiversity, trees deliver important ecosystem services (e.g. erosion control, water quality maintenance, and microclimate regulation) which are critical to the long term function, resilience, and liveability of agricultural landscapes (Jose, 2009). While the two drivers of productivity and conservation have shaped much of the land sharing vs land sparing debate, a potentially more useful lens through which to assess land management strategies is one that considers provision of ecosystem services necessary to sustain our activities in agricultural landscapes in the long term. This thesis explores the use of ecosystem service measurement and valuation, within the natural capital accounting framework, as a means to demonstrate the role of trees on farms as important components of multifunctional, productive, and sustainable agricultural landscapes.

1.3 Agroforestry

The integration of woody perennial vegetation (i.e. trees, shrubs) with agricultural crops and/or animals in the same land management unit is described as ‘agroforestry’ (Reid & Wilson, 1985). Agroforestry can be implemented in various designs with common forms including shelterbelts, alley cropping, widely-spaced trees, riparian buffers, and integrated remnant vegetation. Well-designed agroforestry systems can enhance provision of ecosystem services (e.g. microclimate regulation, erosion control, carbon sequestration) to achieve production of food, fibre, and fuel alongside environmental outcomes (e.g. water quality maintenance, climate change mitigation) (Smith et al., 2012).

While focus has historically been on the use of agroforestry as a tool for agricultural subsistence and development in tropical regions, attention has turned more recently to its broader application (i.e. in temperate, industrialised agricultural landscapes) as a strategy to address parallel challenges of sustainable intensification, land scarcity, and climate change mitigation (Agroforestry Network, 2018; Lewis et al., 2019). The promise of agroforestry as a solution to these issues is recognised at a global level by the Food and Agriculture Organisation of the United Nations (FAO, 2019) and at national levels through its inclusion in numerous agri-environment incentive schemes (e.g. Rural Payments Agency, 2021). Despite persistent enthusiasm for agroforestry at these levels, agroforestry adoption remains unrealised in many parts of the world, with evidence to suggest that Australia lags particularly far behind in this regard (Zomer et al., 2014).

A potential factor contributing to this lack of adoption is that promotion of agroforestry is often based on its capacity to provide ecosystem services that deliver public benefits, particularly at regional, national, or global scales (e.g. Palma et al., 2007). There is minimal focus on how broad-scale strategies for ecosystem service provision through agroforestry translate to farm/enterprise scales at which tree planting/retention decisions are often made. This is particularly pertinent in western industrialised agricultural landscapes, including in Australia, where natural resource management decisions (although guided by regional and State policy) are made at the scale of individual farming businesses. Despite extensive efforts over the last 40 years to demonstrate the various farm-scale benefits of agroforestry (e.g. Powell, 2009), recent research undertaken in Tasmania, Australia, suggests that a large proportion of farmers still consider agroforestry to be an uneconomic proposition (Fleming et al., 2019). To increase agroforestry adoption in temperate, industrialised agricultural landscapes, there is a need for novel approaches to demonstrate the broad economic benefits of agroforestry at fine scales (i.e. the farm or paddock scale). This thesis explores the use of natural capital accounting concepts as a means to highlight these benefits and inform optimal design of agroforestry systems. While this thesis focuses on agroforestry in an Australian context, challenges and opportunities relating to agroforestry adoption in Australia are common to other temperate industrialised agricultural landscapes.

1.4 Natural capital accounting to support decision-making

'Natural capital' describes the stock of renewable and non-renewable resources (e.g. plants, animals, air, water, soils, and minerals) that combine to yield a flow of benefits to people (Jansson et al., 1994). All businesses, including agricultural businesses, depend either directly or indirectly on natural capital and the benefits that it yields. Because interactions between businesses and natural capital may not immediately affect prices or cash flows, impacts and dependencies on natural capital are often overlooked. Natural capital accounting (NCA) provides information on stocks and flows of natural capital in a given ecosystem or region, in physical or monetary terms (Natural Capital Coalition, 2016). This information enables us to measure and track the condition of natural capital and examine how actions affect its capacity to provide goods and services on an ongoing basis. A detailed review of NCA concepts and their application to agroforestry is provided in Chapter 2 of this thesis.

Although the NCA framework is founded on over 40 years of research in ecosystem services and environmental economics, the last decade has witnessed rapid development of methodologies to support its application in a range of contexts. For example, initially released in 2012 and continually expanding in scope, the System of Environmental Economic Accounting (SEEA) is an internationally-agreed framework for integration of economic and environmental data that serves as a foundation for constructing natural capital accounts (United Nations et al., 2014). Although it is possible to adapt the SEEA for application at a range of scales, it has been most widely applied at national or regional levels. For example, the Australian Bureau of Statistics is using the SEEA to guide ongoing development of Australian Environmental-Economic Accounts (AEEA), which will include physical and/or monetary stocks relating to land, oceans, ecosystems, and waste (DAWE, 2019).

More recently, fuelled by changing consumer sustainability standards and awareness of the financial risks of natural capital degradation, interest in the application of NCA at the level of individual businesses or institutions has grown. This transition from national to business-scale assessments of natural capital requires adaptation of existing NCA concepts and methods (reviewed in more detail in Chapter 2). In agriculture specifically, government and agribusiness lenders/insurers are seeking methods to assess natural capital impacts and dependencies in order to design schemes that reward farmers for addressing associated risks to their business and the environment (e.g. DAWE, 2021a; National Australia Bank, 2019). Where existing regulatory frameworks have largely failed (Hamman et al., 2021), such schemes offer a more flexible, farm-scale approach to protection of natural capital in agricultural landscapes, with greater integration of public and private interests. While frameworks for farm-scale NCA have been proposed (e.g. Ogilvy, 2015), efforts are still ongoing to develop practical methods that accurately capture interactions between farming businesses and their various forms of natural capital (i.e. soil, water, vegetation, biodiversity).

In the context of an agroforestry system, the tree and/or shrub component is considered one form of natural capital on the farm. These agroforestry 'assets' provide a range of ecosystem services (e.g. microclimate regulation or carbon sequestration), which in turn deliver various benefits (e.g. productivity improvements or climate change mitigation) to different beneficiaries (e.g. the farmer or the public). There are two key ways in which it could be useful to consider agroforestry through this NCA framework. The first is in encouraging greater adoption of agroforestry either directly, by demonstrating the broad range of values that agroforestry assets can provide to individual farmers, or indirectly, by incorporating agroforestry assets into government or private incentive schemes for natural capital enhancement. The second is in informing optimal design of agroforestry systems, by demonstrating the impacts of agroforestry asset 'condition' (i.e. the structure, composition, and configuration of vegetation) on delivery of farm-scale benefits. This thesis focuses on application and adaptation of NCA concepts to achieve these purposes.

1.5 Aims

The overall aim of this thesis is to examine whether concepts of NCA can be usefully and practically applied to improve farm-scale decision-making and encourage adoption of agroforestry. To address this aim, the thesis examines three main questions:

1. Can measurement and valuation of multiple ecosystem services, within the NCA framework, be usefully applied as a novel approach to valuation of agroforestry assets?
2. Can ecosystem service valuation be adapted for application to agroforestry systems at fine scales (i.e. farm or paddock scale)?
3. How can the concept of 'condition' be adapted for application in this context, and is the use of structural characterisation a practical method for assessing the fine-scale condition of agroforestry assets?

Shelterbelts (linear stands of trees and/or shrubs designed to reduce windspeed) are a popular form of agroforestry and a common feature of temperate agricultural landscapes. For this reason, shelterbelts were chosen as the focus agroforestry example of this thesis. While some ecosystem services provided by shelterbelts were found to be well-understood (e.g. wind speed reduction and associated effects on microclimate in adjacent pasture), significant gaps were identified in the evidence base for provision of ecosystem services relating to biodiversity. To address these gaps, biodiversity (specifically invertebrate biodiversity) became the focus of experimental work that explored connections between agroforestry asset 'condition' and ecosystem service provision.

1.6 Overview of chapters

This thesis consists of seven chapters. Chapter 2 has been published in the journal *Forests* (Marais et al., 2019). Chapter 3 is under review by the *Journal of Agriculture, Ecosystems and Environment*. All chapters have been developed for publication with the assistance of co-authors. My contributions and those of my co-authors are listed at the beginning of each chapter. For all chapters, I was the lead author and developed the experiments and analyses with the help of co-authors. Publications have been modified slightly for inclusion in this thesis.

In Chapter 2, I review the literature to determine whether existing NCA tools and concepts can be usefully applied in the context of farm-scale agroforestry. Shelterbelts are introduced as the focus agroforestry example of the thesis.

In Chapter 3, I examine how shelterbelt vegetation structure and ecosystem service provision are affected by tree species choice and age. This chapter introduces structural characterisation a practical method for assessing the fine-scale condition of agroforestry assets.

In Chapter 4, I examine how species choice affects local invertebrate community composition within shelterbelts. This chapter addresses key gaps in the evidence base (identified in Chapters 2 and 3) for effects of agroforestry asset condition on biodiversity at fine scales.

In Chapter 5, I examine how species choice in shelterbelts affects the distribution of functionally-important invertebrates in adjacent pasture. While Chapter 4 focuses on invertebrate biodiversity within the shelterbelts themselves, this chapter focuses on the provision of invertebrate-related ecosystem services (e.g. pollination and pest control) in areas adjacent to shelterbelts.

In Chapter 6, I explore how farmer preferences for ecosystem services and agroforestry design attributes influence farm-scale decision-making.

In Chapter 7, I synthesise findings from Chapters 2-6 to address the three aims of the thesis and determine whether NCA is an effective tool for improving farm-scale decision-making and encouraging adoption of agroforestry.

Chapter 2: A natural capital approach to agroforestry decision-making at the farm scale

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2.1 Abstract

Agroforestry systems can improve the provision of ecosystem services at the farm scale whilst improving agricultural productivity, thereby playing an important role in the sustainable intensification of agriculture. Natural capital accounting offers a framework for demonstrating the capacity of agroforestry systems to deliver sustained private benefits to farming enterprises, but traditionally is applied at larger scales than those at which farmers make decisions. Here we review the current state of knowledge on natural capital accounting and analyse how such an approach may be effectively applied to demonstrate the farm-scale value of agroforestry assets. We also discuss the merits of applying a natural capital approach to agroforestry decision-making and present an example of a conceptual model for valuation of agroforestry assets at the farm scale. Our findings suggest that with further development of conceptual models to support existing tools and frameworks, a natural capital approach could be usefully applied to improve decision-making in agroforestry at the farm scale. Using this approach to demonstrate the private benefits of agroforestry systems could also encourage adoption of agroforestry, increasing public benefits such as biodiversity conservation and climate change mitigation. However, to apply this approach, improvements must be made in our ability to predict the types and amounts of services that agroforestry assets of varying condition provide at the farm or paddock scale.

2.2 Introduction

2.2.1 Background

The projected increase in global demand for agricultural commodities is expected to be met mainly through the continued intensification of agricultural production (Smith et al., 2010). Production gains to-date have placed pressure on stocks of natural capital and the ecosystem services that they provide (Mackay, 2008; Parris, 2011; Sánchez-Bayo & Wyckhuys, 2019). Future strategies for intensification must balance the need to increase yields with objectives such as climate change mitigation and adaptation, improved soil and water management, and the protection of ecosystem services that support production (World Resources Institute, 2018). Agroforestry is one land management strategy that farmers could employ to meet this challenge. Agroforestry describes any land-use system, practice, or technology, where woody perennials are integrated with agricultural crops and/or animals in the same land management unit (e.g. shelterbelts, alley cropping, integrated remnant vegetation) (Reid & Wilson, 1985). Proponents of agroforestry describe it as a ‘win-win’ approach, as carefully designed systems can balance the production of food, fibre, and fuel while restoring natural capital and thereby enhancing the provision of ecosystem services (e.g. erosion control, microclimate regulation) (Smith et al., 2012). Increasing forest cover is also the cheapest and most direct method to reduce atmospheric concentration of greenhouse gases (Bastin et al., 2019), and while most of this is likely to occur on land unsuitable for agriculture, agroforestry has been recognised as an important component of this reforestation effort (Lewis et al., 2019).

Although the benefits of agroforestry systems are well-researched, adoption of agroforestry in temperate developed agricultural systems, particularly in Australia, remains constrained (Black et al., 2000; Stewart, 2009). While technical, social and policy impediments exist (Race & Curtis, 2007), studies have shown that the perceived economic value of trees is often an important determinant of a farmer’s decision to adopt agroforestry (Cary & Wilkinson, 1997; Fleming et al., 2019). Clear demonstration of the capacity of agroforestry systems to deliver long-term economic benefits to the farm enterprise may therefore improve levels of uptake (Pannell, 1999), which could increase delivery of public benefits such as biodiversity conservation and climate change mitigation. Concepts that capture both commercial and non-commercial benefits, such as the valuation of ecosystem services as part of a broader natural capital accounting approach, may be useful tools in this regard. These concepts may also be useful for developing tools that improve agroforestry-related decision-making at the farm scale (i.e. deciding what type of agroforestry system best suits the objectives of the enterprise). This review considers how a natural capital approach, which has traditionally been applied at national or regional scales, may be practically applied to demonstrate the value of agroforestry systems and improve agroforestry decision-making at the farm or paddock scale.

2.2.2 Natural capital and agriculture

Natural capital is the stock of renewable and non-renewable resources (e.g. plants, animals, air, water, soils, and minerals) that combine to yield a flow of ecosystem services, which in turn provide a variety of benefits to people (Atkinson & Pearce, 1995; Jansson et al., 1994). All industries depend to some extent on natural capital and its benefits, and most businesses also impact on natural capital through their operations or use of products. Primary industries are particularly reliant on stocks of natural capital. In the case of agriculture, producers manage stocks of natural capital to deliver provisioning services in the form of food and fibre. At the same time, management activities may affect the capacity of the same natural capital to provide services into the future. Because interactions between agricultural businesses and natural capital may not immediately affect market values, cash flows, or prices, impacts and dependencies on natural capital are typically considered externalities and are often under-valued or not considered at all in valuation. Intensified production coupled with a failure to account for impacts on natural capital has led to the depletion of natural capital stocks (e.g. soil, biodiversity, water, vegetation) across many of the world's agricultural landscapes (CBD, 2014; Jackson et al., 2016; Smith et al., 2014).

To address this, approaches that account for impacts and dependencies on natural capital have recently been developed (Natural Capital Coalition, 2016; United Nations et al., 2014; Wentworth Group, 2016). Building on several decades of environmental economics research (Pearce, 2002), natural capital accounting provides information on the stocks and flows of natural resources in a given ecosystem, region, or indeed enterprise, in physical or monetary terms. This information facilitates measurement and tracking of natural capital and an examination of how actions inhibit or improve its capacity to generate goods and services on an ongoing basis. Most natural capital accounting work that has been undertaken to-date focuses on valuing natural capital stocks for the purpose of conserving biodiversity at global, national, and regional scales (TEEB, 2010; The World Bank, 2017; United Nations et al., 2014). While interest in the application of natural capital accounting to agriculture is increasing, particularly with the recent release of The Economics of Ecosystems and Biodiversity (TEEB) AgriFood report (TEEB, 2018), the System of Environmental-Economic Accounting for Agriculture, Forestry and Fisheries (FAO, 2016), and the Natural Capital Finance Alliance Agriculture Sector Guide (Ascui & Cojoianu, 2019), the concept is rarely applied in the context of farm-scale decision-making.

When applied to agriculture at the farm scale, natural capital accounting can be used to determine the nature and magnitude of a farming operation's impacts and dependencies on natural capital and the associated business risks and opportunities (Ascui & Cojoianu, 2019; FAO, 2015; Natural Capital Coalition, 2016). This can help farmers and investors identify the specific types and levels of farming activity that pose material risks in terms of impacts or dependencies on natural capital. Conversely, the same approach can be used to identify management interventions that reduce these risks. In the case of agroforestry, there may be unexploited potential to increase adoption by using natural capital accounting to demonstrate farm-scale benefits or avenues for risk mitigation. Where sufficient information is available, these concepts can also be applied to compare the benefits of alternative agroforestry scenarios at the paddock or farm scale (Section 2.4).

2.2.3 Approach

In Section 2.3, we consider how the natural capital accounting framework could be applied to demonstrate the economic benefits of agroforestry at the farm scale and whether existing methods for quantifying and valuing ecosystem services are suitable in this context, based on a review of:

1. The conceptual framework for natural capital accounting (Section 2.3.1);
2. Methods for quantifying ecosystem services at the farm scale (Section 2.3.2);
3. Methods for valuing ecosystem services at the farm scale (Section 2.3.3).

In Section 2.4 we discuss how natural capital accounting may be usefully and practically applied to improve farm-scale agroforestry decision-making (Sections 2.4.1, 2.4.2). We present an example of a conceptual model that could be used to this effect (Section 2.4.3). This conceptual model is based on the findings of existing reviews on ecosystem services in agroforestry systems, as well as direct references from farmers. We also highlight the challenges and opportunities presented by this decision-making approach and suggest areas for further research (Section 2.4.3).

2.3 Natural capital at the farm scale

2.3.1 Applying the natural capital accounting framework to agroforestry

The conceptual framework underpinning natural capital accounting (Figure 2.1) consists of natural capital *assets* which, depending on their condition, provide a flow of ecosystem *services* from which we derive value in the form of *benefits* to business and society. In the context of agroforestry systems, the asset is the integrated 'woody' component, e.g. shelterbelts, woodlots, or integrated remnant vegetation. Ecosystem services and benefits provided by these assets are likely to be numerous and diverse and will depend on the condition of the vegetation (e.g. composition, structure, configuration) (Czúcz et al., 2019).

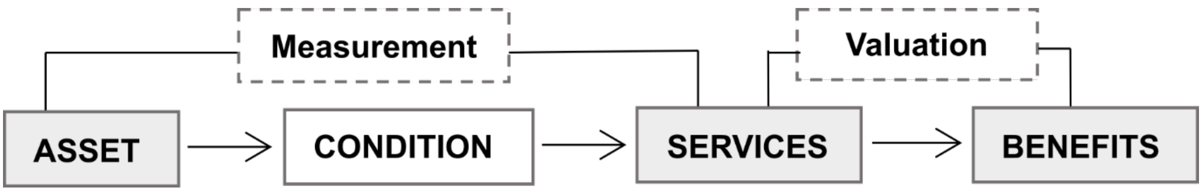

Figure 2.1: Flowchart adapted from the Natural Capital Protocol (Natural Capital Coalition, 2016) illustrating the relationship between a natural capital *asset*, the *condition* of that asset, the ecosystem *services* that flow from the asset, and the *benefits* that those services provide to people.

Identification of ecosystem services and the benefits that they yield is central to the natural capital accounting framework. To reduce inconsistencies in measurement and valuation of services due to omission and/or double counting, the concept of 'final ecosystem services' (Boyd & Banzhaf, 2007) has been developed within ecosystem accounting frameworks. 'Final services' are directly obtained by specific human beneficiaries and are distinct from ecosystem functions/processes, or 'supporting services' (e.g. photosynthesis) (Boyd & Banzhaf, 2007; Daily, 1997; De Groot et al., 2002; Fisher et al., 2009; Haines-Young & Potschin, 2018; Millennium Ecosystem Assessment, 2005; Nahlik et al., 2012; Wallace, 2007). Although the term 'final services' has been retained there is growing consensus among experts that, to reflect the role that they play in producing final services, 'intermediate' services (e.g. pollination) must also be considered in ecosystem accounting (United Nations, 2017a). This is an important development in the context of agroforestry systems, as most of the services provided by agroforestry assets are considered intermediate. Although the debate on ecosystem accounting approaches and ecosystem service classification is ongoing (Obst et al., 2019), coverage of this debate is beyond the scope of this review. Rather, current classification concepts are used in this review to identify relevant ecosystem services for the purpose of discussing the merit of valuing these services to aid in farm-scale decision-making. The classification system currently used in the System of Environmental-Economic Accounting–Experimental Ecosystem Accounting (SEEA-EEA), Common International Classification of Ecosystem Services (CICES) (Haines-Young & Potschin, 2013), applies a suitably broad interpretation of final ecosystem services, which includes several intermediate services and is therefore well-suited to agroforestry systems. An example of the application of CICES (V5.1) classification is provided below (Table 2.1) for a list of services compiled from several reviews on agroforestry ecosystem services (Asbjornsen et al., 2014; Jose, 2009; Smith et al., 2012). The CICES system provides an efficient means of identifying and classifying ecosystem services in an agroforestry context, reduces double-counting, and allows for inclusion of the full range of services described in the cited reviews.

Table 2.1: Farm-scale ecosystem services provided by agroforestry assets. Services are adapted from CICES V5.1 (Haines-Young & Potschin, 2018).

Section	**Group**	**Service**
Provisioning	Cultivated terrestrial plants for nutrition, materials, or energy	Cultivated trees or shrubs grown for nutritional purposes (food), livestock fodder, fibres and other materials (timber), or energy (fuel)
Regulation & Maintenance	Mediation of wastes/toxic substances by living processes	Sequestration of atmospheric carbon
	Mediation of nuisances of anthropogenic origin	Noise attenuation Visual screening Mediation of olfactory nuisance
	Regulation of baseline flows and extreme events	Control of erosion rates Hydrological cycle and water flow regulation (including flood control) Wind protection
	Lifecycle maintenance, habitat, and gene pool protection	Pollination (habitat for pollinators)
	Pest and disease control	Pest control (habitat for pest-predators)
	Regulation of soil quality	Decomposition and fixing processes and their effect on soil quality
	Water conditions	Regulation of the chemical condition of freshwaters through run-off control and nutrient uptake by trees and shrubs
	Atmospheric composition and conditions	Regulation of temperature and humidity, including ventilation and transpiration
Cultural	Physical and experiential interactions with natural environment	Characteristics of agroforestry systems that enable activities promoting health, recuperation, or enjoyment through active or immersive interactions or passive or observational interactions
	Intellectual and representative interactions with natural environment	Characteristics of agroforestry systems that are resonant in terms of culture or heritage or enable aesthetic experiences
	Other biotic characteristics that have a non-use value	Characteristics of agroforestry systems that have an existence value or an option or bequest value

While there is a good understanding of the services that can be provided by agroforestry systems (Table 2.1), measurement or valuation of these services at the farm or paddock scale has been more limited. However, research in this area is developing rapidly, and there have been several recent studies that value a combination of private and public ecosystem services at the farm scale (Campos et al., 2019; Kay et al., 2019; Ovando et al., 2016). In Sections 2.3.2 and 2.3.3 we consider the current methodologies for both measurement and valuation to determine their application to agroforestry at the farm scale.

2.3.2 Measuring ecosystem services at the farm scale

Measurement of ecosystem services (Figure 2.1) is often a pre-requisite to their valuation (Alkemade et al., 2014). High demand for information to support decision-making in resource management has stimulated rapid progress in the development of approaches to measuring ecosystem services (Burkhard et al., 2013; Crossman et al., 2013). Here we provide an overview of the leading methods and tools for measuring ecosystem services and their suitability in the context of farm-scale measurement of services provided by agroforestry assets (see Table 2.1).

Availability and quality of primary data varies between different ecosystem services, but for many services, a lack of data is the most significant constraint to their quantification (Burkhard et al., 2009; Eigenbrod et al., 2010). As a result, most quantitative estimates of ecosystem service provision at the landscape scale are based on secondary data or spatial proxies, which tend to be derived from either topographical data or land use land cover (LULC) datasets (Egoh et al., 2012; Martínez-Harms & Balvanera, 2012). While estimates based on LULC proxies are useful for broad or rapid assessments over large areas (Metzger et al., 2006; Naidoo et al., 2008), they are generally unsuitable for fine-scale (e.g. farm-scale) assessments as the coarse resolution of LULC data may not account for actual spatial variability in biophysical measurements of ecosystem services (Eigenbrod et al., 2010). Importantly for farm-scale agroforestry assessments, readily available remotely-sensed LULC data often fail to capture fine-scale landscape features such as shelterbelts and individual trees, which provide important ecosystem services at smaller scales. Use of LULC proxies also requires well-established links between land cover and ecosystem service provision. At a fine scale, ecosystem service provision is highly dependent on the condition of the natural capital (e.g. vegetation structure and composition). Although resolution of LULC data is improving, many aspects of condition remain difficult to establish from remotely-sensed land cover data. This makes proxy-based techniques particularly unsuitable for farm-scale agroforestry assessments, where the condition of the asset (e.g. the configuration and height of a shelterbelt) has a significant influence on provision of key services (e.g. wind speed reduction).

One alternative to proxy-based measurement is the use of models that can capture processes at finer scales (Volk, 2013). Models consider a wider set of local ecological variables as inputs and are therefore more reliable for fine-scale assessments, compared to LULC proxy-based measurement. One widely applied fine-scale modelling tool is InVEST: Integrated Valuation of Ecosystem Services and Trade-offs (Kareiva, 2011). InVEST estimates levels of ecosystem services and their economic value using a suite of models ranging in complexity from proxy-based mapping e.g., carbon sequestration, to complex site-specific process models, e.g. pollination services (Lonsdorf et al., 2009). Its ability to capture relatively fine-scale processes makes InVEST a potentially useful tool for measuring agroforestry ecosystem services at the farm scale, although to our knowledge, it has yet to be used for such purposes. Several other advanced models exist that cater specifically for agroforestry systems, although they focus primarily on provisioning services and typically require a high degree of technical competency, e.g. CABALA, Farm Forestry Toolbox, for predicting quantities of timber/fibre; Yield-SAFE, SCUAF, APSIM, for predicting crop growth with tree interactions; and SPIF, for timber and environmental outcomes (Battaglia et al., 2004; Ensis, 2006; Keating et al., 2003; Private Forests Tasmania, 2008; van der Werf et al., 2007; Warner, 2007; Young et al., 1998).

To improve the breadth and usefulness of fine-scale models, we first need to improve our understanding of how different natural capital assets influence ecosystem service inflows to agricultural systems and how the condition of these assets affects the types and amounts of services provided. Simple field measurements could then be used as either direct indicators, or model inputs, to accurately quantify multiple ecosystem services. For example, the USDA Forest Service's online toolkit 'i-Tree' contains a series of models that estimate ecosystem services provided by trees based on their physical properties (USDA Forest Service, 2018). Using simple input requirements, e.g., diameter at breast height, species, total height, alongside environmental and location variables, i-Tree employs a suite of models to forecast the provision of a range of services such as pollution reduction, public health benefits, carbon sequestration, and avoided run-off. While services important in an agroforestry context such as crop/livestock shelter, erosion control, indirect pollination and biological control are not yet included, an approach similar to i-Tree could be taken to quantify ecosystem services provided by agroforestry systems at the farm scale. However, we first need to address gaps in our understanding of how the condition of agroforestry assets (e.g. species composition, height, root depth, and configuration in relation to crops, livestock, and other landscape features) affects the services that they provide. Using these physical characteristics as inputs alongside environmental data, existing models may be able to predict quantities for several services including provisioning services (e.g. timber/fibre, food, and fuel), and regulating services (carbon sequestration, erosion control, and microclimate regulation).

One approach to improve accuracy of existing models is to conduct 'bottom-up' assessments where services are measured directly at the farm or paddock level, providing fine-resolution site-specific information that is directly relevant to the farmer. Sandhu et al. (2008) and Porter et al. (2009) measured biophysical indicators of multiple ecosystem services in order to compare land management techniques based on the value of services that they provide. These studies provide examples of how a wide range of services may be quantified at the farm or paddock scale based on observational data. In the case of cultural services where supply is more closely related to user appreciation than to ecosystem condition, measurement can also be achieved through incorporation of qualitative techniques such as interviews and surveys (Crossman et al., 2010; Ovando et al., 2016; Petz & van Oudenhoven, 2012). Participatory methods could be used in conjunction with biophysical measurement to ensure cultural ecosystem services are adequately represented in farm-scale assessments of agroforestry systems. Although broad uptake of bottom-up approaches is limited by the practical constraints and costs of data collection, they are likely to play an important role in improving the accuracy and relevance of existing models.

For measurement of ecosystem services provided by agroforestry systems at the farm scale, the key is striking an appropriate balance between practicality and the suitability of outputs for decision-making. While rough estimates of ecosystem service supply can be derived relatively easily from an LULC proxy, farmers are generally faced with decisions at finer scales (i.e. the paddock or farm scale) that require more detailed site-specific information. In these cases, use of fine-scale models supported by quantitative and qualitative primary data appears to be the most appropriate approach to measuring a wide range of ecosystem services at the farm scale. While there are many promising techniques and packages that could be applied to agroforestry systems, there are still key gaps that need to be addressed, e.g. quantifying the impact of condition.

2.3.3 Valuing ecosystem services at the farm scale

Once ecosystem services have been quantified, the next step is to determine the extent to which these services are valued by relevant beneficiaries. Ecosystem service valuation may also be conceptualized as the measurement of the dividends or 'ecosystem income' yielded by natural capital (Fenichel et al., 2018; Fisher, 1906; Krutilla, 1967). As described by Fenichel et al. (2018), marginal valuation of natural capital for the purpose of constructing accounts requires an understanding of the links between natural capital, human behaviour, and valued service flows. They identify the importance of political and social institutions in driving the management of ecosystem assets which impact upon the ecosystem income, or flow of value from ecosystem services. They further relate the values of ecosystem income and ecosystem stocks to sustainability at a country level, in essence as a measure of genuine savings (Pezzey & Toman, 2005). However, accounting for the value of stocks of natural assets at this macro level is beyond the scope of this review, which focuses instead on the valuation of ecosystem service flows from agroforestry assets to inform decision-making at the farm or paddock scale. Here we describe methods for economic valuation of relevant ecosystem services (Table 2.2) and discuss different approaches to valuation of agroforestry systems at the farm scale. For the purposes of this review, the value of an asset refers to its Total Economic Value (TEV) (Figure 2.2), which encompasses both 'use' and 'non-use' values (Pearce & Moran, 2013). In this context, TEV is defined as the aggregation of the values of all service flows generated by natural capital both now and in the future (Pearce & Moran, 2013).

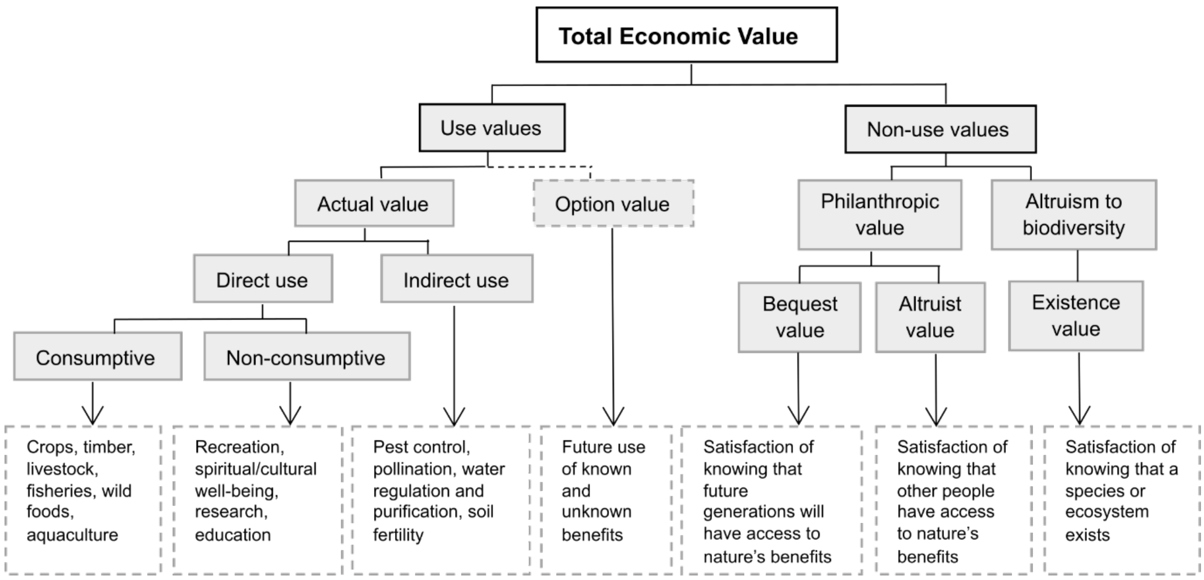

Figure 2.2: Total Economic Value (TEV) typology adapted from Pascual et al. (2010), which classifies values associated with direct use, indirect use, and non-use of service flows generated by natural capital.

An important consideration when valuing ecosystem services is defining the beneficiary. Ecosystem services provided by agroforestry assets can be valued based on the benefits that they provide to the public (e.g. erosion control for improved downstream water quality), to the farmer (e.g. erosion control for retention of soil), or a combination of these approaches. As the purpose of valuation in the context of this review is to demonstrate the long-term benefits of agroforestry to farmers, we reviewed valuation strategies focusing on the farmer as the beneficiary.

It is important to note that valuation pathways of ecosystem services provided directly and indirectly by agroforestry assets vary in complexity (Table 2.2). While agroforestry provides provisioning services that are directly harvested from the trees/shrubs themselves (e.g. food, fodder, fibre, or fuel), agroforestry assets also provide regulating services that indirectly influence other flows of provisioning services on the farm (e.g. increasing lamb survival through regulation of microclimate). In addition, agroforestry assets can also influence stocks of other forms of natural capital (e.g. by providing habitat for insects) which can indirectly influence flows of regulating services such as pollination. Therefore, some valuation pathways lead to monetary values (e.g. market value for provisioning services), whereas others lead to less-tangible forms of value (e.g. farmer well-being). In many cases, particularly where the intention is to justify an investment in agroforestry assets, valuation pathways that lead to a marketable product will form a compelling case. However, non-market values such as amenity, cultural value, and bequest value can also be important drivers for decision-making on farms. Monetary values alone will often fail to capture the full value of an agroforestry asset, which is why it is important to consider a range of ecosystem services that provide a broader perspective of value.

Economic valuation of agroforestry as a land-use system usually takes one of two forms: either a financial analysis of revenues received by the landowner at the enterprise or farm-scale, or an expanded analysis that includes ‘externalities’ or impacts beyond the farm boundaries (Thompson & George, 2009). Although some farm-scale financial analyses include hypothetical payments for regulating ecosystem services or taxes for disservices (e.g. pollution) (Kay et al., 2019; Stainback et al., 2004), non-provisioning ecosystem services are generally not included in traditional farm-scale profitability studies. In studies where regulating and cultural services such as soil protection, carbon sequestration, air quality, and amenity are included, these services tend to be valued with public beneficiaries in mind, rather than as ‘inflows’ to the agricultural enterprise (Kulshreshtha & Kort, 2009). Exceptions do exist, including work by Ovando et al. (2016), in which the private amenity of Mediterranean agroforestry farms is considered to ultimately be ‘consumed’ by the farmer through its effect on land prices (Campos et al., 2019; Oviedo et al., 2017). Despite increasing demand for information in this space, there are still a limited number of studies assessing farm-scale economic benefits of agroforestry systems based on a broad range of use and non-use values, and fewer still that focus on the value of regulating services from a productivity perspective.

Table 2.2: Methods for valuing ecosystem services provided by agroforestry assets at the farm scale with examples of how they might be applied if the farmer is considered the primary beneficiary.

Valuation method	Description	Services (see Table 2.1) that could be valued using this method
Direct market valuation	Where commercial markets exist for services, market prices can be used to represent their value.	Food, fodder, fibre, timber, or fuel from cultivated trees or shrubs. Sequestration of atmospheric carbon
Production function	Where a service plays an intermediate role in the production of a marketable good, production functions can be used to estimate the contribution of that service as a proportion of the market price.	Pollination (habitat for pollinators), e.g. Morse and Calderone (2000). Regulation of temperature and humidity, including ventilation and transpiration. Wind protection.
Averted expenditure	Service is valued based on costs associated with declining benefits due to the loss of that service.	Control of erosion rates, e.g. Wilson (2008). Regulation of the chemical condition of freshwaters through run-off control and nutrient uptake by trees and shrubs. Hydrological cycle and water flow regulation (including flood control).
Replacement cost	Service is valued based on the cost of replacing that service entirely with an artificial or technical solution. This method is often employed to value regulating services in agriculture.	Pollination (habitat for pollinators), e.g., Winfree et al. (2011). Pest control (habitat for pest-predators). Decomposition and fixing processes and their effect on soil quality, e.g. Sandhu et al. (2008), Alam et al. (2014). Control of erosion rates. Regulation of temperature and humidity, including ventilation and transpiration.
Revealed preference: hedonic pricing	Estimates the value of people’s preferences for characteristics of a place based on their contribution to property prices.	Various (potentially difficult to isolate value of individual services) e.g. Polyakov et al. (2015).
Stated preference: contingent valuation or choice experiment	These methods use questionnaires about hypothetical scenarios of environmental change to estimate economic value.	Use and non-use values of a broad range of services including: amenity, cultural heritage, recreation, aesthetics, and existence or bequest value, e.g. Shrestha and Alavalapati (2004).
Benefit transfer	Where resources do not allow for original economic valuation using one of the above methods, it is possible to use data from comparable studies to value services.	Any of the above

Most agroforestry valuation studies that incorporate a broad suite of ecosystem services employ an equally broad suite of valuation methods (e.g. Porter et al., 2009). This usually includes market valuation, avoided expenditure, replacement costs, and some form of stated preference. Benefit transfer is often used for some or all of these valuations, depending on the focus of the study and the resources available to the investigator.

Some agroforestry valuation studies consider multiple beneficiaries, combining private and public perspectives. For example, De Jalón et al. (2018) and Kay et al. (2019) use a range of valuation techniques to compare the productivity and profitability of different agroforestry landscapes against conventional agricultural and forestry systems. Across these studies, the value of sequestered carbon is based on a carbon price, disservices of soil erosion and nitrogen/phosphorus surplus are valued based on the cost of removing these materials from public watercourses, and pollination is valued according to a production function (De Jalón et al., 2018; Kay et al., 2019). Services such as wind speed reduction and noise reduction are excluded despite their potential to deliver significant private benefits to farmers. Valuation studies that combine private and public benefits may be appropriate in some cases, for example when designing payments for ecosystem services. However, the objectives of the agroforestry venture must be clear to ensure that key ecosystem services are included and that the results of the valuation are relevant to the decision-maker.

If the purpose of the valuation is to encourage private investment in agroforestry, it makes sense to focus on ecosystem services that deliver private benefits to farmers and value those services accordingly. Porter et al. (2009) and Alam et al. (2014) take this approach, borrowing techniques used by Sandhu et al. (2008) to value field-scale ecosystem services in agroforestry systems. Production of food and raw materials is valued at market prices; nitrogen regulation, soil formation, groundwater recharge, and pollination are valued according to replacement costs; biological control of pests according to avoided cost of pesticides; and aesthetics through benefit transfer, derived from a contingent valuation study. The broad range of services included in these studies, and the focus on the farmer as the beneficiary in most valuation methods, ensures that the final estimate of each system’s economic value reflects a range of values that are directly relevant to the farmer.

In natural capital accounting, valuation methods should be chosen to suit the purpose of the study and the types of services that are being valued. In the case of agroforestry systems, there is merit in recognising the role of farmers as decision-makers and ensuring that the information produced is directly relevant to them. Strategies for achieving this could include incorporating a broad range of use and non-use values and valuing regulating services from a productivity perspective, rather than as externalities.

2.4 A natural capital approach to agroforestry decision-making at the farm scale

As farmers consider strategies to enhance the long-term productivity of their enterprise while protecting the natural capital base that supports it, they are likely to benefit from the availability of tools that support their decision-making. Here we draw on findings from Section 2.3 to discuss the usefulness and feasibility of applying a natural capital approach to farm-scale agroforestry decision-making.

2.4.1 Advantages of a natural capital approach

As demonstrated in Section 2.3, a natural capital accounting framework can be applied to agroforestry systems to establish the value of agroforestry assets at the farm scale. The framework identifies links between stocks of natural capital, ecosystem service provision, and farm-scale benefits (value). Farmers who conceptualise their farm in this way and understand these links may be more inclined to adopt strategies that protect or enhance natural capital. Natural capital accounting can therefore be useful in justifying private investment in agroforestry. Farmers may also choose to communicate their awareness and management of natural capital impacts and dependencies to internal or external stakeholders to attract new investors or customers. Indeed, agribusiness lenders are showing increasing interest in using natural capital approaches to account for the value of natural capital stocks in farm valuations and credit risk assessments (National Australia Bank, 2019).

The natural capital approach also highlights the flexibility of agroforestry systems, i.e., that they can be designed to deliver a range of benefits depending on the objectives of the farm enterprise. Farmers who are looking to adopt agroforestry will be faced with decisions about the type, extent, location, and configuration of agroforestry assets. Natural capital approaches can be used to compare the benefits of different agroforestry options, in terms of the value of the ecosystem services that each might provide. In this way, there is potential for the natural capital framework to be used as the basis for the development of tools that assist farmers in choosing between alternative agroforestry scenarios based on costs and benefits to the enterprise (Section 2.4.3).

2.4.2 Existing frameworks for natural capital accounting at the farm scale

While general awareness of the role of natural capital in agriculture is increasing (Cojoianu & Ascui, 2018), the concept is rarely applied in the context of farm-scale decision-making. There are still relatively few studies that attempt to value or account for stocks of natural capital at a scale that is useful for decision-making on farms. Although natural capital accounting is being used broadly to appeal for changes in agricultural practice that will protect the natural capital base (TEEB, 2018), little practical guidance exists for farmers and other practitioners looking to construct accounts of their own. This may be due in part to a lack of consensus on the best approach for farm-scale natural capital assessment and accounting. Here we describe several tools and frameworks that may fill this gap and bring us a step closer to a standardised, practical natural capital approach to farm-scale decision-making.

At the outset, it is necessary to undertake some form of natural capital assessment to understand risks and dependencies relating to natural capital stocks and to gain an appreciation of the value of specific natural capital assets to the farm. The Natural Capital Protocol provides a general approach for natural capital assessments, enabling organisations to identify, measure and value their direct and indirect impacts and dependencies on natural capital (Natural Capital Coalition, 2016). Although the Natural Capital Protocol offers little guidance on how their approach may be implemented in practice, other projects have applied the framework to undertake natural capital assessments in agriculture, e.g. the FAO's report on Natural Capital Impacts in Agriculture, which highlights trade-offs between different farming practices (e.g. organic vs. conventional) based on costs to human health and ecosystems (FAO, 2015). Although some case studies touch on internal benefits, most valuations are not considered from the perspective of the farmer, and this approach is therefore not useful as a template for assessments to support farm-scale decision-making. In a more transferable approach, Ascui and Cojoianu (2019) provide a generic procedure for lenders to undertake farm-specific natural capital credit risk assessments (based on the Natural Capital Protocol). In their approach, biophysical indicators (such as percentage vegetation cover) are valued based on evaluation of risks to the lender, which informs whether credit should be extended to the farmer. Although their approach focuses on the value perspective of the lender, there is scope for this procedure to be used by farmers to prioritise management interventions based on assessment of key risks to their business.

Once natural capital risks, dependencies, and the value of natural capital assets have been established, farmers may wish to track the value or condition of natural capital assets through time to inform decisions around investment and operations. Three frameworks exist that provide a standardised approach to natural capital accounting at the farm scale. These are founded on the SEEA-EEA, which has not yet developed to cover farm-scale accounts but nonetheless provides a framework for tracking changes in the extent, condition, and monetary value of ecosystem assets over time across a given spatial area (United Nations et al., 2014). There is also potential for SEEA-EEA itself to be developed for use at the farm scale in the future. The Wentworth Group's 'Accounting for Nature' method is currently being adapted for use at the farm scale (Wentworth Group, 2016) and focuses on the construction of 'asset condition accounts' which provide information about changes to the condition of assets over time, based on measuring biophysical indicators. The second framework proposes an 'ecological balance sheet' (EBS) that enables the application of accrual accounting principles to ecological assets at the farm scale (Ogilvy, 2015). The advantage of the EBS is that it deliberately incorporates natural capital accounts into the farm's existing accounting system so that financial and environmental performance can be tracked simultaneously. Perhaps the most advanced of the existing frameworks is the 'Agroforestry Accounting System' (AAS) which estimates total income accrued from a range of market and non-market products delivered by agroforestry systems (Campos et al., 2001; Caparrós et al., 2003). While application of the AAS to-date has focused on comparing the value of woodland agroforestry systems to other forest types (Ovando et al., 2016), there is potential for this framework to be applied more broadly: at different scales and for different types of agroforestry systems. Each of these existing frameworks brings us closer to tracking the condition and value of natural capital assets through time at a scale that is useful for decision-making on farms.

Although these frameworks form a sound theoretical foundation for farm-scale natural capital accounting, it is important to recognise that they all rely on evidence-based conceptual models that demonstrate how agricultural systems function. In agriculture, key forms of natural capital may include soils, vegetation, fauna (including livestock and fisheries), and water (Ogilvy, 2015). Although it is conceptually easy to calculate stocks of the asset (woody vegetation) and determine its condition (i.e. age, structure, species composition, configuration, etc.), each form of natural capital yields multiple ecosystem services and disservices that may interact in additive, synergistic, or detractive ways. Many of these services are difficult to quantify, interactions between them are often poorly understood, and condition is rarely tracked. Additionally, there is a gap in our ability to predict the types and amounts of services that assets of varying condition provide at the farm scale, and how these services translate to benefits received by the farmer. While efforts are underway to improve our understanding of the value of some natural capital assets in complex agricultural systems (Dominati et al., 2014), we do not yet have an adequate model for agroforestry assets. Conceptual models must also account for the impact that changes in asset condition have on value, particularly in agroforestry systems where the condition of the asset can significantly affect service provision. Such a model would greatly improve the applicability of existing natural capital accounting tools to farm-scale agroforestry decision-making.

2.4.3 A conceptual model for agroforestry decision-making

A conceptual model for valuation of agroforestry assets may serve multiple purposes: firstly, to establish common understanding of causal pathways for the flow of benefits from agroforestry assets and, secondly, to facilitate rapid assessment of the benefits of various agroforestry options. Here we present an example of a conceptual model for farm-scale valuation of an agroforestry asset (Figure 2.3) and discuss how it may be used as the basis for farm-scale decision-making.

The model in Figure 2.3 illustrates how the framework in Figure 2.1 can be applied conceptually to an agroforestry system where the 'asset' is a shelterbelt, and the farmer is considered the beneficiary. This conceptual model is based on studies describing the ecosystem services provided by agroforestry systems (Asbjornsen et al., 2014; Baker et al., 2018; Jose, 2009; Smith et al., 2012) and was developed in consultation with farmers and colleagues working in the field. This model (Figure 2.3) illustrates benefits in a temperate pasture/livestock system but could be adapted to suit other systems such as dairy or horticulture.

Although many of the services listed in Table 2.1 are featured in the model, some have been adapted or broken down into a series of biophysical processes to highlight interactions and trade-offs within the system. For example, the service of 'regulation of temperature' is captured in the provision of shade and the reduction in wind speed provided by the shelterbelt. Each pathway within the conceptual model linking the asset to a benefit involves a combination of measurement and valuation of one or more ecosystem services. For example, the extent of wind speed reduction caused by the shelterbelt can be measured, as can the resulting effects on evaporation and pasture growth on the leeward side of the shelterbelt (Bird et al., 1992; Cleugh, 1998). Once the relationship between wind speed reduction and pasture yield has been quantified, this service can be valued based on the extent to which the increase in yield reduces costs associated with supplementary feeding and the positive effect that this has on gross profit margin. Depending on the situation, the effect of competition may also be measured, and the associated pasture yield decrease accounted for. Potential valuation pathways in the conceptual model will vary considerably in terms of methods and complexity.

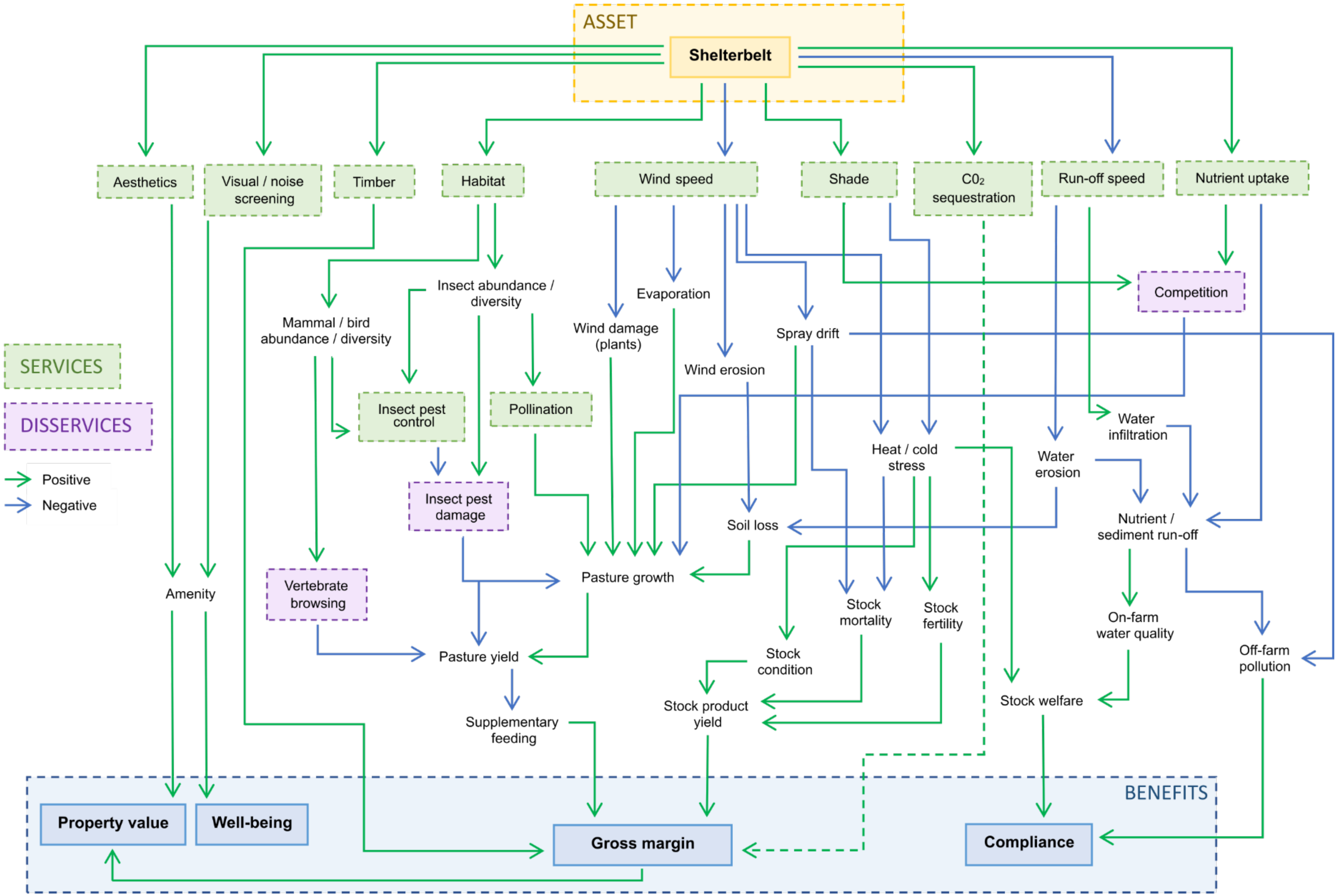

Figure 2.3: Conceptual model for ecosystem services and associated benefits provided by one common type of agroforestry asset (shelterbelt) in a temperate pasture/livestock system. Blue lines represent negative effects (i.e. reduction), and green lines positive effects (dashed where effect is contingent on policy).

From an accounting perspective, the development of conceptual models is an important first step in valuing and accounting for changes in natural capital assets on farms. Conceptual models are useful for establishing common understanding of key causal pathways amongst experts and stakeholders (Olander et al., 2018). In this case, it is useful for practitioners to build an understanding of the multiple ecosystem services that may flow from agroforestry assets, and the types of benefits that these services provide. This common understanding will enable more consistent valuation of agroforestry assets in accounting exercises at various scales (e.g. Accounting for Nature, AAS, SEEA-EEA). Conceptual models can be developed further to include a broader range of beneficiaries (e.g. the general public) and used as a 'blueprint' for valuation to suit a range of purposes. For example, government agencies may use an adapted version of the model in Figure 2.3 to determine the return on investment in agroforestry assets at the farm or landscape scale, considering both private and public benefits. Lenders and investors may also use similar models to conceptualise the value of agroforestry assets from a risk management perspective (Ascui & Cojoianu, 2019). Conceptual models are an ideal tool for this purpose given their flexibility and capacity to clearly communicate relationships within complex systems such as agroforestry systems. These models can be more powerful if underpinned by an evidence-based review (England et al., 2020).

The conceptual model also provides the basis for development of tools that can assist in agroforestry-related decision-making at the farm or paddock scale. Farmers are the primary decision-makers and creating tools that cater for them and the types of decisions that they face is crucial. The farm-scale value of services provided by agroforestry assets may be highly dependent on the location of the farm, the objectives of the farm enterprise, and the context of the asset within the farm (Müller et al., 2019). Farmers require tools that enable them to make decisions about investing in agroforestry systems and designing them in such a way that maximises benefits to their particular enterprise. Conceptual models can enable them to make these decisions without having to undertake complex, expensive natural capital assessments that would require direct measurement and valuation of ecosystem services. For example, a farmer planning to invest in agroforestry would first need to decide what type of asset best suits the objectives of their enterprise. They may seek to maximise provision of services that improve productivity or reduce operational risk while waiting for longer-term returns from marketable wood products. If one of their priorities is to reduce lamb losses due to cold winds they may decide to invest in shelterbelts, based on the benefits demonstrated in a conceptual model of this system (Figure 2.3). The next phase will involve deciding how many shelterbelts to plant, the dimensions and orientation of each shelterbelt, and their location in relation to other elements of the farm. In making these decisions they may refer to other sections of the conceptual model to consider a wider range of potential benefits (e.g. amenity, reducing spray drift) and disbenefits (e.g. competition effects). Used in this way, conceptual models can provide a low-cost, rapid approach to agroforestry decision-making at the farm or paddock scale.

Although the evidence base that supports conceptual models for farm-scale valuation of agroforestry assets is growing (Smith et al., 2012), there are still gaps in our biophysical understanding of agroforestry systems (Baker et al., 2018). While a lack of quantitative evidence may not necessarily restrict the usefulness of these models for farm-scale decision-making, it is helpful to have confidence in the direction of relationships (i.e. positive or negative) and the relative quantities of ecosystem services provided by different types of assets. Where conceptual models currently fall short is in demonstrating the impact of asset condition on the flow of services and benefits. Having chosen to plant shelterbelts, a farmer may eventually have to decide on the configuration and composition of the shelterbelts. They are also likely to be interested in changes to the flow of services and benefits over time, from planting to harvest/senescence. The effect of asset condition at fine scales is an important research gap that must be filled in order to improve the usefulness of these conceptual models.

Where sufficient quantitative evidence exists, conceptual models can also form the basis of more precise, predictive tools for decision-making. These tools may facilitate fine-scale, quantitative valuation of services that are of particular importance to farmers (e.g. shelter). Increasingly, valuation methods are being incorporated into ecosystem service models (e.g. InVEST, i-Tree Eco v6) and economic analysis tools, some of which are designed specifically for integrated farming systems (e.g. Imagine, Farm-SAFE) (Abadi et al., 2003; Graves et al., 2011). Conceptual models can guide the development of these tools by demonstrating the complexity of the system as a whole, ensuring that the tools account for interactions and trade-offs that might otherwise be missed. To improve useability, it may be advantageous to compile all relevant models into a single toolkit (similar in style to InVEST or i-Tree) or to incorporate ecosystem service models into an existing package (e.g. Farm Forestry Toolbox) or farm enterprise platform (e.g. DAS Rural Intelligence Platform, FarmMap4D) (Digital Agriculture Service, 2019; FarmMap4D, 2018; Private Forests Tasmania, 2008). Data accessibility (including cost and usability) is an important consideration in the development of such a toolkit, as a collaborative approach is likely to greatly improve the scope and reliability of outputs.

Conceptual models can enhance the applicability of existing natural capital accounting tools to farm-scale agroforestry decision-making. They can improve consistency in the valuation of agroforestry assets for accounting purposes, guide rapid decision-making at the farm or paddock scale, and form the basis for development of quantitative decision-making tools. To improve the useability of conceptual models in this context, we need to expand the evidence base that supports them with particular focus on the impact of asset condition on ecosystem service provision.

2.5 Conclusions

The natural capital accounting framework provides a logical and increasingly consistent approach to the valuation of impacts and dependencies on natural capital. Findings from this review suggest that there is potential for this framework to be usefully applied to demonstrate the capacity of agroforestry systems to deliver sustained private benefits to farming enterprises.

Despite difficulties in obtaining information for many ecosystem services, tools and models for measuring services continue to advance and improve. In the case of measuring ecosystem services provided by agroforestry systems, the key is striking an appropriate balance between practicality and the relevance of outputs to decision-making. Use of fine-scale models supported by quantitative and qualitative primary data may be the most appropriate approach to measuring a wide range of ecosystem services at the farm scale. While promising advancements continue to be made in the development of tools to model service provision at these fine scales, there are still some key gaps that need to be addressed, e.g. quantifying the impact of condition.

As the evidence base for the value of natural capital in agriculture continues to grow, methods and tools for measuring this value are also improving. Methods for valuing ecosystem services should be chosen to suit the purpose of the valuation and the types of services that are being valued. In the context of demonstrating farm-scale benefits of agroforestry, valuations should be directed at farmers as key beneficiaries, incorporate a broad range of use and non-use values, and value regulating services from a productivity perspective rather than as externalities. Natural capital accounting can be applied to communicate the broad range of values that farmers can derive from agroforestry assets, thereby encouraging appropriate levels of investment.

A natural capital approach can also be applied to assist farmers in making decisions about agroforestry at the farm or paddock scale. While work is currently underway to develop a standardised natural capital approach to farm-scale decision-making, existing tools rely on conceptual models for the provision and valuation of ecosystem services that flow from natural capital assets in agricultural systems. To usefully apply a natural capital approach to farm-scale agroforestry decision-making, we should look to develop adequate conceptual models for agroforestry systems. Underpinned by evidence-based reviews, these models could be useful for improving consistency in the valuation of agroforestry assets, guiding decision-making at the farm or paddock scale, and supporting development of quantitative decision-making tools.

Chapter 3: Shelterbelt species composition and age determine structure: consequences for ecosystem services

This chapter is under review as:

Marais, Z.E., Baker, T.P., Hunt, M.A. and Mendham, D.S. (under review with *Agriculture, Ecosystems, and Environment*). Shelterbelt species composition and age determine structure: consequences for ecosystem services.

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3.1 Abstract

Shelterbelts are a popular form of agroforestry, providing a wide range of ecosystem services (e.g. wind speed reduction and wood production) which deliver farm-scale benefits. Variation in species composition and planting density drives structural differences in shelterbelts which directly influence the provision of ecosystem services and consequently the range of benefits received by farmers. In many cases, the specific structural characteristics of shelterbelts that determine provision of these services have been identified. However, little is known about how these characteristics vary with shelterbelt species composition and age, and how such variation may affect provision of a range of key services and benefits. This study explores the effects of shelterbelt composition and age on structural characteristics that determine ecosystem service provision. Structural characteristics (including vegetation height and porosity) were measured and compared across shelterbelts with three common species compositions (*Eucalyptus nitens*, *Pinus radiata*, mixed native) and three age classes (2-5 years, 6-14 years, 15-30 years) in the Midlands region of Tasmania, Australia. Results showed that species composition and age were key determinants of structural characteristics. For example, height, carbon sequestration, and stand basal area increased and porosity decreased with shelterbelt age, with rates of increase/decrease varying significantly between species compositions. We outlined how these structural characteristics affect provision of ecosystem services and showed that fine scale benefits are highly dependent on the species composition of the shelterbelt. These findings can assist agroforestry practitioners in designing shelterbelts that maximise benefits to their enterprise. There is value in expanding the approach used in this study to develop decision-making tools for practitioners, and to facilitate more meaningful application of natural capital accounting to agroforestry at the farm scale.

3.2 Introduction

Shelterbelts, i.e. linear stands of trees and/or shrubs designed to reduce wind speed, are a popular form of agroforestry and a common feature of agricultural landscapes worldwide. Shelterbelts provide a range of ecosystem services which deliver benefits to agroforestry practitioners (farmers or landholders who practice agroforestry). These services include wind speed reduction, habitat provision, wood production, and carbon sequestration (Asbjornsen et al., 2014; Jose, 2009), and benefits include agricultural productivity improvements and risk mitigation (Baker et al., 2018; England et al., 2020; Smith et al., 2012). While private, farm-scale benefits are often key drivers for establishment, shelterbelts can also provide public benefits including climate change mitigation and biodiversity conservation (George et al., 2012).

Practitioners investing in shelterbelts will seek to maximise specific benefits depending on their enterprise objectives. The capacity of a shelterbelt to deliver benefits over time depends on its long-term condition, which encompasses the dimensions, configuration, species composition, age, and structure of the shelterbelt. Understanding how shelterbelt design choices influence condition, and how this in turn affects the provision of ecosystem services and benefits, will aid practitioners in designing shelterbelts to maximise benefits that align with their objectives.

Structural characteristics that determine various ecosystem services are well-understood (Figure 3.1). For example, the extent of wind speed reduction achieved by shelterbelts is a function of width, length, aerodynamic porosity, vegetation distribution, and height (Cleugh, 1998; Wu et al., 2018). Characteristics such as stem density, under/mid-storey cover, species diversity, and leaf litter are important for supporting birds and invertebrates (McElhinny et al., 2006; Salt et al., 2004). Further, models predicting services such as wood production and carbon sequestration are often based on structural information (Goodwin, 2017; Richards & Evans, 2000).

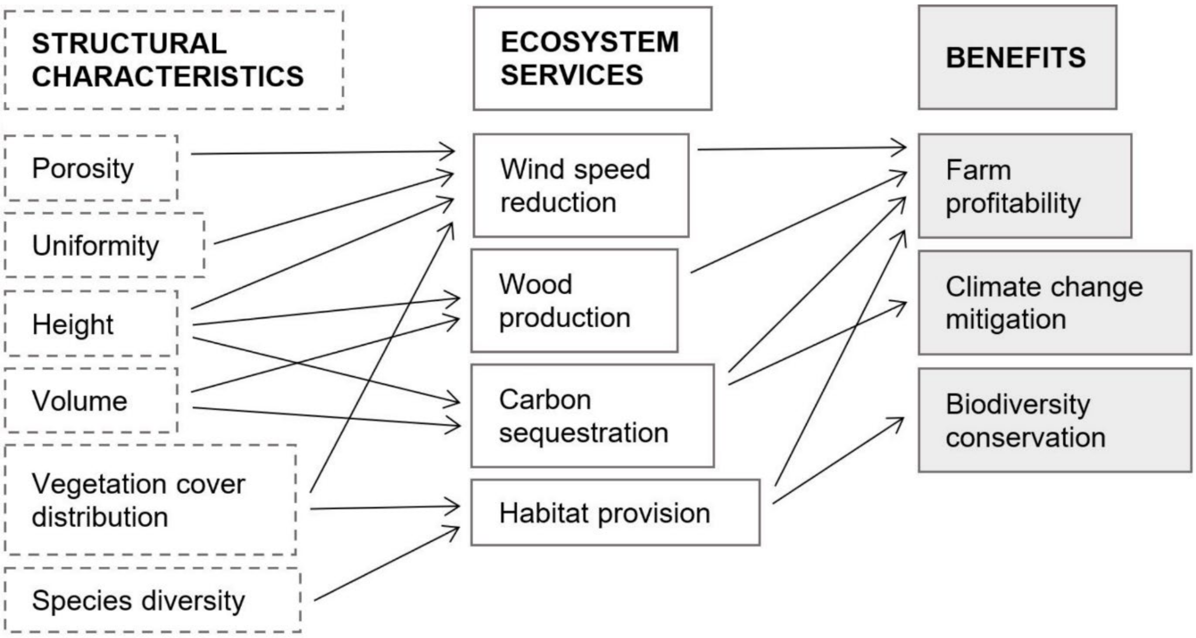


Figure 3.1: Arrows represent established links between shelterbelt structural characteristics measured in this study, ecosystem services, and potential benefits (private and public).

Structural characteristics of a shelterbelt will be determined by many factors, including species composition, but will change over time as the vegetation grows. While links between structural characteristics and provision of many ecosystem services have been established (Figure 3.1), we know relatively little about how structural characteristics vary depending on the species composition and age of a shelterbelt. Understanding the effects of species composition on structure and the subsequent effects on ecosystem services is required to enable practitioners to select optimal species, while understanding the effects of age provides a timeframe for the delivery of potential benefits. In cases where practitioners seek to balance multiple objectives, structural characterisation of shelterbelt composition types also enables an examination of trade-offs and synergies between a wide range of products and ecosystem services.

Understanding how shelterbelt structural differences influence farm-scale benefits also enables more effective application of natural capital assessment and accounting to agroforestry. Natural capital accounting facilitates measurement and tracking of natural capital (e.g. vegetation) and examines how actions influence the capacity of that capital to generate ecosystem services over time (Natural Capital Coalition, 2016). There is increasing interest in using natural capital accounting concepts to demonstrate farm-scale benefits provided by agroforestry assets as one form of natural capital on farms (Marais et al., 2019). One key barrier to effective application of natural capital accounting in this context is a lack of understanding of the links between the condition of the agroforestry asset (e.g. structural characteristics) and the flow of services and benefits at fine scales.

Here we provide a method to characterise and compare a broad range of structural characteristics of shelterbelt species composition types and ages. We then use this information to predict the likely impact that species composition will have on ecosystem service provision through time. To achieve this, we investigate the extent to which shelterbelt composition and age affect key structural characteristics in three common composition types and three age categories of shelterbelt in the Midlands region of Tasmania, Australia. We discuss how results can assist in predicting and comparing the benefits of different types of shelterbelts, and how this approach could be applied to support farm-scale natural capital assessment and accounting.

3.3 Methods

3.3.1 Study region

Thirty-three study sites in the Midlands region of Tasmania, Australia (Figure 3.2) were selected. Study sites included three shelterbelt species composition types that are common to temperate Australia (*Pinus radiata*, *Eucalyptus nitens*, and mixed native) and three age categories (2-5 years, 6-14 years, 15-30 years since planting). Shelterbelt composition types are hereafter referred to as pine (*P. radiata*), eucalypt (*E. nitens*), and mixed native. All study sites were located within an area between 41˚30'S-42˚30'S and 146˚50'E-147˚27'E spanning elevations of 132-373 m a.s.l. This area experiences an annual temperature range of -4 to 32˚C and receives an average annual rainfall of 400–700 mm.

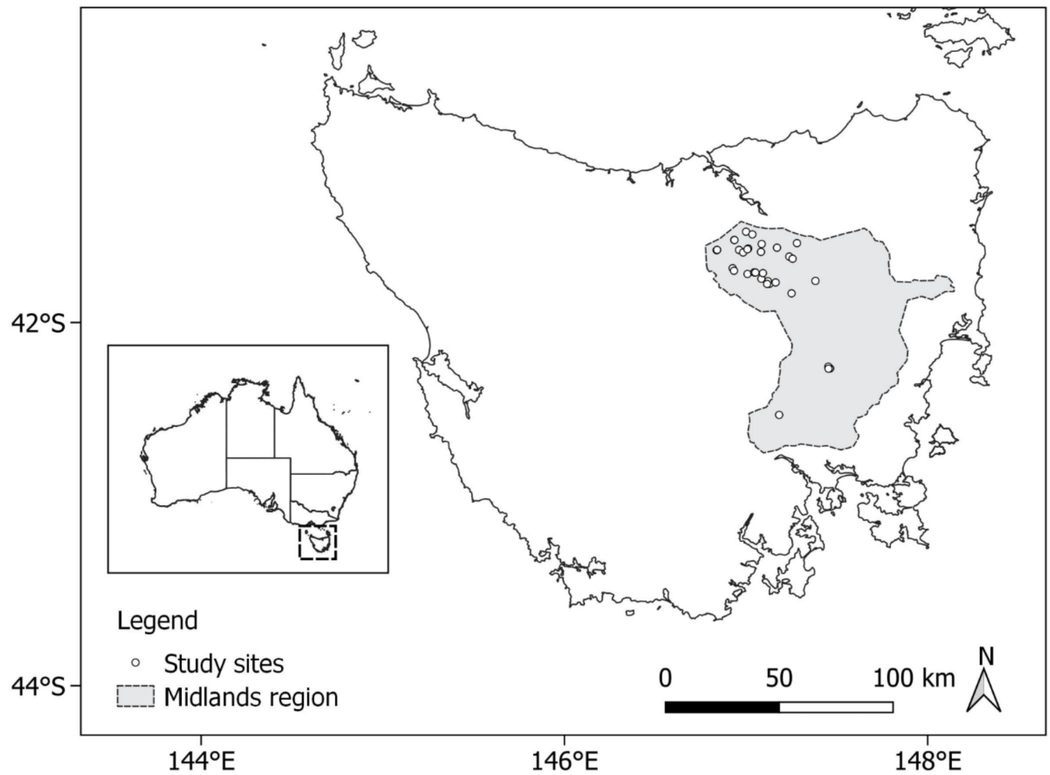


Figure 3.2: Map of Tasmania, Australia, showing the extent of the Midlands region (grey) and location of study sites.

Eucalypt and pine sites were planted as single-species shelterbelts, typically for wood production. Mixed native shelterbelts, typically planted for biodiversity or carbon rather than wood production, varied in their composition but were all dominated by *Eucalyptus* sp. and *Acacia* sp. and contained a total of at least four shrub/tree species. Study sites were chosen that had a single shelterbelt adjacent to pasture, with most shelterbelts oriented approximately perpendicular to the prevailing wind direction. As winds in the study area originate mostly from the west and north, shelterbelts included in this study generally had a south-eastern aspect on the leeward side. Three sites per composition type were used for the 2-5 year and 6-14 year age categories, and five sites per composition type for the 15-30 year age category. Increased replication was used in the older age category as this is when services are most likely to be at their maximum. For one site in the 2-5 year-old pine category, one in the 2-5 year-old eucalypt category, and one in the 6-14 year-old eucalypt category, appropriate shelterbelts could not be found. In these cases, the paddock-facing edges of woodlots were used. Exact ages of each site were recorded if known, otherwise ages were estimated using available historical aerial imagery. All sites in this study received minimal silvicultural management post planting (i.e. pruning or thinning). All shelterbelts were at least 100 m in length and 15-25 m wide (excluding the three woodlot plantings), measured to the edge of the crown. Stock were excluded from all shelterbelts by fencing. All shelterbelts consisted of multiple rows of trees (ranging from 3 to 8 rows), with typical stem densities of 2,500 stems per hectare at planting. Further information on individual study sites is provided in Appendix A.

3.3.2 Data collection

Three sample plots, each 20 m long and as wide as the shelterbelt (varying between 15-25 m), were established within the shelterbelt at each site. Variation in shelterbelt width was similar across sites within each treatment type and age category and was therefore deemed not to have a high level of influence on treatment effects. Plot locations were randomly selected although were a minimum of 20 m apart and 20 m from the shelterbelt ends. For the three woodlot sites, 20 m x 20 m plots were established on the edge bordering the farm paddock.

Structural characteristics were measured at each sample plot including external characteristics (height, optical porosity, uniformity) and internal characteristics (vegetation cover distribution, stand basal area, species diversity) (see Figure 3.1). Measurements were taken during summer/autumn months (December – May) across 2018 and 2019. Although efforts were made to measure sites at a similar time of year to limit seasonal variation, timing in some cases was determined by property access restrictions. Potential for seasonal variation in measured characteristics was also relatively low as dominant species within the shelterbelts included in this study were not deciduous. Characteristics were derived from the following field measurements:

Optical porosity

Optical porosity was only measured for shelterbelts in the 15-30 year age category. Photographs of each sample plot were taken on the leeward side at 30 m from the canopy edge. Images were converted to black and white, and cropped to the mean dominant height of the shelterbelt. Both 'total porosity' (the full height of the shelterbelt) and 'reduced porosity' (the lower two thirds of the shelterbelt) were analysed to account for variability in optical porosity of the upper third of the shelterbelts due to uneven canopy height (Středová et al., 2012). The optical porosity of each horizontal section was assessed using area fraction analysis in the image processing software Image J2 (Rueden et al., 2017).

Vegetation cover distribution

Two 2.5 m radius sub-plots were established within each sample plot: one in the middle of the shelterbelt (equidistant from the leeward and windward canopy edges) and the other starting at the canopy edge on the leeward side. For each sub-plot, vegetation assessments were conducted in line with the TASFORHAB methodology (Peters, 1984). This involved measuring the percent cover of live and dead vegetation at the following height increments: 0-0.1 m, 0.1-0.3 m, 0.3-1 m, 1-2.5 m, 2.5-5 m, 5-8 m, 8-15 m, and 15-27 m. Percent cover was measured by visual inspection to allow for a high volume of rapid, repeated measurements. Measurements were taken by the same recorder each time for consistency.

The percent cover of coarse woody debris (CWD) was recorded separately, with CWD defined as fallen dead trees and branches with a diameter >10 cm at the widest point (Harmon & Sexton, 1996). Debris that did not meet this size threshold was incorporated into the percent cover measurement of dead vegetation in the appropriate stratum.

Height

A height pole was held at the canopy edge at the midpoint of each sample point and observed from approximately 30 m from the canopy edge. 'Mean dominant height' was measured as the average height of dominant canopy tree species within each sample plot.

Diameter at breast height (DBH), basal area, and above ground carbon

DBH of all trees/shrubs in the sample plot was measured over bark, at 1.3 m. Stems under 2 cm DBH and standing/fallen dead trees were excluded. Number of stems per tree was also recorded, and DBH of multi-stemmed trees was taken as the square root of the sum of squares of the DBH of each stem.

Stand basal area (*G*) was calculated from DBH measurements of all stems within the area (*a*) (ha) of each sample plot:

$$G = \frac{\pi}{4000} \times \frac{\sum dbh^2}{a} = 0.0000785398 \times \frac{\sum dbh^2}{a}$$

Species-specific allometric models were used to calculate above-ground biomass of individual trees/shrubs from DBH measurements (Paul et al., 2013). Above-ground carbon (tonnes per hectare) was calculated by totalling the above-ground biomass of individual trees within each sample plot and dividing this by two (Penman et al., 2003).

Uniformity

Over the lifespan of a shelterbelt, 'gaps' may form due to factors such as senescence or windthrow, potentially resulting in wind-tunnelling. The proportion of each shelterbelt comprising gaps (i.e. areas clear of mid and over-storey vegetation) was measured as a percentage of the total length of the shelterbelt. The complement of this percentage was recorded as a measure of uniformity, i.e. 10% gaps = 90 uniformity units.

Species diversity

The total number of plant species, including trees, shrubs, and grasses, was measured at each sample plot.

3.3.3 Data analysis

For each available combination of composition type and age category (three replicates per composition type for 2-5 year and 6-14 year categories, and five per composition type for the 15-30 year category) a summary of average structural characteristics was calculated. Site level means were derived from plot level data for: mean dominant height, stand basal area, uniformity, total porosity, reduced porosity, carbon, species diversity, and cover of CWD. Standard deviations were calculated from site level averages. All statistical analyses were performed within R V4.0.2 (R Core Team, 2020).

For total porosity and reduced porosity, separate one-way analyses of variance (ANOVA) were conducted to test the effect of shelterbelt composition (for the 15-30 year age category only). Site level means were used for these analyses, with a square root transformation applied. For species diversity, a two-way ANOVA was conducted to test the effects of composition type and age, as well as their interaction. Site level means were used for this analysis, with a log transformation applied. Data were insufficient to statistically test the extent of variation between sites due to the limited availability of suitable sites within the study area. For all ANOVA, post hoc pairwise comparisons were undertaken using the 'lsmeans' package in R (Lenth, 2016).

For mean dominant height, stand basal area, and carbon, linear regression models were used to examine relationships between each of these characteristics and shelterbelt age. For these analyses, age was treated as a continuous variable, i.e. exact ages of each shelterbelt were used instead of the three age categories. ANOVA were conducted on the linear models to test the effect of species composition and the interaction between composition and age. Although relationships between these characteristics and shelterbelt age were expected to be non-linear, the limited sampling size and spread of exact ages meant that linear models were the most appropriate option.

To further illustrate the nature of potential changes in growth-related characteristics through time, readily-available models were used to predict mean dominant height, stand basal area, and above-ground carbon. In some cases these models extend beyond the age range of sites included in this study, providing a useful indication of how characteristics are likely to behave in later stages of growth. Mean dominant height and stand basal area for eucalypt and pine composition types were modelled using species-specific models (*E. nitens* and *P. radiata*) in the Farm Forestry Toolbox Version 5.4 (Goodwin, 2017). The Farm Forestry Toolbox is a collection of computer programs designed to assist with measurement and management of small to medium-sized forest estates. Default plantation growth models for Tasmania were used, with scenarios manually adjusted to remove silvicultural management. Modelling capability was not available for the mixed native composition type, however the shelterbelt field measurements for all three composition types were presented alongside the modelled results for comparison.

FullCAM Version 6.19.07.114 (pre-release of 2020 update) (Richards & Evans, 2000) was used to model the above-ground carbon mass of trees per hectare for each composition type over a 25-year rotation, in line with the Carbon Farming Initiative (CFI) methods for new farm forestry plantations (Department of the Environment and Energy, 2016). Below-ground carbon was not included in the outputs as differences between composition types are likely to be negligible and largely site-dependent (Paul et al., 2002; Paul et al., 2003). Spatial data were downloaded through the FullCAM 'Data Builder' using the latitude and longitude of ten of the study sites that represented the range of environmental conditions across the study area. Model outputs were produced per composition type for each of the ten sites, then averaged for each composition type. Within FullCAM, species-specific models for *E. nitens* and *P. radiata* and the model for 'mixed environmental species plantings – belt configuration' were used to represent the respective composition types. Default 'regimes' (management scenarios comprised of individual events) within FullCAM were edited to remove silvicultural management.

Vegetation cover distributions across all strata were examined graphically using histogram plots for each composition type and age category. Graphical analysis was based on site averages. Cover of live and dead vegetation were calculated separately.

To quantify impacts of shelterbelt composition and age on provision of multiple ecosystem services (shelter, wood production, carbon sequestration, and habitat provision), empirical results for structural characteristics were considered in the context of relevant literature to inform a simplified system of scoring with 'points' (+) allocated for low (+), medium (+ +), and high (+ + +) levels of relative benefit (refer to Figure 3.1).

3.4 Results

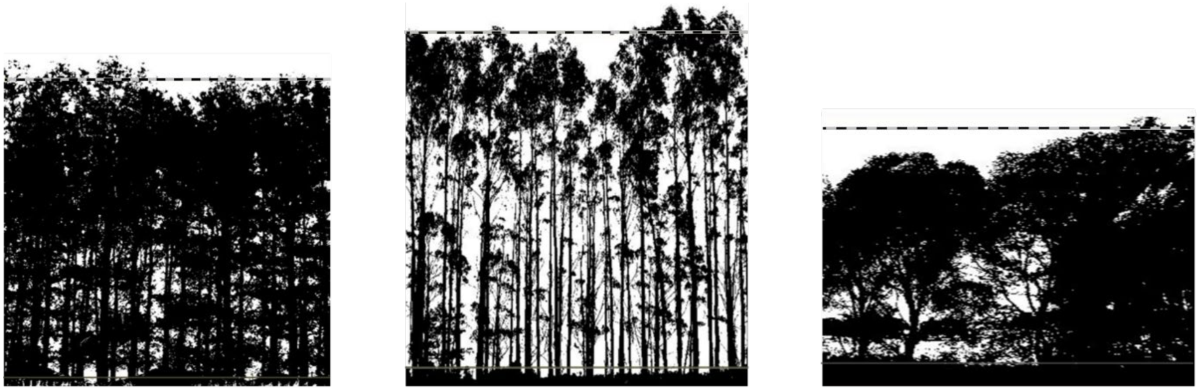
There were clear differences between shelterbelt composition types across all structural characteristics (optical porosity, mean dominant height, uniformity, stand basal area, carbon, species diversity, and CWD cover) (Table 3.1). Mean dominant height, stand basal area, carbon, and CWD also varied temporally (Table 3.1). Details of specific differences relating to each structural characteristic are explained in the subsequent sections (Section 3.4.1- 3.4.7).

Table 3.1: Mean structural characteristics of shelterbelts of different composition types and age categories. Standard deviations are presented in parentheses.

Composition type	Age (years)	Number of sites	Total optical porosity (%)	Reduced optical porosity (%)	Mean dominant height (m)	Uniformity (%)	Stand basal area (m^2 ha^{-1})	Above-ground carbon (tC ha^{-1})	Total number of plant species	Cover of coarse woody debris (%)
P. radiata	2-5	3	-	-	7.1 (1.5)	100.0 (0.4)	16.4 (4.7)	32.6 (10.0)	2.2 (0.4)	0.3 (0.8)
	6-14	3	-	-	9.5 (2.0)	99.9 (0.7)	27.7 (11.0)	60.0 (25.6)	2.8 (0.7)	0.9 (1.2)
	15-30	5	14.5 (2.8)	7.1 (5.0)	13.2 (2.6)	99.6 (0.2)	57.8 (17.2)	132.0 (41.8)	1.9 (0.2)	2.5 (3.0)
E. nitens	2-5	3	-	-	5.3 (1.7)	96.0 (0.4)	9.3 (4.9)	23.6 (13.5)	4.1 (0.4)	1.3 (1.7)
	6-14	3	-	-	13.2 (1.1)	99.4 (0.8)	32.3 (11.1)	98.2 (41.0)	4.1 (0.8)	4.0 (3.8)
	15-30	5	35.5 (8.8)	34.4 (5.8)	17.2 (1.9)	97.7 (1.5)	46.4 (13.1)	161.0 (39.6)	4.8 (1.5)	8.0 (4.9)
Mixed native	2-5	3	-	-	4.4 (0.8)	87.0 (2.0)	3.1 (1.0)	7.7 (3.1)	13.8 (2.0)	0.2 (0.7)
	6-14	3	-	-	6.4 (1.3)	84.9 (1.7)	14.8 (9.9)	43.8 (30.4)	9.3 (1.7)	2.4 (2.4)
	15-30	5	19.1 (2.0)	12.0 (3.4)	8.7 (1.7)	97.0 (2.3)	35.0 (9.2)	125.0 (37.6)	12.1 (2.3)	4.4 (3.2)

3.4.1 Optical porosity

Analyses of variance showed a significant effect of composition type on both total porosity ($F(2,12) = 24.52$, $p < 0.001$) and reduced porosity ($F(2,12) = 37.42$, $p < 0.001$). Post hoc pairwise comparisons showed that for both total and reduced porosity, eucalypt shelterbelts (total: 35.5, reduced: 34.4) had significantly higher porosity than both mixed native (total: 19.1, reduced: 12.0) and pine (total: 14.5, reduced: 7.1) at $p < 0.050$, but mixed native and pine composition types were not significantly different ($p = 0.107$ for reduced, and $p = 0.211$ for total porosity) (Table 3.1, Figure 3.3).



Pinus radiata
14.5(2.8)

Eucalyptus nitens
35.5(8.8)

Mixed native
19.1(2.0)

Figure 3.3: Example images of shelterbelt composition types in the 15-30 year age category. Average total porosity for each composition type is reported with standard deviations in parentheses. The grey and black dashed line indicates mean dominant height.

3.4.2 Vegetation cover distribution

Shelterbelt composition altered the distribution of vegetation cover across the measured strata (Figure 3.4). Distribution patterns within each composition type changed over time, although edge plots showed more variation with age compared to middle plots (Figure 3.4b). Patterns of change in vegetation cover distribution over time were similar for eucalypt and pine shelterbelts, although pine shelterbelts retained higher densities of vegetation across most strata at their edges compared to eucalypt shelterbelts, which explains their lower optical porosity (Section 3.4.1). In general, contrasts in vegetation cover distribution between different composition types were greater in the middle plots, compared to the edge plots.

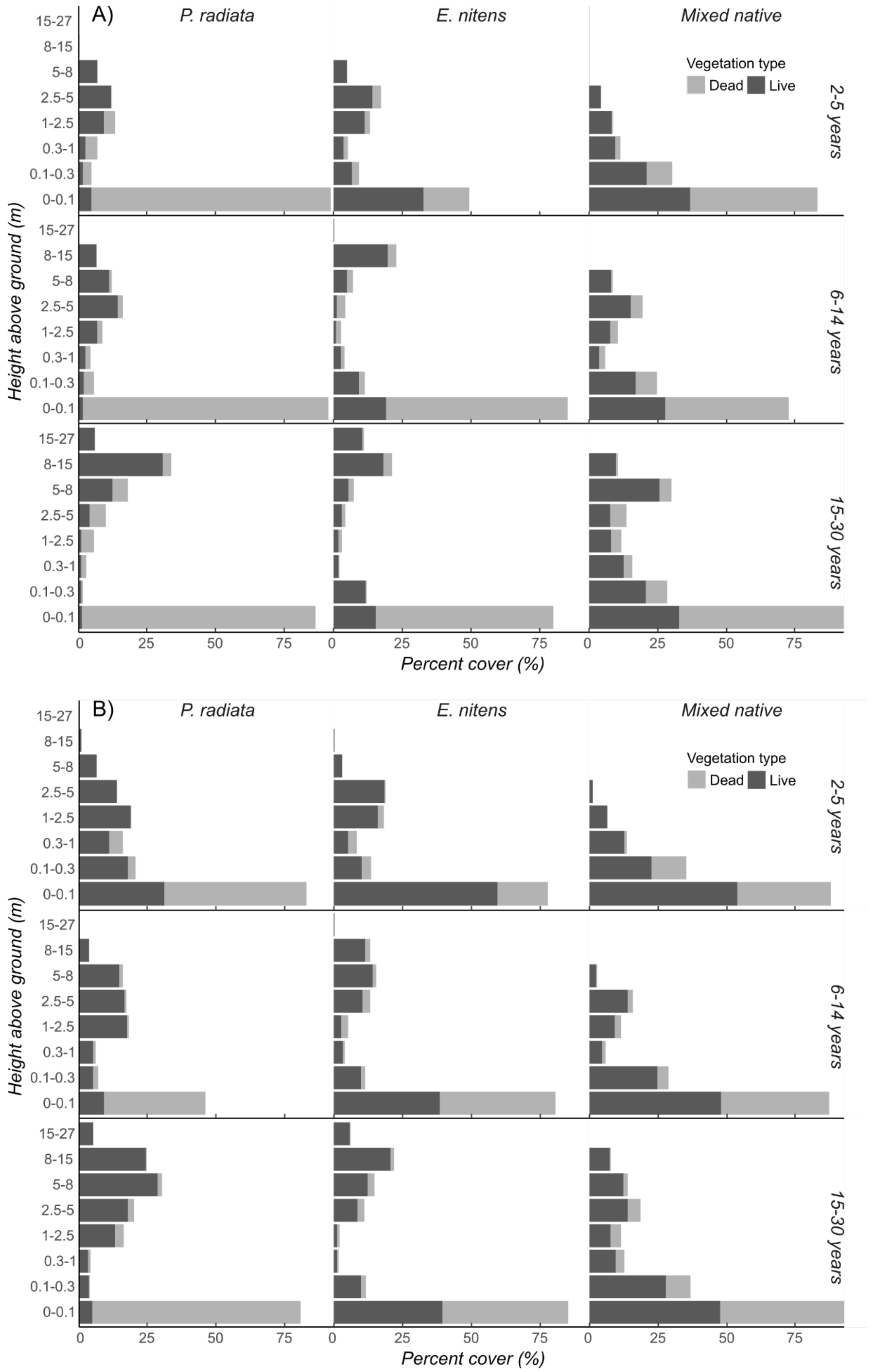


Figure 3.4: Mean percent cover of vegetation at different heights in middle (a) and edge (b) vegetation assessment plots for each shelterbelt composition type and age. Proportions of live and dead vegetation are differentiated by shading (see legend). Litter is represented as the proportion of dead vegetation in the 0-0.1 m stratum.

Across all ages, eucalypt shelterbelts had the largest amount of CWD, followed by the mixed native and pine composition types (Table 3.1). For all composition types, the amount of CWD present increased over time.

Total vegetation cover in the 0-0.1 m stratum (including litter, represented as 'dead' vegetation) was high across all composition types. Across all age categories, mixed native shelterbelts had the highest amounts of both total and live vegetation in the 0.1-1 m strata, followed by eucalypt shelterbelts. Over time, the mixed native shelterbelts retained higher vegetation density in the lower strata (0.1-1 m) compared to both eucalypt and pine shelterbelts, potentially as a result of canopy closure in these compositions and the deliberate planting of under-storey species in mixed native shelterbelts. Whereas mid-storey (1-5 m) vegetation density in the mixed native shelterbelts appeared to persist or increase over time, the density in these strata decreased over time for both eucalypt and pine shelterbelts, particularly in the centre of the shelterbelts. The height and overall density of canopy vegetation was lowest for mixed native shelterbelts, while pine and eucalypt shelterbelts had comparable levels.

3.4.3 Mean dominant height

Field data showed that, across all ages, the mean dominant height of eucalypt shelterbelts was slightly higher than pine shelterbelts, while the mixed native shelterbelts were considerably shorter (Figure 3.5). Mean dominant height of eucalypt shelterbelts increased at the fastest rate, followed by pine and mixed native shelterbelts (Figure 3.5). ANOVA showed a significant effect of species composition on mean dominant height ($F(2,27) = 38.88$, $p < 0.001$). In the oldest age category (15-30 years) the mean dominant height of eucalypt shelterbelts was highest (17.2 m), followed by pine (13.2 m) and mixed native (8.7 m) (Table 3.1).

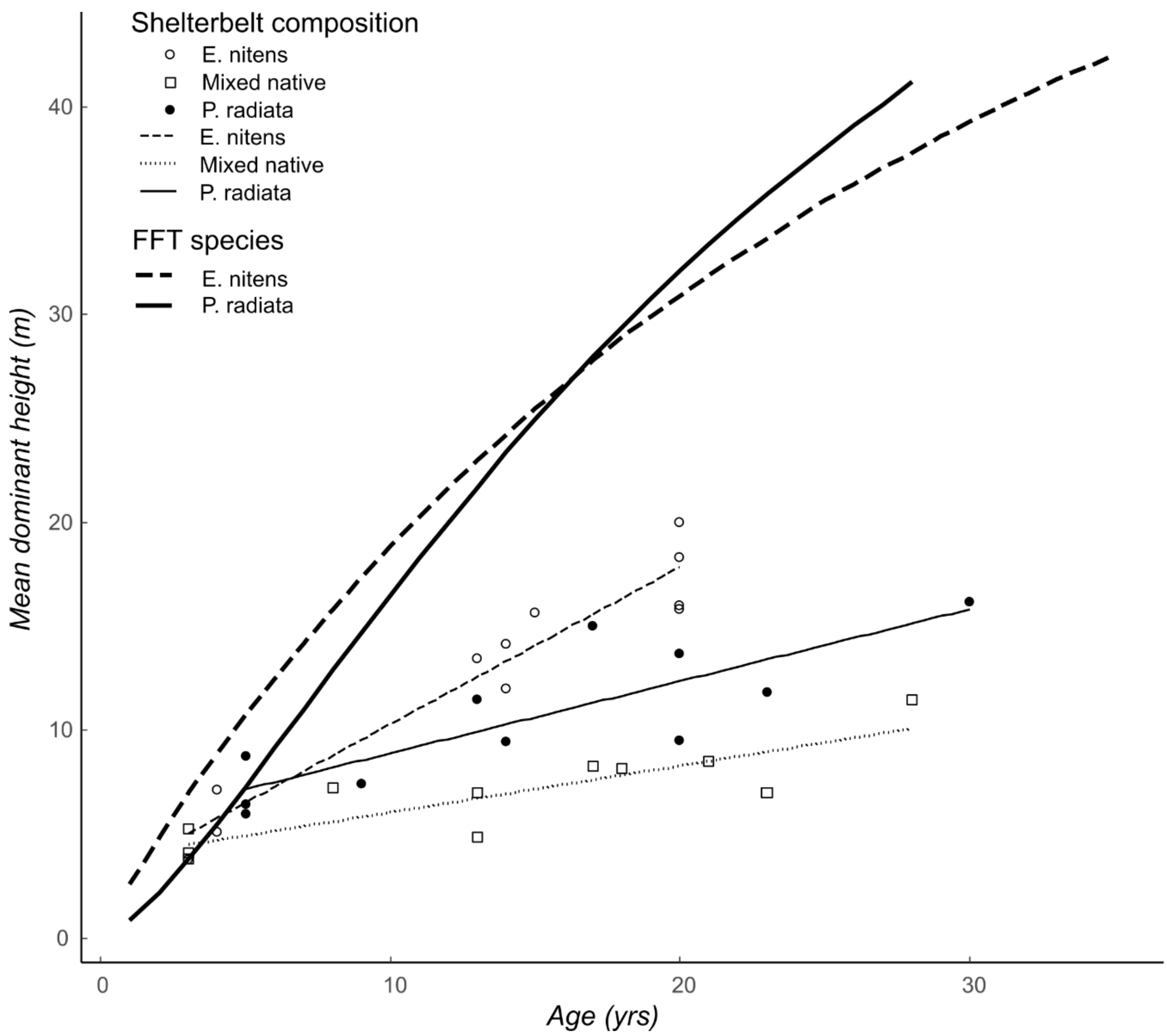


Figure 3.5: Relationships between mean dominant height of shelterbelts and age for three composition types. Shelterbelt field data (thin lines show linear regressions) are presented alongside Farm Forestry Toolbox (FFT) projections for mean dominant height of plantations of *P. radiata* and *E. nitens*.

Farm Forestry Toolbox projections suggest that mean dominant height of pine and eucalypt shelterbelts should be similar but that the rate of height increase in eucalypt shelterbelts should slow earlier than pine shelterbelts (Figure 3.5). Field-measured mean dominant heights of both eucalypt and pine shelterbelts were substantially lower than the projections modelled using the Farm Forestry Toolbox, particularly in the 15-30 year age category. This may be due to conditions in the Midlands being considerably drier than the regions to which the Farm Forestry Toolbox was calibrated. Differences between field-measured data and modelled projections may also be attributed to differences in the growth habits of shelterbelts compared to standard plantation configurations. Faster initial rates of height increase were measured for eucalypt shelterbelts compared to pine shelterbelts, which aligns with the modelled projections. Due to the lack of spread in the ages of shelterbelts tested, the field data do not capture potential changes in the rate of height increase over time. Considering the slowing rates of increase depicted in the modelled projections, it could be expected that the difference in height between pine and eucalypt shelterbelts would decrease in these later stages of growth.

3.4.4 Stand basal area

Field data on stand basal area for both pine and eucalypt composition types showed similar trends to the Farm Forestry Toolbox modelled projections. Pine and eucalypt shelterbelts had a similar stand basal area, although it was slightly higher in pine shelterbelts (Figure 3.6). The stand basal area of mixed native shelterbelts was lower across all age categories and increased at a slightly slower rate than eucalypt and pine shelterbelts. ANOVA showed a significant effect of species composition on stand basal area ($F(2,27) = 6.66$, $p = 0.005$). In the oldest age category (15-30 years) the stand basal area of pine, eucalypt, and mixed native shelterbelts were 57.8 m^2 ha^{-1}, 46.4 m^2 ha^{-1}, and 35 m^2 ha^{-1} respectively (Table 3.1). The high variation and large outliers in the field data may be due to the higher proportion of edge trees present in shelterbelts compared to plantations.

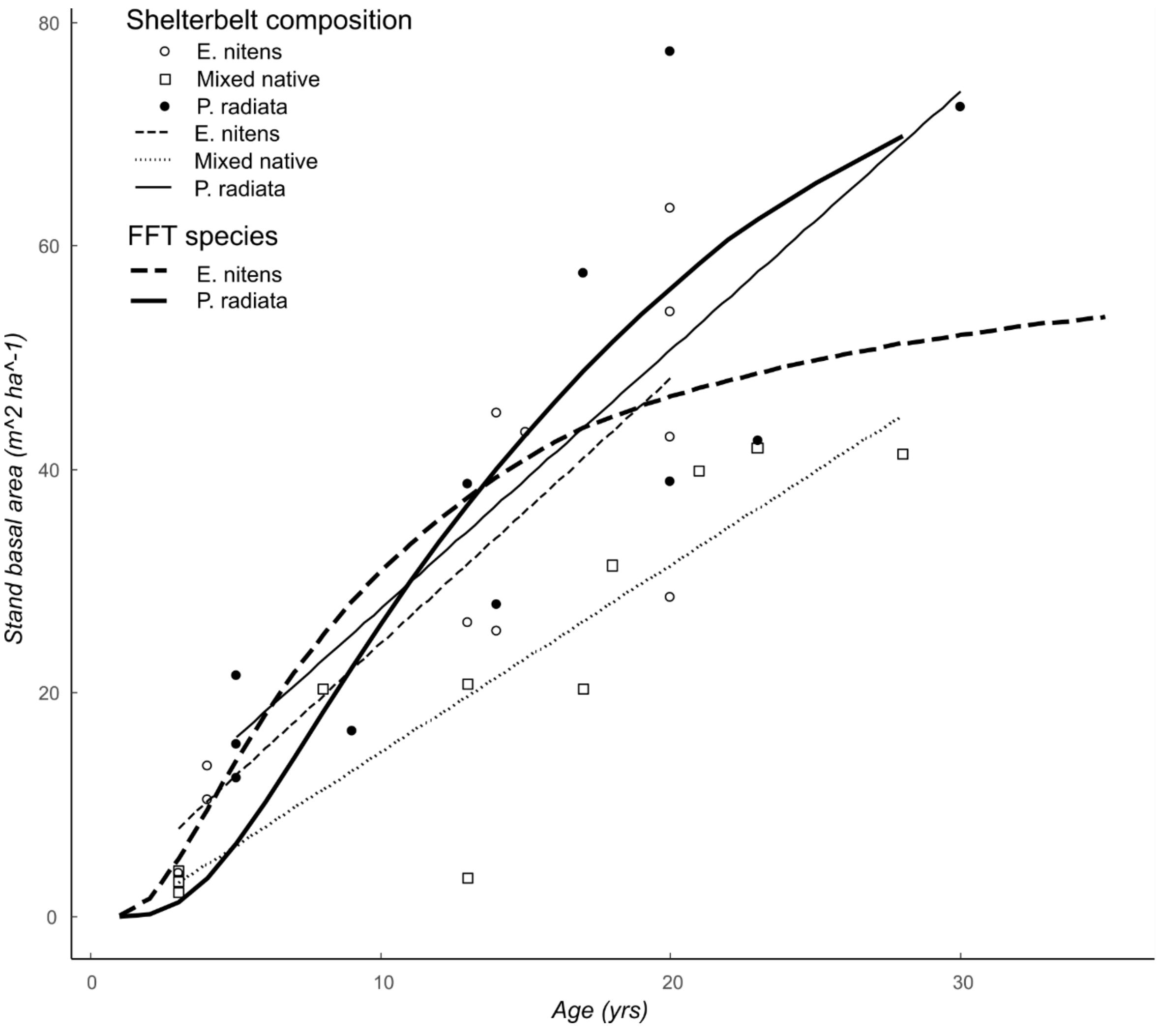


Figure 3.6: Relationships between stand basal area of shelterbelts and age for three composition types. Shelterbelt field data (thin lines show linear regressions) are presented alongside Farm Forestry Toolbox (FFT) projections for stand basal area of plantations of *P. radiata* and *E. nitens*.

Field-measured stand basal areas of both eucalypt and pine shelterbelts were similar to Farm Forestry Toolbox projections (Figure 3.6), although the projections do show a slowing rate of stand basal area increase over time, particularly for *E. nitens*, which may be lacking in the field data due to sampling strength. The slower rates of increase depicted in the modelled projections for *E. nitens* after about 15 years increases the difference in stand basal area between *P. radiata* and *E. nitens* compared to the shelterbelt field data.

Wood volume was also modelled using the Farm Forestry Toolbox, with *P. radiata* predicted to achieve a stand volume of 889 m^3 ha^{-1} at the end of a 25 year rotation, compared to 610 m^3 ha^{-1} for *E. nitens*.

3.4.5 Uniformity

Shelterbelts measured in this study were generally uniform: 29 of the 33 shelterbelts had less than 5% clear gaps as a proportion of their total length and 11 had no gaps at all. Of all composition types, pine shelterbelts had the most consistently high levels of uniformity across all three age categories (Table 3.1) and the mixed native shelterbelts were the least uniform, particularly in the two youngest age categories. Of the four shelterbelts with more than 5% clear gaps, one was a 15-30 year old eucalypt shelterbelt that had experienced severe windthrow, another was a 6-14 year old mixed native shelterbelt which had suffered high levels of insect attack, and two (one eucalypt and one mixed native shelterbelt, both in the 2-5 year category) had high rates of establishment failure.

3.4.6 Species diversity

ANOVA showed a significant effect of composition type on species diversity ($F(2,24) = 190.63$, $p < 0.001$). Across all age categories, mixed native shelterbelts had the highest species diversity, followed by eucalypt and pine shelterbelts (Table 3.1). The effect of shelterbelt age on species diversity was not significant ($F(2,24) = 0.3$, $p = 0.744$), although there was a statistically significant interaction between composition type and age ($F(4,24) = 3.023$, $p = 0.038$). There was little variation in mean species diversity across age categories, particularly for the eucalypt and pine composition types. In the case of mixed native shelterbelts, species diversity decreased in the 6-14 year category before increasing slightly in the 15-30 year category (Table 3.1).

3.4.7 Carbon

Field data showed that eucalypt shelterbelts accumulated the greatest amount of above-ground carbon over time, followed by pine and mixed native shelterbelts (Table 3.1, Figure 3.7). In the oldest age category (15-30 years) the above-ground carbon of pine, eucalypt, and mixed native shelterbelts were 132 tC ha^{-1}, 161 tC ha^{-1}, and 125 tC ha^{-1} respectively (Table 3.1). Eucalypt shelterbelts also accumulated above-ground carbon at a faster rate than both pine and mixed native shelterbelts. ANOVA showed a significant effect of species composition on above-ground carbon ($F(2,27) = 4.79$, $p = 0.017$).

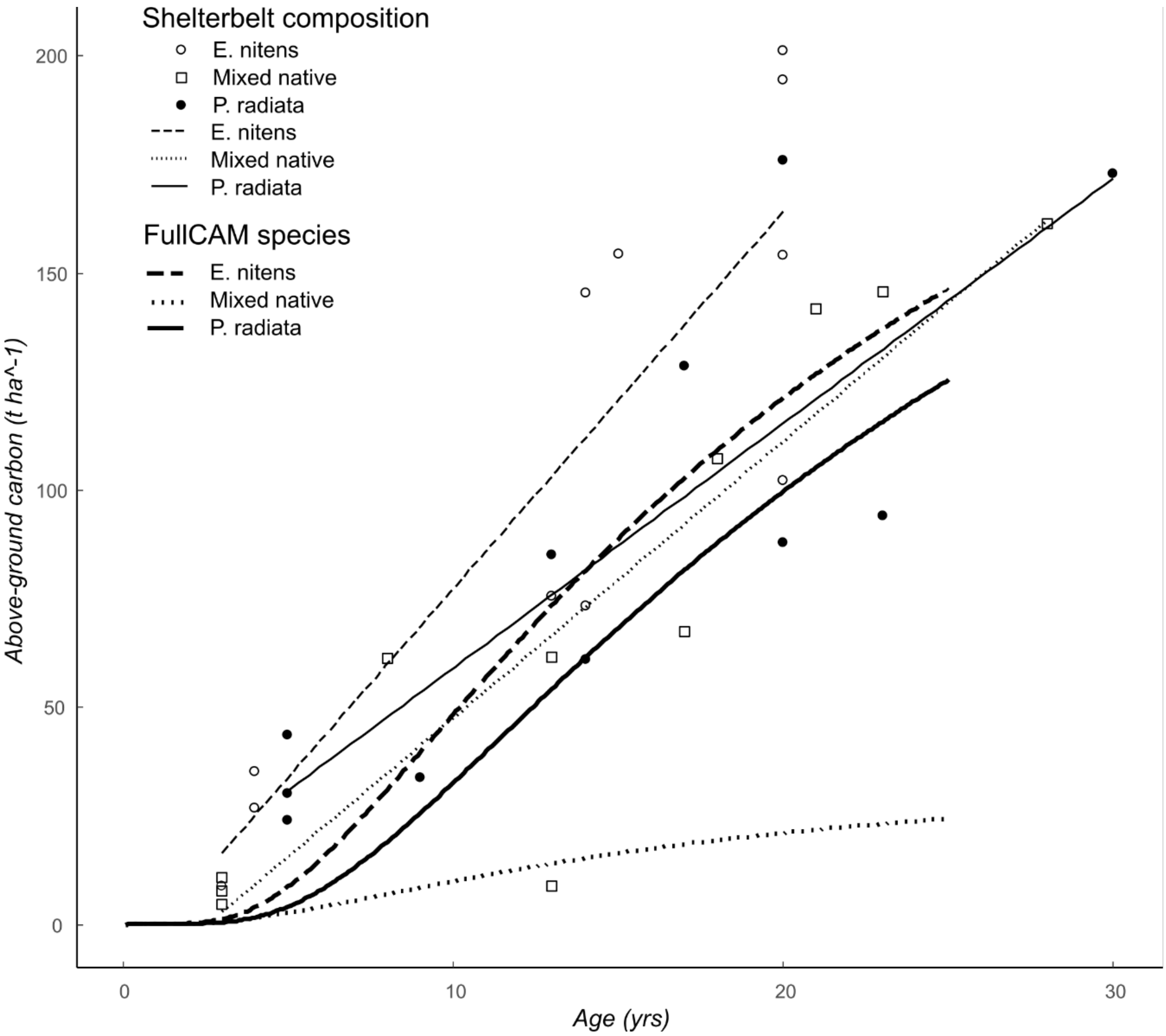


Figure 3.7: Relationships between above-ground carbon mass of shelterbelts and age for three composition types. Shelterbelt field data are presented (thin lines show linear regressions) alongside FullCAM (Version 6.19.07.114) projections for above-ground carbon mass for each shelterbelt composition type.

Field measurements of above-ground carbon for both eucalypt and pine shelterbelts were similar to FullCAM projections (Figure 3.7). However, field measurements of above-ground carbon for mixed native shelterbelts were substantially higher than the modelled projections. This may reflect the impact of species selection, as mixed native shelterbelts included in this study may contain a higher proportion of faster-growing tree species than that assumed by the FullCAM models.

3.5 Discussion

Analysis of shelterbelts in the Midlands region of Tasmania shows that structural characteristics including optical porosity, vegetation cover distribution, species diversity, height, stand basal area, and carbon sequestration are affected by species composition. Age also affects structure, although the impact varies depending on composition type. These findings suggest that species choice is a key factor in determining structural characteristics and therefore ecosystem services provided by agroforestry assets. While this finding is likely to be true globally, direct applicability of specific results is limited to comparisons of similar shelterbelt species compositions in regions with a similar climate (e.g. New Zealand's Canterbury Plains). Results will also vary depending on tree health, genotype, management, and a range of environmental factors (e.g. soil type, rainfall).

While the nature of the effects of species composition and age on shelterbelt structural characteristics (as outlined in the results of this study) may be intuitive, the scale of these effects and their implications for ecosystem service provision are more complex. Here we link our empirical results to existing literature on ecosystem services, discussing how structural differences affect provision of individual ecosystem services and how this information can be applied to compare the broad benefits of different types of shelterbelts.

3.5.1 Wind speed reduction

Wind speed reduction by shelterbelts is known to improve productivity by reducing; wind-associated crop damage (Cleugh, 1998), irrigation spray loss (Kilaka, 2015), and mortality of vulnerable livestock due to wind chill (Sudmeyer et al., 2007). Studies have also shown that microclimate changes associated with wind speed reduction (e.g. reduced evapotranspiration, reduced extremes in apparent temperature) can positively affect crop growth (Nuberg, 1998) and improve livestock productivity and welfare (He et al., 2017). Depending on the objective, practitioners may seek to maximise the extent of the sheltered area and/or the magnitude of wind speed reduction. A reasonable comparison of the effectiveness of different shelterbelt composition types can be made based on key structural characteristics: height, porosity, and uniformity (Bird et al., 2007).

Effects of height on wind speed reduction

Assuming that it is of sufficient length and width, the height of a shelterbelt will influence the extent of the area in which wind speed is reduced (Heisler & Dewalle, 1988). The sheltered area can extend out to a distance of 2-5 tree heights (TH) on the windward side and 10-30 TH on the leeward side (Brandle et al., 2004). Results showed that eucalypt and pine shelterbelts are both likely to reach greater mean dominant heights more quickly compared to mixed native shelterbelts. Although eucalypt shelterbelts reached the greatest mean dominant height in the oldest age category (17.2 m), this may not necessarily confer advantages as relationships between height and wind speed reduction are less predictable for shelterbelts taller than 15 m (Loeffler et al., 1992; Wu et al., 2018). However, initial eucalypt shelterbelt growth was faster than pine shelterbelts, suggesting that they may start delivering shelter benefits earlier.

Effects of porosity on wind speed reduction

Studies have shown that optical porosity provides a reasonable proxy for determining effectiveness of narrow shelterbelts (up to 20 m wide) (Loeffler et al., 1992; Středová et al., 2012; Wu et al., 2018). An optimal range of 20-40% optical porosity may exist for narrow shelterbelts (Wu et al., 2018) as turbulence created by very dense shelterbelts (< 10% optical porosity) reduces the extent of the sheltered area (Judd et al., 1996; Wang & Takle, 1996). Within this range, the magnitude of wind speed reduction generally increases with decreasing optical porosity (He et al., 2017; Heisler & Dewalle, 1988; Středa et al., 2008; Wu et al., 2018). Based on optical porosity alone, field data indicates that the pine (total: 14.5, reduced: 7.1) and mixed native (total: 19.1, reduced: 12.0) compositions are less porous and are predicted to provide a greater magnitude of wind speed reduction compared to the more porous eucalypt shelterbelts (total: 35.5, reduced: 34.4). Although the literature suggests that optical porosities of pine and mixed native shelterbelts are at the lower end of the optimal range, results for both compositions were mostly above 10% and any potential reduction in effectiveness as a result of turbulence is therefore anticipated to be minimal.

For shelterbelts wider than 15 m such as those included in this study, thresholds may not be directly transferable as optical porosity is likely to underestimate true ‘aerodynamic porosity’. While the general principles from the literature still provide a useful basis for comparing the relative effectiveness of each composition type, the gap in our understanding of porosity for wider shelterbelts demonstrates the need for additional research to confirm comparisons. It is also worth noting that differences in shelterbelt width were distributed evenly across the three composition types, and that potential underestimation of porosity will therefore be consistent across composition types.

Effects of vegetation distribution and uniformity on wind speed reduction

Vegetation cover distribution and uniformity also support findings related to porosity. Variation in the distribution of vegetation within a shelterbelt significantly affects wind speed reduction (Wu et al., 2013). Areas of high porosity or gaps at the base of a shelterbelt result in lower levels of wind speed reduction, as the speed of wind that is ‘funnelled’ through these gaps increases (Loeffler et al., 1992; Van Thuyet et al., 2014). An even vertical distribution of vegetation, with relatively low porosity, has been shown to achieve the best results in terms of both magnitude and extent of wind speed reduction (Cornelis & Gabriels, 2005; Wu et al., 2013; Wu et al., 2018). Results for vegetation distribution suggest that the mixed native composition type offers advantages over the other two composition types in the long term, due to retention of vegetation in the lower and middle strata (0.1-2.5 m) (Figure 3.4). Although all three composition types had relatively even vertical distribution of vegetation at younger ages, both pine and eucalypt shelterbelts developed high porosity in the lower (0-1 m) and middle (0.3-2.5 m) strata respectively, over time. The relatively uneven vertical distribution of vegetation in the pine and eucalypt shelterbelts could reduce their effectiveness in reducing wind speed.

Very little variation in uniformity was observed between the three compositions. Although our observations suggest that pine shelterbelts may be slightly less prone to gap-formation followed by eucalypt and mixed native shelterbelts, most shelterbelts were highly or entirely uniform and as such it is difficult for conclusions to be drawn around their relative effectiveness. It is worth considering that factors other than gap-formation (e.g. establishment methods) may have introduced variation in the uniformity data. For example, the high degree of variation in the uniformity of mixed native shelterbelts, particularly in the younger age categories, may be due to combined effects of species selection and less regular spacing of plants compared to the eucalypt and pine shelterbelts.

Summary of results relating to wind speed reduction

Height, optical porosity, and vegetation cover distribution suggest that eucalypt, pine, and mixed native shelterbelt composition types differ in their capacity to reduce wind speed and therefore deliver associated productivity benefits over time. Due to their greater height and the similarly even vegetation distribution across all composition types during early years of growth, eucalypt and pine shelterbelts are likely to deliver greater overall shelter benefits earlier than mixed native shelterbelts. Due to their lower porosity, pine shelterbelts may be more effective at reducing wind speeds compared to eucalypt shelterbelts during this time. After 15 years, mixed native shelterbelts appear to offer the greatest overall benefits due to their low porosity and even vertical distribution of vegetation, although there is a trade-off due to their lower height. Although pine shelterbelts have similarly low porosity and greater height, increases in the porosity of lower strata may lower the magnitude of wind speed reduction to the extent that this outweighs advantages associated with shelter distance. Despite their height, eucalypt shelterbelts offer the least overall benefits at this stage due to their high overall porosity and gaps in the lower-middle strata. The relative benefits of pine and mixed native shelterbelts will ultimately depend on the specific objectives of the practitioner in relation to wind speed reduction, and the timeframe in which they hope to achieve these objectives.

3.5.2 Wood production

Shelterbelts may be established with a view to harvest wood products for on-farm or commercial use. In such cases, the volume of merchantable wood produced would be a key consideration. Stand basal area and mean dominant height both correlate positively with volume and provide a reasonable basis for comparison (Deadman & Goulding, 1978). Results showed that eucalypt and pine shelterbelts achieved similar wood volumes, although pine shelterbelts had greater predicted volumes due to their larger basal area. Although mixed native shelterbelts are rarely planted for wood production, depending on the qualities of individual tree species some mixed native shelterbelts may still provide opportunities to produce wood for on-farm use, e.g. firewood.

Existing species-specific growth models such as those within the Farm Forestry Toolbox are useful tools for comparison, however it should be noted that field measurements of mean dominant height for both species were substantially lower than the modelled results. This may be due to the variable and potentially sub-optimal growing conditions provided by farm sites in the Tasmanian Midlands. This highlights that comparisons between composition types may differ depending on location and that use of existing tools therefore requires consideration of local conditions. Differences between field measurements and modelled results in this study may also be attributed to differences in levels of competition between trees in shelterbelt configurations compared to standard plantation configurations. There may be opportunities to use data from this study and others to broaden the geographic range of calibration and include different configuration options for growth models within the Farm Forestry Toolbox, thereby making them more useful to practitioners in a wider range of settings. Additionally, while modelled results suggest that pine shelterbelts produce greater volumes of merchantable wood compared to eucalypt shelterbelts, the value of the wood produced may also vary considerably between species depending on the timber quality, market, price, and transport distance to processor.

3.5.3 Habitat for fauna

In highly modified agricultural areas where remnant vegetation is sparse, shelterbelts play important roles as permanent habitat, refugia, or dispersal corridors for fauna (Fischer & Lindenmayer, 2002; Kay et al., 2020; Stamps & Linit, 1997). For some practitioners, a general desire to support biodiversity and its associated amenity in the landscape can be a primary driver for shelterbelt establishment. Others may seek to support specific faunal groups associated with ecosystem services that improve productivity e.g. pollinators (Garibaldi et al., 2013), pest-predators (Huang et al., 2018; Rahman & Norton, 2019), and nutrient-cyclers (e.g. dung beetles). It is also important to consider the potential cost of inadvertently providing habitat to ‘non-beneficial’ fauna, i.e. pests. Relationships between fauna and farm productivity are complex and likely to differ between regions and enterprise types.

Effects of vegetation distribution and diversity on fauna

Effects of vegetation structure on the abundance and diversity of fauna are highly specific to faunal groups (McElhinny et al., 2006). However, the most broadly-applicable structural trend is that greater complexity and diversity in vegetation increases the variety of habitat and food resources, which in turn results in higher levels of faunal diversity (Carr et al., 2000; McElhinny et al., 2006). Characteristics such as plant species diversity, floral diversity, fallen timber, leaf litter, structural heterogeneity, maintenance of groundcover, and vegetation density can positively affect abundance and/or diversity of beneficial invertebrates (Aviron et al., 2005; Ng et al., 2018a; Ng et al., 2018b; Saunders & Luck, 2018; Stamps & Linit, 1997) and birds (Bain et al., 2020; Hastings & Beattie, 2006; Montague-Drake et al., 2009). Native ground mammals vary considerably in their preferences for vegetation structure, and are therefore best-supported by small-scale mosaics of dense and open vegetation (McElhinny et al., 2006).

Out of the three composition types, the mixed native composition had the highest levels of plant species diversity across all strata and age categories, and higher levels of live under-storey and mid-storey cover (Table 3.1, Figure 3.4). This suggests that the mixed native composition will provide higher quality habitat for fauna compared to eucalypt or pine over the total lifespan of the shelterbelts, although differences are not as pronounced during early years of growth. The eucalypt composition type had slightly higher levels of live groundcover and species diversity than pine, and the highest amount of CWD out of the three composition types. Although the absence of under-storey and mid-storey vegetation limits their habitat value (Hastings & Beattie, 2006), pine and eucalypt shelterbelts may still provide important foraging or nesting sites for small mammals, birds, and invertebrates (Salt et al., 2004). There is not enough evidence relating to herbivore pests to suggest that any one composition type has disadvantages over the other in this regard. To broaden the application of these findings, there may be value in adopting a traits-based approach for future work to identify key characteristics of tree/shrub species that drive differences in structure e.g. flowering times of species in mixed native belts.

3.5.4 Carbon sequestration

Shelterbelts can assist in mitigating climate change by sequestering carbon and thereby reducing atmospheric concentrations of greenhouse gases (Schoeneberger et al., 2012; Verchot et al., 2007). If practitioners plan to access payments for carbon sequestration through relevant carbon trading schemes, they may be interested in the relative capacity of different composition types to sequester carbon over time. Field measurements showed that eucalypt shelterbelts accumulated the most above-ground carbon mass, followed by pine and mixed native shelterbelts. High carbon accumulation levels in eucalypt and pine shelterbelts are associated with larger stem volumes, as stem volume is highly correlated with above-ground carbon mass (Paul et al., 2006). Stem density may also account for the slightly higher carbon accumulation observed for eucalypt shelterbelts compared to pine shelterbelts, as *E. nitens* has approximately 20% higher basic density than *P. radiata* (Ilic et al., 2000). Differences in potential revenue through carbon trading schemes between composition types will also depend on the scale of the operation, the introduction of management regimes (e.g. silviculture, harvesting), as well as the state of the carbon market and policy environment at the time.

Field measurements showed higher above-ground carbon in mixed native shelterbelts than predicted by the FullCAM models. It is important to consider that due to variation in actual composition of mixed native shelterbelts, the FullCAM models may underestimate the proportion of canopy species and therefore above-ground carbon. This highlights the importance of on-ground structural measurements for accurate quantification of ecosystem services at fine scales, particularly for more complex shelterbelt composition types.

3.5.5 Other services

In this study, comparison of relative benefits was limited to ecosystem services on which above-ground structure has considerable influence (i.e. wind speed reduction, wood production, habitat provision, and carbon sequestration). However, shelterbelts also: improve on-farm water and air quality, regulate soil erosion, provide visual screening or noise attenuation, enhance soil formation and nutrient dynamics, mitigate salinity impacts, and improve the aesthetic appeal of a property (Jose, 2009; Smith et al., 2012) (see Figure 2.3). Although there may be differences in provision of these services between species compositions, quantifying these differences was beyond the scope of this study. In the case of some services (e.g. aesthetics), insufficient evidence linking above-ground structural characteristics to potential benefits restricts potential comparisons.

3.5.6 Combined comparison of farm-scale benefits

In Sections 3.5.1-3.5.4 we discussed how structural differences between different shelterbelt species composition types and ages affects provision of key ecosystem services and their associated benefits. Although comparison based on a single ecosystem service may be of interest in some cases, practitioners will often seek to understand synergies and trade-offs between a range of desired services. The particular range of services of interest will depend on enterprise type and the priorities of the practitioner. Using the collected information, we are able to compare each composition type and age category based on the combined farm-scale benefits associated with the following key services: wind speed reduction, wood production, habitat provision, and carbon sequestration (Table 3.2). Depending on the priorities of the landholder and availability of evidence, the range of services in this exercise could potentially be expanded to include others from Figure 2.3. A simplified system of scoring is used with 'points' (+) allocated for low (+), medium (+ +), and high (+ + +) levels of relative benefit. These points are based on comparisons of structural characteristics presented in Table 3.1 and Sections 3.4.1-3.4.7, considered in the context of relevant literature linking these characteristics to farm-scale benefits (Sections 3.5.1-3.5.4). Specific structural characteristics considered were: stand basal area and height for 'wood production'; height, porosity, and vegetation cover distribution for 'shelter'; above-ground carbon for 'carbon sequestration', and vegetation cover distribution, species diversity, and coarse woody debris cover for 'habitat for fauna'.

Table 3.2: Relative benefits of shelterbelt composition types and age categories based on provision of ecosystem services. Points (+) are allocated for low (+), medium (+ +), and high (+ + +) levels of relative benefit.

Composition type	Age (years)	Wood production	Shelter	Carbon sequestration	Habitat for fauna
P. radiata	2-5	+	+ + +	+	+
	6-14	+ +	+ + +	+ +	+
	15-30	+ + +	+ +	+ +	+
E. nitens	2-5	+	+ +	+	+
	6-14	+ + +	+ +	+ + +	+
	15-30	+ +	+	+ + +	+
Mixed native	2-5	+	+	+	+ +
	6-14	+	+ +	+	+ + +
	15-30	+	+ + +	+ +	+ + +

Comparison of multiple benefits for each composition type and age category suggests that pine, eucalypt, and mixed native shelterbelts offer similar levels of overall benefit, although pine shelterbelts deliver slightly greater benefits over time and begin delivering benefits sooner than the other two compositions (Table 3.2). Mixed native shelterbelts deliver fewer benefits in early stages of growth but deliver the greatest level of benefit in the 15-30 year age category. Although this system of scoring is simplistic, it demonstrates the relative potential of each composition type to deliver a range of benefits over time, based on structural differences. The scoring system also illustrates trade-offs between different ecosystem services, i.e. how consideration of different combinations of services can affect the overall benefit of any one composition type. For example, in this case if provision of habitat for fauna is not a key consideration for the practitioner, pine becomes the optimal composition type in all age categories.

Although all services have been given equal weighting in this system, some ecosystem services may generate proportionally greater levels of benefit than others depending on the type of enterprise. For example, if multiple benefits are expected to be generated through wind speed reduction (e.g. fewer livestock losses and improved pasture growth), scores for 'shelter' may need to be weighted accordingly. A more thorough comparison could be made if valuation tools (e.g. production functions) and market prices are incorporated to estimate specific monetary values for each benefit. It also needs to be recognised that the suitability of a given species for growing on any particular site needs to be taken into consideration, as the benefits will not accrue to the same extent if the growth and/or survival is unsatisfactory.

3.5.7 Application in natural capital accounting

There is increasing interest in the use of natural capital assessment and accounting to demonstrate potential farm-scale benefits associated with agroforestry. Previous studies have identified that meaningful application of natural capital accounting in this context is limited by our inability to quantify the impact of agroforestry asset condition on provision of ecosystem services at fine scales (Marais et al., 2019). This study quantifies the extent to which species composition and age affect structure, an important component of condition, and how this in turn affects provision of several ecosystem services. Used in conjunction with conceptual models (e.g. Figure 2.3) and ecosystem service valuation tools, this information could improve the accuracy of farm-scale natural capital assessments and accounts for systems that include shelterbelts. Methods used in this study could also be adapted to suit other types of agroforestry assets in other systems or regions. If this approach is sufficiently expanded to include a broad range of asset types, characteristics, and ecosystem services, it could be used to inform the development of comprehensive decision-making tools for agroforestry practitioners.

Results from this study may also have relevance beyond the farm scale. Ecosystem services provided by shelterbelts on farms can contribute to the delivery of public benefits such as climate change mitigation, improvement of amenity for visitors or peri-urban dwellers, and biodiversity conservation. Policies to support establishment of shelterbelts (e.g. Canada's 'Prairie Shelterbelt Program' and China's 'Three-North Shelterbelt Program') are generally developed with the aim of enhancing provision of these broader benefits (Lyu et al., 2020; Mayrinck et al., 2019). Findings from this study suggest that the fine scale condition of a shelterbelt influences its capacity to provide ecosystem services, including those which contribute to provision of public benefits. This has important implications for policy design, as the influence of factors such as species selection, when amplified across broad scales, may significantly affect policy outcomes. Structural characterisation of shelterbelts and other agroforestry assets could inform predictions of service provision at multiple scales, thereby improving outcomes of policies aimed at enhancing public benefits.

3.6 Conclusions

In this comparison of three shelterbelt species compositions (pine, eucalypt, and mixed native), composition was shown to affect structural characteristics including optical porosity, vegetation cover distribution, species diversity, height, stand basal area, and carbon sequestration. These structural differences affect the capacity of each composition type to provide ecosystem services and benefits at the farm scale, although in some cases the evidence base for these effects is lacking. Effects of shelterbelt age on structure varied between composition types, demonstrating that shelterbelt composition also affects timeframes for delivery of benefits.

Results were consolidated to 'score' each shelterbelt composition type based on their relative capacity to deliver a combination of benefits over time. This provides a novel, albeit simplistic, approach for quantifying the impacts of agroforestry asset 'condition' on provision of multiple ecosystem services at the farm scale. With some extension and adaptation, this approach could facilitate more meaningful application of natural capital accounting in this context and inform development of decision-making tools for practitioners.

Chapter 4: Effects of tree species selection on local invertebrate community composition in shelterbelt agroforestry systems

This chapter is in preparation as:

Marais, Z.E., Baker, T.P., Hunt, M.A., and Barmuta, L.A. (in preparation). Effects of tree species selection on local invertebrate community composition in shelterbelt agroforestry systems.

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4.1 Abstract

Continued intensification of agriculture threatens invertebrate communities and the critical ecosystem services that they provide. Agroforestry assets such as shelterbelts support invertebrate communities in agricultural landscapes through provision of refuge, food resources, breeding/nesting sites, and microclimate regulation. To design agroforestry systems that effectively support invertebrate communities and enhance provision of invertebrate-related ecosystem services (e.g. biodiversity, pollination), gaps must be addressed in our understanding of how the fine-scale condition of agroforestry assets (determined by design choices such as tree species selection) affects invertebrate communities. This study compared invertebrate communities (community composition, total invertebrate abundance, and abundance by order/higher taxon and functional sub-group) in three common shelterbelt types (*Pinus radiata*, *Eucalyptus* sp., and mixed native) and in open pastures in the Midlands region of Tasmania, Australia over spring (2018), summer (2019), and autumn (2019). Existing knowledge of structural characteristics of vegetation within the three shelterbelt types was used to infer potential relationships between shelterbelt vegetation structure and invertebrate community composition. Invertebrate communities within shelterbelts were different to those in open pasture in summer and autumn, with shelterbelts having higher abundance of slaters (order: Isopoda) and beetles (order: Coleoptera), and lower abundance of bees (superfamily: Apoidea) and grasshoppers/crickets (order: Orthoptera). Tree species selection affected overall invertebrate community composition within shelterbelts in autumn, as well as local abundance of particular invertebrate taxa including beetles (order: Coleoptera), ants (family: Formicidae), and booklice (order: Psocoptera). Functional sub-groups (pollinators, predators, and pests) within these taxa tended to exhibit preferences for mixed native and eucalypt shelterbelts over pines. These findings suggest that although shelterbelts offer some invertebrate conservation benefits regardless of tree species, supporting specific invertebrate taxa and functional groups requires consideration of impacts of tree species selection on habitat suitability.

4.2 Introduction

Design and management of multifunctional agricultural landscapes are increasingly recognised as critical components of global efforts to conserve biodiversity. Agroforestry systems are proposed as a win-win for production and the environment, as they can enhance provision of multiple ecosystem services which deliver public and private benefits at various scales. Through provision of habitat, resources, and shelter, agroforestry systems may alter the composition of local invertebrate communities, in turn supporting biodiversity and influencing provision of invertebrate-related ecosystem services such as pollination and pest management (Stamps & Linit, 1997). Amid growing concern about the impacts of agriculture on invertebrates and flow-on effects for productivity (Raven & Wagner, 2021), there is value in understanding how different forms of agroforestry compare in terms of their influence on local invertebrate communities. Studies have shown that some forms of agroforestry can be effective in supporting beneficial invertebrate taxa at both the landscape (Graham & Nassauer, 2019) and farm scale (Varah et al., 2013). However, few studies account for the potential effects of differences in the fine-scale condition (e.g. configuration, species choice) of agroforestry assets. To improve the accuracy of ecosystem service modelling and inform farm-scale decision-making, there remains a need to explore how the condition of agroforestry assets affects invertebrate communities at fine scales.

Invertebrates constitute a significant component of terrestrial biodiversity and provide multiple ecosystem services and disservices in agricultural contexts. While some species can cause significant damage to crops when present in high numbers, invertebrate predators and parasitoids are thought to be primarily responsible for control of pests in 33% of cultivated agricultural systems (Hawkins et al., 1999). Supporting pest-predators on farms can provide both economic (reducing loss of yield for little or no cost) (Losey & Vaughan, 2006) and environmental benefits (reducing requirements for environmentally-harmful pesticides) (Chagnon et al., 2015). Wild pollinators are critical to maintaining biodiversity and provide valuable pollination services to agriculture, with about 70% of the world's most important food crops depending at least partly on animal pollination (Klein et al., 2007; Ollerton et al., 2011). Other invertebrates may contribute to ecosystem services such as seed dispersal (e.g. ants), and nutrient cycling through decomposition of organic matter (e.g. dung beetles). Pressures including landscape alteration (conversion or degradation), agricultural intensification, and climate change have been linked to decline of invertebrate groups such as pollinators, generating concern around continued provision of essential ecosystem services (González-Varo et al., 2013). Agricultural practices that are known to support invertebrates, including agroforestry (e.g. Varah et al., 2013), play an important role in alleviating these pressures.

Shelterbelts are a common type of agroforestry, particularly in areas where wind carries significant productivity risks. While often planted primarily for wind speed reduction, shelterbelts can be designed to provide multiple co-benefits including timber production, carbon sequestration, and biodiversity (Chapter 2, Baker et al., 2021b; England et al., 2020). Shelterbelt systems may vary in terms of species selection, configuration (their position in relation to other forms of agriculture on the farm), and extent (area of shelterbelts as a proportion of total property area). Species selection is a key consideration for farmers, particularly if they intend to harvest commercial wood products from shelterbelts. Previous studies show that species selection affects structural characteristics of shelterbelts that determine provision of ecosystem services at fine scales (Chapter 3) (see Figure 3.1). However, while links between structure and services such as carbon sequestration and wind reduction are understood, evidence linking variation in structural characteristics to invertebrate community composition is limited.

Predictions of invertebrate-related ecosystem services in agroecosystems are generally based on the extent and configuration of different vegetation types within the area of interest, as well as any available information on the habitat requirements and dispersal abilities of target taxa (e.g. Olsson et al., 2015). To date, research on habitat requirements has focused heavily on pollinating insects, particularly bees (Kennedy et al., 2013), and invertebrates which are parasitic or predate upon pests (Gurr et al., 2017). In the case of native and exotic bees, habitat suitability is thought to depend on availability of nesting substrates and floral resources (Lonsdorf et al., 2009). For predatory and parasitic invertebrates, studies suggest that mid-storey and canopy vegetation is less important than the physical structure of the ground and its associated flora (Thomas et al., 1991). While structural characteristics could be used to infer the suitability of different types of shelterbelts as habitat for pollinators or predators, effects of vegetation structure on other functional groups of invertebrates are not as well established. Further, assumptions relating to habitat suitability have yet to be widely validated in shelterbelt systems within landscapes of low complexity.

This study explores the effects of shelterbelt species selection on composition of local invertebrate communities, using ground-active and low-flying invertebrate samples from three shelterbelt types that are common in temperate Australia (*Eucalyptus* sp. and *Pinus radiata* monocultures, and mixed native species). This study was based in the intensively-farmed Midlands region of Tasmania, Australia. We aimed to answer the following questions:

1. Do local invertebrate communities differ significantly: a. between shelterbelts and open pasture, and b. between shelterbelt species types, over different seasons (spring, summer, and autumn)?

2. How is the abundance of specific invertebrate orders and functional sub-groups (e.g. predators, pollinators) affected by shelterbelt species type over different seasons?

We draw on known information about structural characteristics of vegetation in each shelterbelt species type (e.g. litter cover, under-storey cover, coarse woody debris cover etc.) (Chapter 3) to infer potential relationships between these characteristics and local invertebrate community composition.

4.3 Methods

4.3.1 Study sites

Nine study sites were selected in the Midlands region of Tasmania, Australia (Figure 4.1), a highly-modified mixed agricultural landscape. Study sites included three shelterbelt species types (three sites per species type) that are common to temperate Australia (*Pinus radiata*, *Eucalyptus* sp., and mixed native). Although some sites were part of mixed farming systems, all sites consisted of a shelterbelt adjacent to managed pasture at the time of this study. All study sites were located within an area between 41°31'S-41°43'S and 146°50'E-147°10'E spanning elevations of 142-202 m a.s.l. This area experiences an annual temperature range of -4 to 32°C and receives an average annual rainfall of 400–700 mm.

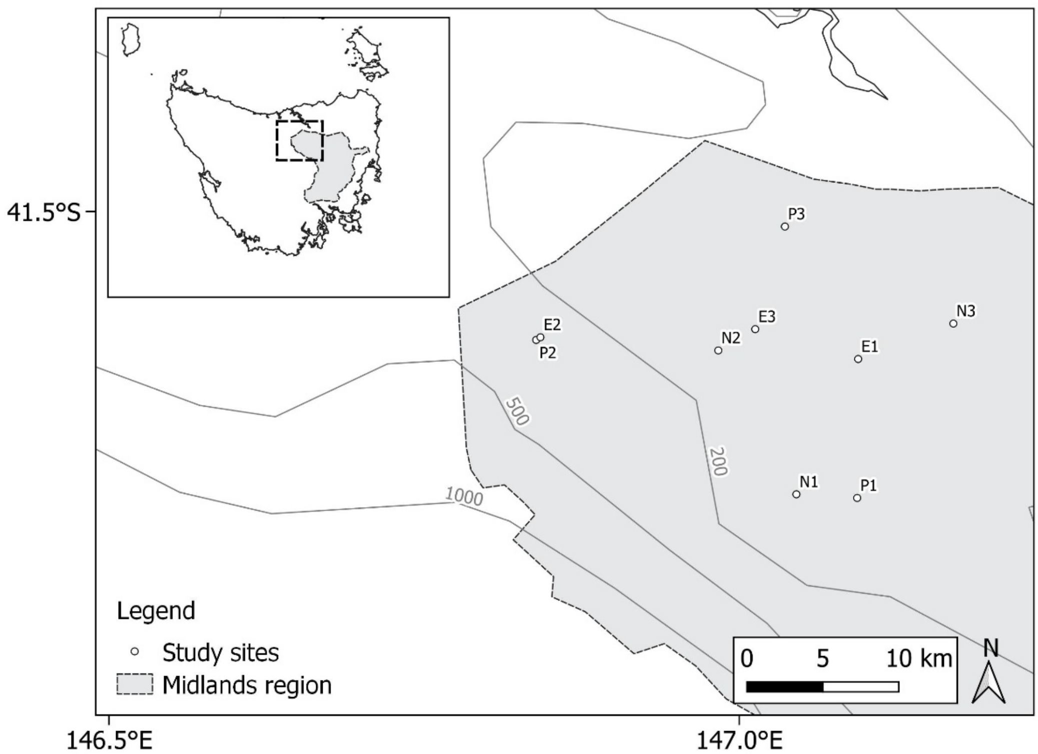

Figure 4.1: Location of study sites within the Midlands agricultural region of Tasmania (grey). 'E', 'P', and 'N' in study site labels denote *Eucalyptus* sp., *Pinus radiata*, and mixed native shelterbelt types, respectively. Topographic lines show height above sea level in metres.

With the exception of one *Eucalyptus* sp. site which consisted of predominantly *E. ovata* with a small amount of *E. pauciflora, Eucalyptus* sp. and *P. radiata* sites were planted as single-species shelterbelts (remaining *Eucalyptus* sp. sites were *Eucalyptus nitens*), typically for wood production. Mixed native shelterbelts, typically planted for biodiversity or carbon rather than wood production, varied in their composition but were all dominated by *Eucalyptus* sp. and *Acacia* sp. and contained a total of at least four Australian native shrub/tree species. Study sites were chosen that had a single shelterbelt adjacent to pasture, with most shelterbelts oriented approximately perpendicular to the prevailing wind direction. As winds in the study area originate mostly from the west and north, shelterbelts included in this study generally had a south-eastern aspect on the leeward side. All sites were planted 15-30 years prior to this study being undertaken. Exact ages of each site were recorded if known, otherwise ages were estimated using available historical aerial imagery. All shelterbelts were at least 200 m in length and 15-30 m wide, measured to the edge of the crown. All shelterbelts consisted of multiple rows of trees (ranging from 4 to 7 rows). Stock were excluded from all shelterbelts by fencing.

All study sites were considered to have relatively low landscape complexity (<15 % cover of semi-natural forest or grassland within a 1000 m radius). Classifications were based on aerial imagery and publicly-available vegetation mapping (TASVEG 3.0), and ground-truthed by visual inspection of study sites. Local landscape composition was similar at each of the nine sites, with agriculture (crops or pasture) comprising >90 % of the area within a 1000 m radius of each site, remnant native forest <7 %, plantation forest <6 %, and native grassland <6 %. Despite efforts to find sites with similar broader landscape composition (within a 3000 m radius), variability of composition at this resolution was greater due to limited availability of suitable sites. All but two of the sites had <12 % remnant native forest within a 3000 m radius and <5 % plantation forest. The remaining two sites had 21-23 % remnant native forest within a 3000 m radius and 14 % plantation forest. All sites were located in areas where there was <5 % native grassland within a 3000 m radius.

To limit the influence of other landscape features, sites were selected so that all invertebrate traps were at least 50 m from other integrated non-pasture vegetation (e.g. paddock trees, hedges), at least 100 m from remnant native vegetation patches or plantations, at least 200 m from flowering crops (although some flowers were present in the studied pasture), and at least 100 m from water bodies (e.g. dams, creeks). Stock were excluded from the pasture during sampling, with the exception of two sites in the spring sampling period. At these sites, traps within stocked areas were enclosed within a 1 x 1 m area using wire fencing with 12.5 x 15 cm mesh size. For the duration of the study, soils in adjacent pastures were not disturbed and pesticides were not applied.

4.3.2 Invertebrate sampling

Insects were sampled within the centre of the shelterbelts, at three separate locations along the length of each shelterbelt (three sample points per study site). These locations were randomly selected, although it was ensured that they were at least 50 m apart from each other and at least 50 m from the ends of each shelterbelt (Figure 4.2). Sample points within the shelterbelts were located at equal distances from the windward and leeward canopy edges of the shelterbelt. 'Control' samples representing the open pasture were also collected at three of the sites, with three sample locations 50 m apart established per site (total of nine control samples). Controls were located approximately 250 m from each shelterbelt (on the leeward side of the dominant wind direction) although minor adjustments were made to maintain minimum buffer distances from other vegetation and water bodies (refer Section 4.3.1). As some invertebrate taxa are known to disperse over distances greater than 250 m (see e.g. Schmidlin et al. (2021)), samples at these distances do not represent true 'controls' for all taxa. However, as it was not possible to access appropriate sites within the study area that could accommodate sampling at greater distances from vegetation or water bodies, these controls offered the best available representation of invertebrate communities in open pasture.

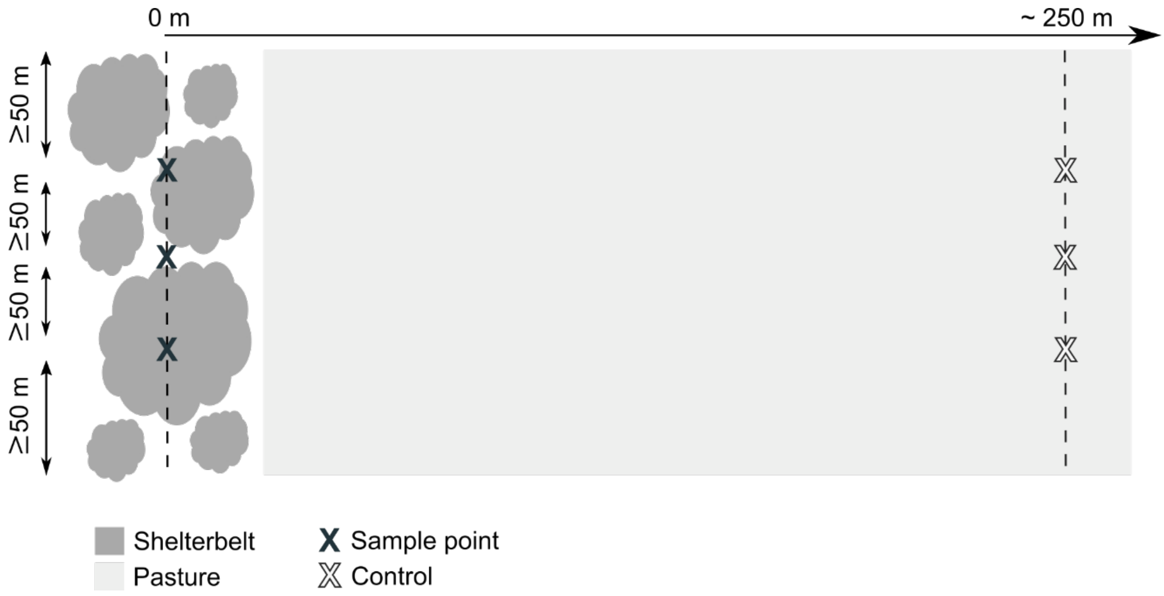

Figure 4.2: Study site configuration showing the location of sample points in relation to the shelterbelt.

At each sample point, two different types of trap were installed (Figure 4.3). Ground-active invertebrates were sampled using a single pitfall trap which consisted of a 350 mL plastic cup, placed in a hole so that the lip of the cup was flush with the soil surface. The cups were filled with 100 mL of preservative (1:2 propylene glycol-water) with a small amount of detergent added to reduce surface tension. To protect pitfall traps from rain and disturbance by animals, a 17 cm diameter plastic plate was suspended approximately 5 cm above each trap using three wooden skewers. Low-flying invertebrates were sampled using a single window flight intercept trap, placed 30 cm away from the pitfall trap. Each trap consisted of a white 11 L plastic bucket with two intersecting 28 x 30 cm transparent Perspex panels secured over the rim of the bucket, resulting in samples being collected 30-60 cm above ground level. The buckets were filled with 500 mL of the same preservative solution used in the pitfall traps.

Figure 4.3: Photograph showing the window flight intercept (left) and pitfall traps (right) in the field.

Samples were collected over three consecutive seasons: spring (October 2018), summer (January 2019), and autumn (April 2019). Sampling across all four seasons was not possible as the broader study (Chapters 4 and 5) involved large numbers of samples per site, and capacity for processing samples was limited. Although shelterbelts may provide refuge for invertebrates during winter, winter was excluded to prioritise sampling of taxa in groups of particular functional interest (e.g. pollinators) which are generally more abundant and active in warmer months. In each season, traps were operated at all sites concurrently for seven days. During the summer sampling period, intercept traps were topped-up with additional preservative due to high rates of evaporation.

After collection, samples were poured through a 340 µm stainless steel mesh sieve and transferred to a sorting tray with 70% ethanol solution. Invertebrates were sorted to order and sub-order groups (Appendix B), with reference to CSIRO (1991) and the Tasmanian Institute of Agriculture Insect Collection. Sub-order groupings at the family, genus, or species level were aligned with feeding guilds (herbivores, predators, parasitoids, pollen/nectar feeders, and detritivores) based on the predominant feeding behaviour of adults, where known (see references in Appendix B). Identification of all individuals to family or lower was not possible due to the high quantity and diversity of individuals within each sample. Counts of Acari, Collembola, and Thysanoptera were excluded from analyses, as sampling methods were not considered appropriate for obtaining reliable counts for these taxa. All invertebrates other than Acari, Collembola, and Thysanoptera were removed using forceps and stored in 70% ethanol after sorting.

4.3.3 Data analysis

All statistical analyses were performed within R V4.0.2 (R Core Team, 2020). Analysis involved two stages: model-based multivariate analysis to determine effects of treatment (shelterbelt types and controls) on invertebrate community composition, followed by univariate tests to test effects of treatment on abundance of specific invertebrate groups of a priori interest.

The package 'mvabund' (Wang et al., 2012) was used to model (*sensu* Warton et al., 2015) the effects of treatment (shelterbelt types and controls) on composition of invertebrate communities at order or higher taxon level. Tests were run separately for each sampling period (season) using data from the sample level. This method employs a generalised linear model (GLM) framework to analyse multivariate abundance data, fitting a separate GLM to each taxon. This approach to multivariate analysis accounts better for mean-variance relationships in count data, compared to distance-based approaches. In all analyses of multivariate data, a negative binomial distribution was specified as this rectified overdispersion that occurred when Poisson GLMs were used. Taxa which occurred at only two or fewer study sites were excluded from analyses.

The ‘manyglm’ function in ‘mvabund’ was used to fit the model, with ‘treatment’ (shelterbelt types and controls) as the predictor variable. In the first round of multivariate analyses, two treatments were compared: controls (representing open pasture), and shelterbelts (all three shelterbelt types combined). In the second round, the three different shelterbelt types were compared as treatments: (*Pinus radiata*, *Eucalyptus* sp., and mixed native), with controls excluded. For each of these analyses, trap types were combined to compare composition of communities comprised of both ground-active and low-flying invertebrates. Likelihood ratio tests were then applied via the ‘anova’ function to test the effect of treatment on community composition. *P*-values were calculated from 999 resampling iterations using the PIT-trap technique. The ‘gllvm’ package (Niku et al., 2019) was used to plot ordinations of invertebrate count data to visualise differences in community composition between treatments.

From the fitted models in ‘mvabund’, adjusted univariate *p*-values were calculated for individual taxa to test their response to treatment. For those taxa shown to be significantly affected by treatment, and for sub-groups of a priori interest, univariate tests were performed to determine the nature of the treatment effects.

For each season, mean abundance of ground-active (pitfall traps) and low-flying (intercept traps) invertebrates (total, and by order/higher taxa/functional sub-group) was calculated at the site level, using sample level data. Standard deviations for each of the shelterbelt types and the controls were calculated from site level means. To test the effect of treatment on abundance of invertebrates (total, and by functional sub-groups of a priori interest), separate analyses of variance (ANOVA) were conducted on linear mixed-effects models fitted using the ‘nlme’ package in R (Pinheiro et al., 2017), with treatment as the predictor variable and ‘site’ included as a random nested variable. Site level means were used for these analyses, with a fourth root transformation applied for normality. Due to low abundance, counts for some taxa could not be normalised even with the fourth root transformation. These taxa were not analysed further (see Appendix C). For all ANOVA, post hoc comparisons were undertaken using the ‘emmeans’ package in R (Lenth et al., 2018). Where abundance was sufficiently high, trap types were analysed separately to enable more detailed examination of potential habitat-related drivers of treatment effects.

4.4 Results

Results for invertebrate community composition, mean total invertebrate abundance, and mean abundance of select invertebrate taxa and functional sub-groups, are presented in Sections 4.4.1-4.4.3 below. A full list of results is provided in Appendix C.

4.4.1 Invertebrate community composition

When comparing all shelterbelts to open pasture, model-based multivariate analyses showed that invertebrate communities were significantly distinct in summer ($p = 0.005$, Figure 4.4b) and autumn ($p = 0.0120$), although there was some overlap in community composition in autumn (Figure 4.4c). There was no significant difference at the community level in spring ($p = 0.056$, Figure 4.4a). Adjusted univariate tests showed that individual taxa (at the order or higher taxon level) rarely differed significantly between the two treatments (controls vs. all shelterbelts combined), with the exception of Isopoda in spring ($p = 0.016$) and summer ($p \leq 0.001$), and Orthoptera in autumn ($p = 0.046$). Isopoda (slaters), Neuroptera (lacewings), Lepidoptera (moths and butterflies), Orthoptera (grasshoppers and crickets), Diptera (flies), and Hymenoptera (bees, ants, and wasps) accounted for most of the differentiation between these two treatments.

Community composition between the three shelterbelt types only differed strongly in autumn ($p = 0.017$, Figure 4.5c), but not in the other two seasons ($p > 0.350$, Figure 4.5a, b). Post-hoc analysis showed that in autumn, pine was clearly different from eucalypt and mixed native shelterbelts (Figure 4.5c) and the most significant taxa driving this difference were Psocoptera (booklice) ($p = 0.019$), Hymenoptera ($p = 0.025$), and Hemiptera (true bugs) ($p = 0.039$). Other orders which differed between shelterbelt types were Neuroptera, Isopoda, and Coleoptera (beetles) (particularly in spring).

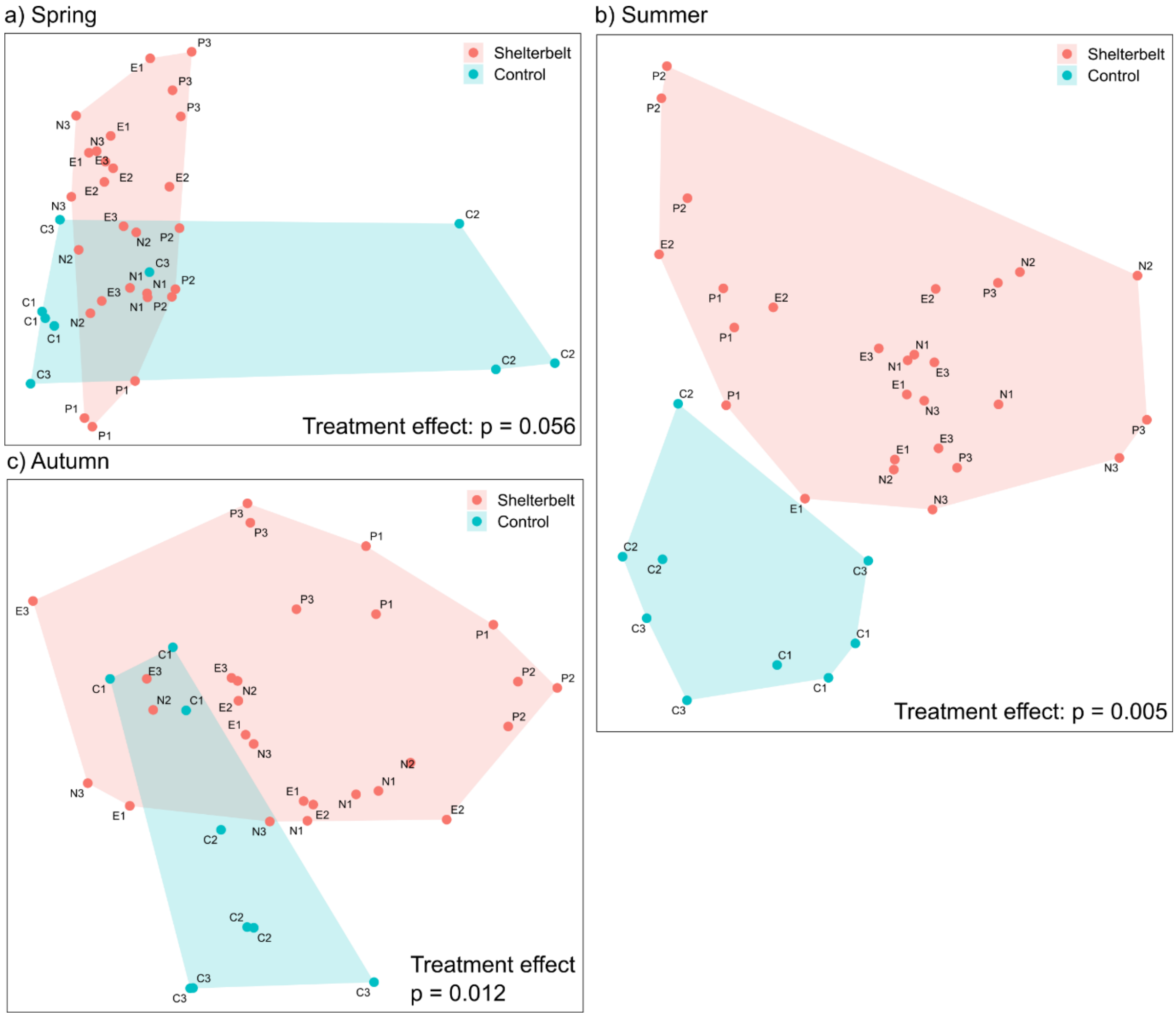

Figure 4.4: Latent variable model-based ordination of invertebrate abundance (ground-active and low-flying invertebrates) at order or higher taxon level, showing differences between controls (open pasture) (blue) and all shelterbelt types combined (red), for (a) spring, (b) summer, and (c) autumn. Point labels refer to individual study sites (e.g. E1 = eucalypt shelterbelt 1).

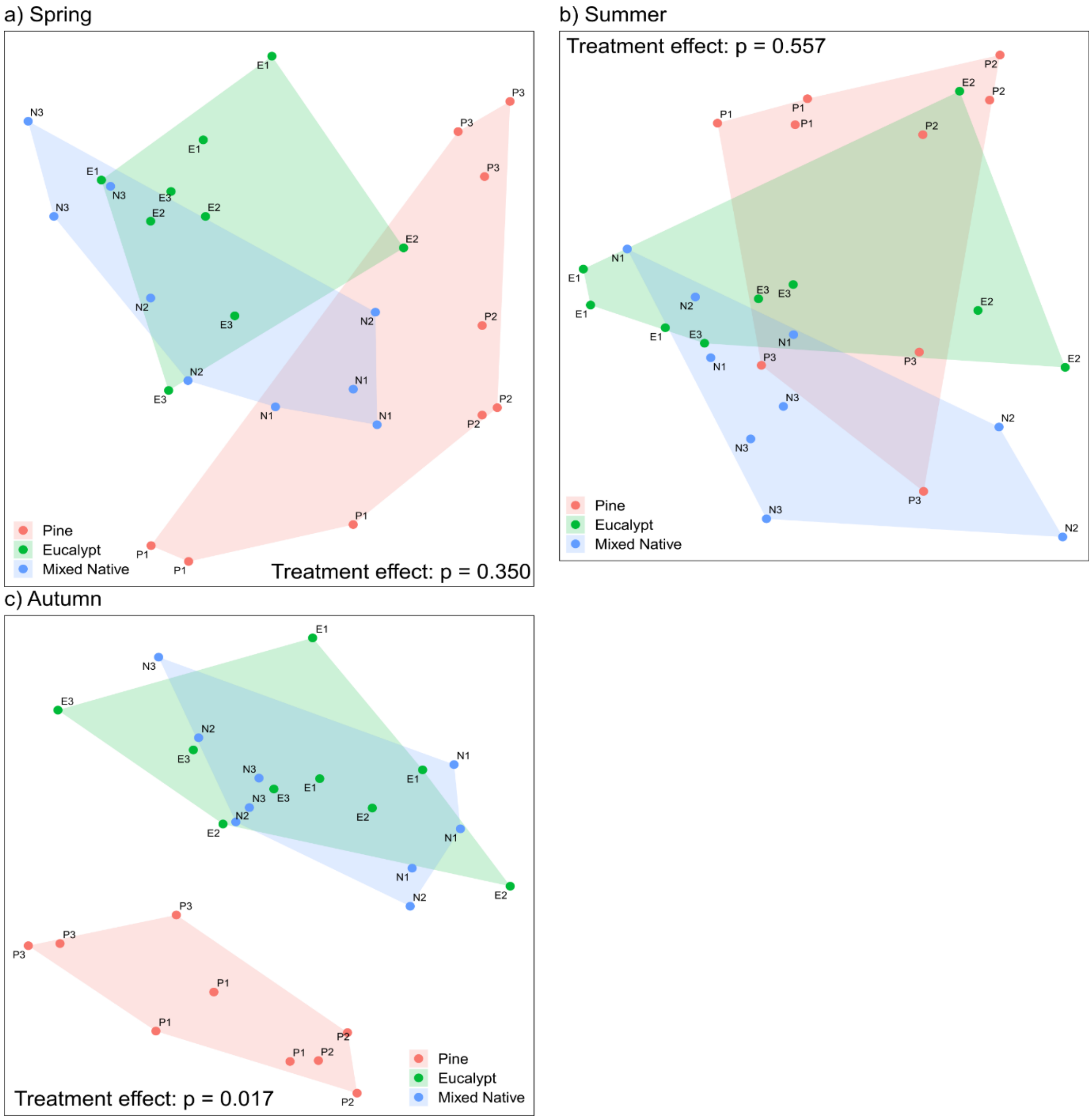

Figure 4.5: Latent variable model-based ordination of invertebrate abundance (ground-active and low-flying invertebrates) at order or higher taxon level, showing differences between three shelterbelt types; pine (red), eucalypt (green) and mixed native (blue). Ordinations are shown separately for (a) spring, (b) summer, and (c) autumn. Point labels refer to individual study sites (e.g. E1 = eucalypt shelterbelt 1).

4.4.2 Mean total invertebrate abundance

Autumn yielded the lowest mean total abundance of both ground-active and low-flying invertebrates, and summer the highest (Figure 4.6). Effects of treatment on mean total invertebrate abundance were generally not significant, although in spring, there were significantly fewer low-flying invertebrates in pine shelterbelts compared to open pasture ($p = 0.015$) and eucalypt shelterbelts ($p = 0.046$). Relatively high abundance of several orders (Isopoda, Coleoptera, Hymenoptera, and Diptera) contributed to higher mean total abundance of invertebrates in mixed native shelterbelts in summer, although this effect was not significant due to high levels of variation between sites.

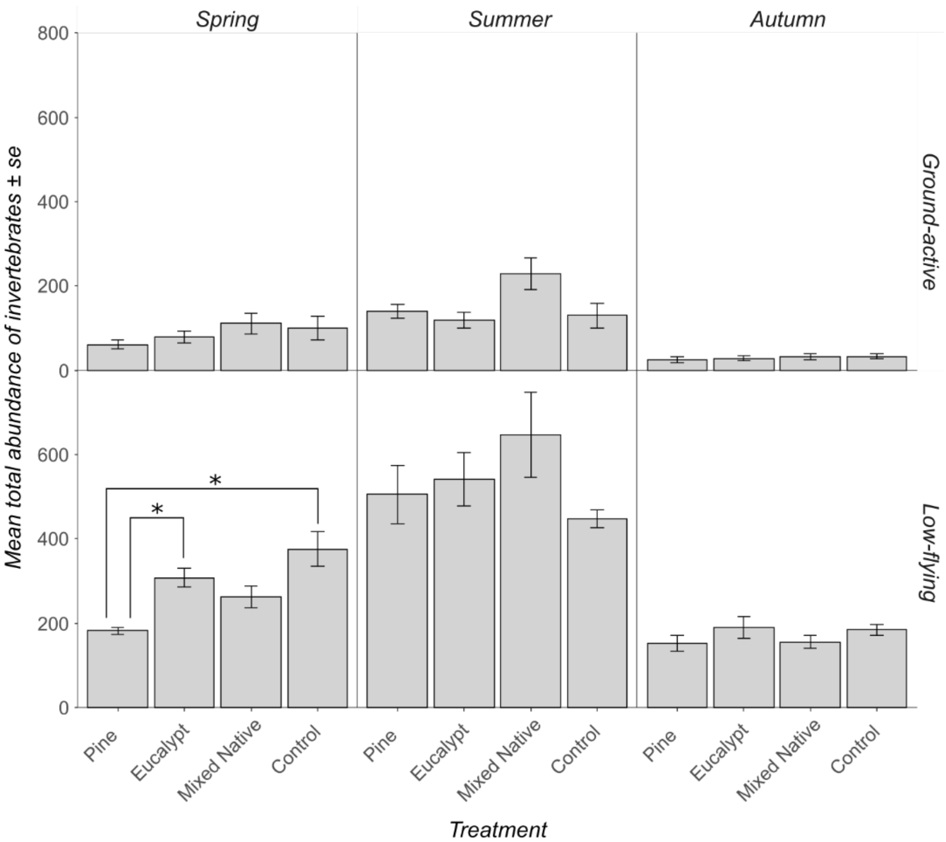

Figure 4.6: Mean total abundance of ground-active (pitfall trap) and low-flying (intercept trap) invertebrates in each shelterbelt type and in open pasture over spring, summer, and autumn. Figure displays mean and standard error for untransformed data, whereas statistics presented in text refer to transformed data (fourth root transformation). Asterisks on brackets between bars denote statistical significance of differences between means (*p < 0.05, **p < 0.01, ***p < 0.001).

4.4.3 Mean abundance of invertebrates by order and functional sub-group

Univariate tests were undertaken to examine treatment effects on abundance of those taxa which contributed most to differentiation in community composition between treatments: Isopoda, Neuroptera, Lepidoptera, Orthoptera, Diptera, Hymenoptera, Psocoptera, Coleoptera, and Hemiptera. In addition, treatment effects were examined for functional sub-groups of invertebrates (i.e. pests, pollinators, and predators) within the orders Hymenoptera, Coleoptera, and Hemiptera.

Across all three seasons, Isopoda (slaters) were recorded in all shelterbelt types but never in open pasture. There were significantly higher numbers of Isopoda in eucalypt shelterbelts in spring (p = 0.046) and summer (p = 0.031), and in mixed native shelterbelts in summer (p = 0.020), compared to the open paddock. There was a significant treatment effect on mean abundance of ground-active Neuroptera (lacewings) in summer (p = 0.004), with pine shelterbelts having significantly greater mean abundance compared to all other treatments. In the case of low-flying Neuroptera, higher numbers were recorded in open pasture in spring compared to all three shelterbelt types, and in summer there were more low-flying Neuroptera in pine shelterbelts compared to open pasture (p = 0.025). In autumn, there were more ground-active Orthoptera (grasshoppers and crickets) in open pasture compared to mixed native shelterbelts (p = 0.031), and more low-flying Orthoptera in open pasture compared to pine (p = 0.016) and mixed native shelterbelts (p = 0.044). In summer and autumn, both ground-active and low-flying Psocoptera (booklice) were generally recorded in higher numbers in pine shelterbelts compared to the other treatments, although this effect was only found to be significant for low-flying Psocoptera in autumn (p = 0.003). Despite their relatively high abundance, no significant effects of treatment on mean abundance of Diptera (flies), or functional sub-groups within this order, were found.

Significant effects of treatment on mean abundance of Coleoptera (beetles) were only found for low-flying Coleoptera in spring (p = 0.037) and summer (p = 0.016). In spring, there were fewer low-flying Coleoptera in pine shelterbelts compared to open pasture (p = 0.040). In summer, there were fewer low-flying Coleoptera in open pasture compared to both eucalypt (p = 0.045) and mixed native shelterbelts (p = 0.014). Of the functional sub-groups analysed within Coleoptera, significant effects of treatment on mean abundance were found for: ‘pest Scarabidae’, ‘pollinating beetles’, and ‘Elateridae’, which is a large family containing several species that are considered pasture pests in their larval stage (Figure 4.7).

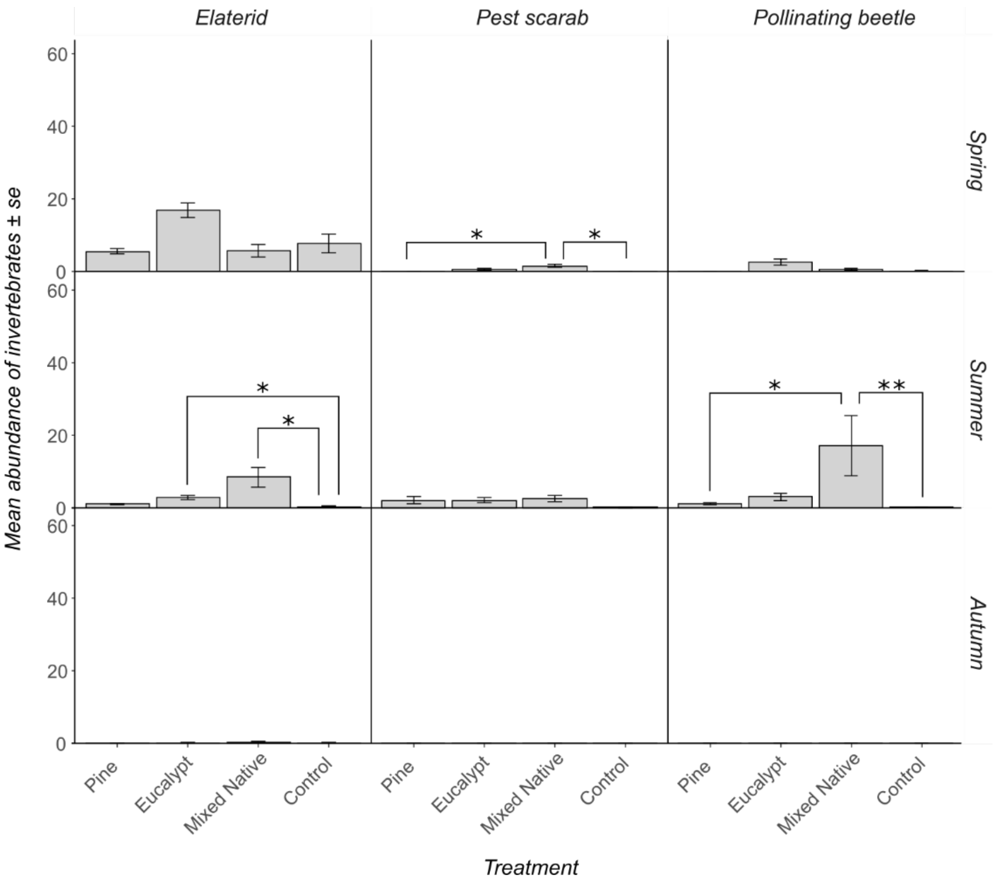

Figure 4.7: Mean abundance of invertebrates in the order Coleoptera by functional sub-group in spring, summer, and autumn. Figure displays mean and standard error for untransformed data, whereas statistics presented in text refer to transformed data (fourth root transformation). Asterisks on brackets between bars denote statistical significance of differences between means (*p < 0.05, **p < 0.01, ***p < 0.001).

The sub-group 'pest Scarabidae', which included pasture and tree pest species of Scarabidae excluding chafers (Appendix B), were generally scarce (<10 individuals per sample). In spring, there were more 'pest Scarabidae' in mixed native shelterbelts compared to pine shelterbelts ($p = 0.017$) and open pasture ($p = 0.017$). In summer, there were lower numbers of beetles in the family Elateridae in open pasture compared to mixed native ($p = 0.019$) and eucalypt shelterbelts ($p = 0.046$). For pollinating beetles in summer, higher numbers were recorded in mixed native shelterbelts compared to open pasture ($p = 0.002$) and pine shelterbelts ($p = 0.023$). The relatively abundant sub-group 'other beetles', which included all Coleoptera not belonging to functional sub-groups, were generally less abundant in open pasture compared to the shelterbelts.

Effects of treatment on mean abundance of Hemiptera (true bugs) were minimal, although in autumn there were significantly more Hemiptera in pine shelterbelts compared to eucalypt shelterbelts ($p = 0.018$), mixed native shelterbelts ($p = 0.047$), and open pasture ($p = 0.006$). Of the functional sub-groups analysed within Hemiptera, significant effects of treatment on mean abundance were found for Aphididae and Psyllidae (Figure 4.8).

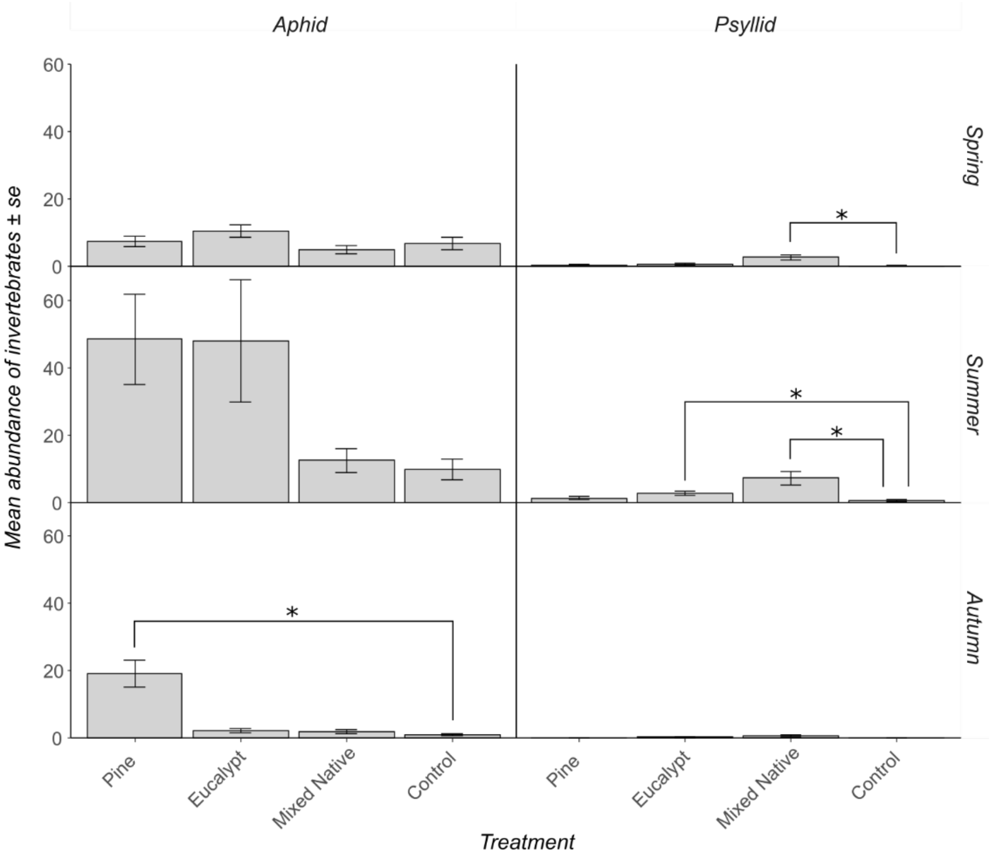

Figure 4.8: Mean abundance of invertebrates in the order Hemiptera by functional sub-group in spring, summer, and autumn. Figure displays mean and standard error for untransformed data, whereas statistics presented in text refer to transformed data (square root transformation). Asterisks on brackets between bars denote statistical significance of differences between means (*p < 0.05, **p < 0.01, ***p < 0.001).

In autumn, more Aphididae (aphids) were recorded in pine shelterbelts compared to open pasture ($p = 0.036$). In spring, more Psyllidae (psyllids) were recorded in mixed native shelterbelts compared to the open pasture ($p = 0.021$). In summer, there were fewer Psyllidae in the open pasture compared to mixed native ($p = 0.010$) and eucalypt shelterbelts ($p = 0.030$) (Figure 4.8).

For ground-active Hymenoptera (bees, ants, and wasps), a significant effect of treatment on mean abundance was found in summer ($p = 0.043$), when more were recorded in mixed native shelterbelts compared to pine shelterbelts ($p = 0.043$). In spring and summer, there were more low-flying Hymenoptera in open pasture compared to pine shelterbelts ($p = 0.021$ and 0.025, respectively). In autumn, there were more low-flying Hymenoptera in open pasture compared to both pine ($p = 0.002$) and mixed native ($p = 0.045$) shelterbelts. Of the functional sub-groups analysed within the order Hymenoptera, significant effects of treatment on mean abundance were found for: ‘predatory ants’, ‘bumblebees’, ‘honey bees’, and ‘native bees’ (Figure 4.9).

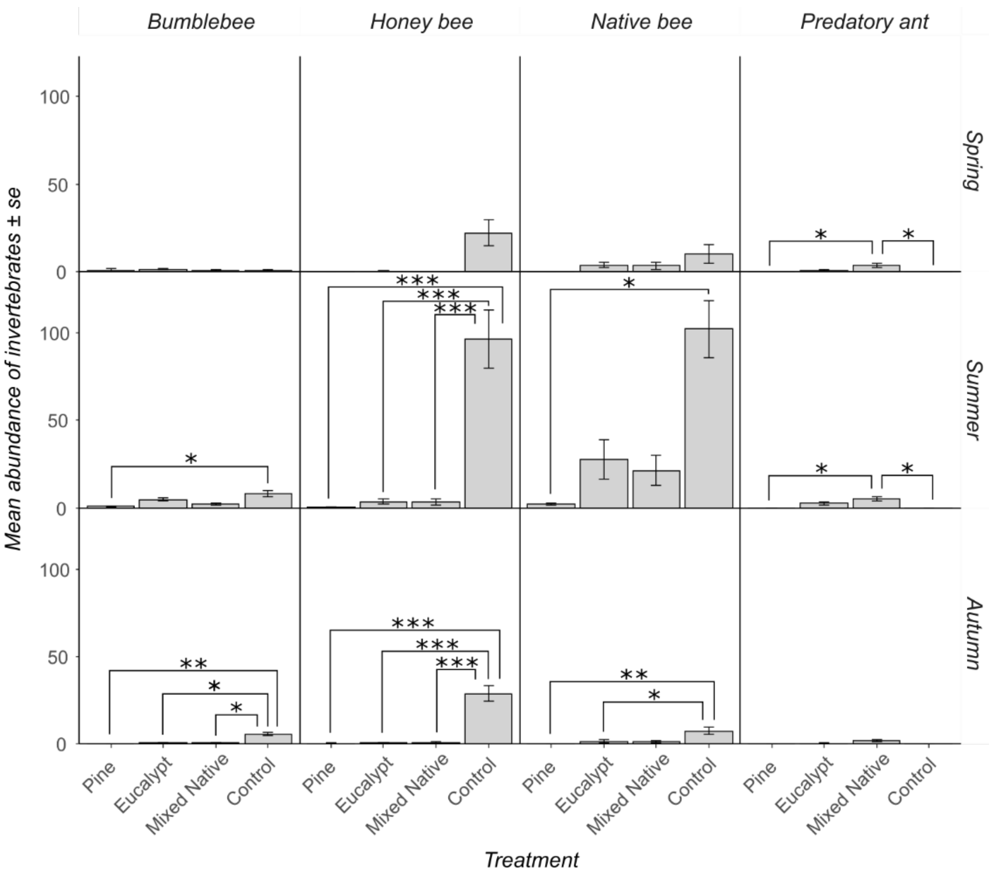

Figure 4.9: Mean abundance of invertebrates in the order Hymenoptera by functional sub-group in spring, summer, and autumn. Figure displays mean and standard error for untransformed data, whereas statistics presented in text refer to transformed data (fourth root transformation). Asterisks on brackets between bars denote statistical significance of differences between means (*p < 0.05, **p < 0.01, ***p < 0.001).

Ants (family: Formicidae) were separated into two functional sub-groups: 'predatory ants' and 'other ants'. In spring and summer, there were more predatory ants in mixed native shelterbelts compared to pine shelterbelts ($p = 0.019$ for both seasons) and open pasture ($p = 0.019$ for both seasons) (Figure 4.9). Bees (superfamily: Apoidea) were separated into three functional subgroups: 'bumblebees', honey bees', and 'native bees'. For all sub-groups within Apoidea, there were consistently higher numbers in open pasture compared to the three shelterbelt types. In summer there were more bumblebees (*Bombus terrestris*) in open pasture compared to pine shelterbelts ($p = 0.050$). In autumn there were more bumblebees in open pasture compared to pine ($p = 0.005$), eucalypt ($p = 0.015$), and mixed native shelterbelts ($p = 0.033$). There were also significant effects of treatment on mean abundance of honey bees (*Apis mellifera*) in summer ($p < 0.001$) and autumn ($p < 0.001$) (Figure 4.9). In both seasons there were more honey bees in open pasture compared to all three types of shelterbelt. In summer there were more native bees (all other Apoidea) in open pasture compared to pine shelterbelts ($p = 0.010$), and in autumn there were more native bees in open pasture compared to both pine ($p = 0.004$) and eucalypt shelterbelts ($p = 0.017$) (Figure 4.9).

4.5 Discussion

This study examined how changes in the condition of shelterbelts affect invertebrate communities at fine scales. Shelterbelts were shown to support invertebrate communities that differed in composition to those in open pasture, with key differences observed in Isopoda, Neuroptera, Hymenoptera, and Orthoptera. Tree species selection in shelterbelts was also shown to affect community composition, as well as local abundance of specific taxa including Hymenoptera, Hemiptera, Neuroptera, Psocoptera, and Coleoptera. Most effects of tree species selection varied between seasons (spring, summer, and autumn) likely due to seasonal changes in invertebrate activity and vegetation characteristics (e.g. flowering). These findings highlight the role of shelterbelts as habitat for invertebrates on farms, as well as their potential role in contributing to agricultural production through enhancement of key ecosystem services such as pollination.

4.5.1 Effects of shelterbelt type on overall abundance and community composition

There were stronger and more consistent contrasts in invertebrate community composition when comparing communities within shelterbelts to those in open pasture, compared to comparisons between communities within the three different shelterbelt types. However, communities within each of the three shelterbelt types were found to differ significantly from one another in autumn. These results suggest that shelterbelts play an important role in supporting invertebrate communities in agricultural landscapes (i.e. by supporting communities that differ to those in open pasture). Minor differences in invertebrate community composition between shelterbelt types indicates that structural differences between shelterbelts (Chapter 3) result in different levels/types of support for these communities. These findings are consistent with other studies which have demonstrated that agroforestry assets such as shelterbelts can influence local invertebrate communities (e.g. Bowie et al., 2014; Rowe et al., 2011). There was also greater variation in the composition of communities within shelterbelts of the same type (Figure 4.4), compared to variation observed in open pasture. This high level of between-site variation in shelterbelts compared to the open pasture may reflect the influence of subtle differences in condition or vegetation structure between individual shelterbelts within the same treatment group, or the relative lack of diversity in habitat niches within open pasture. It is important to acknowledge that identification of invertebrates to lower taxonomic levels, although not practical in this study, would likely increase the compositional differences between communities.

The strongest differentiation between communities in open pasture and shelterbelts occurred in summer (Figure 4.4b). This suggests that microclimate (e.g. temperature, exposure) may be a key factor in determining invertebrate community composition in shelterbelt pasture systems. Microclimatic differences between shelterbelts and open pasture are likely to be greatest in summer (i.e. hotter, drier conditions in the open paddock), with differences attenuating into autumn, and spring being generally moist following winter rains. However, stronger treatment effects in summer could also be linked to higher abundance/activity of invertebrates during this season (resulting in larger sample sizes). Future studies could test effects of microclimate by directly measuring variables such as temperature and soil moisture, or by comparing irrigated and non-irrigated pastures.

While results showed clear differences in community composition, no clear difference was found between the total abundance of invertebrates within shelterbelts compared to open pasture, or between shelterbelt types. The lack of a discernible treatment effect may be partly due to the high levels of variation in counts of total abundance, resulting from localised and temporary fluctuations in abundance of particular invertebrates. Paddocks adjacent to shelterbelt sites selected for this study were also subject to minimal disturbance (i.e. no spraying of pesticides and no tillage) during the year over which sampling occurred. This may have influenced results for total abundance, as shelterbelts may ordinarily serve as important refugia from mortality events associated with disturbance (Marshall & Moonen, 2002). There were however some key differences in total abundance observed in certain seasons e.g. pine shelterbelts contained fewer low-flying invertebrates in spring, and mixed native shelterbelts generally supported higher numbers of invertebrates in summer. These results suggest that the specific benefits of shelterbelts and shelterbelt type vary seasonally. For example, the benefits of mixed native shelterbelts in summer could be due to greater diversity and abundance of flowering plants. Results relating to relative abundance of individual orders (or higher taxa) and functional sub-groups provide further insight on the influence of shelterbelts on particular invertebrate groups.

4.5.2 Effects of shelterbelt type on relative abundance of invertebrate orders and sub-groups

Some invertebrate taxa (e.g. Coleoptera, Psocoptera, and Hemiptera) were generally more abundant in shelterbelts than in open pasture, and some (e.g. Isopoda and Scorpiones) only occurred within shelterbelts. This suggests that certain taxa prefer microclimatic conditions within shelterbelts or use resources (e.g. habitat or food) that are present in shelterbelts and potentially absent in open pasture. This is consistent with the large body of literature that demonstrates the use of non-crop vegetation by invertebrates as habitat (Gurr et al., 2017). Within these taxa, some preferences for particular shelterbelt types were observed. For example, Isopoda were more abundant in eucalypt and mixed native shelterbelts, Psocoptera and Hemiptera were more abundant in pine shelterbelts, and Coleoptera (particularly phytophagous sub-groups) were generally more abundant in mixed native shelterbelts. Preferences of these groups for particular shelterbelt types suggests the influence of vegetation characteristics (e.g. structure, coarse woody debris, and plant species diversity) that are known to differ between the three shelterbelt types (Section 4.5.3).

Conversely, taxa such as Orthoptera and some Hymenoptera (namely Apoidea) were found almost exclusively in open pasture. Others such as Neuroptera exhibited seasonal preferences, mostly occurring in open pasture in spring, but then concentrating in pine shelterbelts in summer. This suggests that some taxa may seek resources that are more readily available in open pasture, such as grass, pollen, or prey. These particular groups are also relatively mobile, which may contribute to their ability to forage widely and avoid disturbance. In the case of Neuroptera, concentration in pine shelterbelts in summer may be linked to the simultaneous high abundance of Aphididae, which are common prey for many species of lacewing (Bailey, 2007). Although Apoidea exhibited an overall preference for open pasture, some sub-groups also showed subtle preferences for particular shelterbelt types. Compared to pine shelterbelts, Apoidea sub-groups (particularly native bees) were more abundant in eucalypt and mixed native shelterbelts, which may reflect the relative availability of food and/or nesting resources in each shelterbelt type (given that pine shelterbelts are non-flowering).

In the case of some taxa (e.g. Diptera) and sub-groups, taxonomic resolution may have been too coarse, and variation in abundance too high, to discern a treatment effect. Identification of individuals to lower taxonomic levels may yield more detail on the effects of shelterbelt type on these invertebrate groups.

4.5.3 Potential influence of vegetation structural characteristics

The suitability of a shelterbelt as habitat for invertebrates is influenced by the structural characteristics of the vegetation within it, as different types of vegetation vary in their capacity to provide food resources (e.g. foliage, pollen, nectar, prey) and shelter (e.g. microclimate, cover, nesting/breeding sites). Characteristics of vegetation structure were measured in a concurrent study (Chapter 3) which characterised vegetation structure of the same three shelterbelt species types using a larger set of study sites, including those used in this study (Chapter 3). In the concurrent study, vegetation cover distribution, percent cover of litter, percent cover of coarse woody debris, and plant species diversity all showed high levels of variation between the three shelterbelt types (Table 4.1). These differences are likely to drive the relationships that were observed between invertebrate community composition and shelterbelt species type.

Table 4.1: Mean structural characteristics of *P. radiata*, *E. nitens*, and mixed native shelterbelts aged 15-30 years. Data is a summary from Chapter 3, Table 3.1. Standard deviations are presented in parentheses. A detailed description of the methods used to measure these structural characteristics can be found in Chapter 3.

Shelterbelt type	**Total number of plant species**	**Cover of coarse woody debris (%)**	**Litter cover (%)**	**Understorey cover (%)**	**Mid-storey cover (%)**
P. radiata	1.9 (0.2)	2.5 (3.0)	78.1(23.5)	1.2(1.8)	1.1(1.8)
E. nitens	4.8 (1.5)	8.0 (4.9)	50.3(29.7)	11.7(20.7)	1.7(1.8)
Mixed native	12.1 (2.3)	4.4 (3.2)	50.5(27.3)	20.7(20.0)	8.1(9.6)

Although differences in invertebrate community composition between shelterbelt types were subtle, particularly at the coarse taxonomic resolution used in this study, several taxa exhibited preferences for mixed native and/or eucalypt shelterbelts over pines. In the case of Isopoda, this may be due to the greater amounts of coarse woody debris present in these shelterbelt types, as Isopoda are most often found in cryptozoic micro-habitats (e.g. under tree bark) (Warburg, 1993). Higher abundance of Apoidea sub-groups in eucalypt and mixed native shelterbelts compared to pines may be linked to a combination of characteristics such as plant species diversity, understorey/mid-storey cover, or coarse woody debris. In the case of native bees, these characteristics affect habitat suitability through provision of nesting sites and alternative pollen/nectar sources (Threlfall et al., 2015; Wilson et al., 2021). Phytophagous sub-groups of Coleoptera exhibited preferences for mixed native shelterbelts, which may be linked to their higher plant species diversity and associated capacity to provide a more diverse range of plant food resources. Higher abundance of these taxa in eucalypt and/or mixed native shelterbelts may also reflect preferences for vegetation that more closely resembles native remnant habitat. Groups such as Psocoptera and some Hemiptera were more abundant in pine shelterbelts. Although it is unclear which vegetation characteristics may have produced this response, similar patterns have been observed for these taxa in other studies (Carver & Kent, 2000; Cole et al., 1989).

Future studies could expand upon these findings by gathering relevant structural data (e.g. litter type/depth and floral availability) at each site during each trapping period to directly test the response of invertebrate community composition to specific vegetation characteristics. More targeted invertebrate sampling techniques may also be required for this approach (e.g. litter sampling and canopy sampling).

4.5.4 Implications for shelterbelt design and ecosystem services

For farmers, implications of findings from this study may depend on whether their objective is to enhance provision of specific invertebrate-related ecosystem services, or to support invertebrates in the landscape more generally. Our results highlight that shelterbelts in low-complexity agricultural landscapes, regardless of tree species selection, will support invertebrate communities that differ from those in open pasture. Although higher numbers of invertebrates were sometimes observed in mixed native shelterbelts compared to other shelterbelt types, these differences were not significant due to high levels of between-site variation. If the farmer's objective is to contribute to conservation of invertebrates in these landscapes, it follows that any of the three shelterbelt types included in this study could contribute to this objective. However, it is important to note that invertebrate diversity was not measured in this study. Identification to species would be required to understand levels of diversity present in each treatment type, which could then be used to infer their relative contribution to conservation of invertebrate biodiversity.

If the objective of establishing shelterbelts is to support particular functional groups of invertebrates, findings from this study confirm the importance of tree species selection. In the case of pest-predators, differences between shelterbelt types were minimal, although more ground-active Hymenoptera including predatory ants were present in mixed native shelterbelts compared to pines. Mixed native shelterbelts, and to a lesser extent eucalypts, were shown to be better at supporting pollinating Hymenoptera and Coleoptera, compared to pines. In the case of pests, differences between treatment types were minimal, although pines supported higher numbers of aphids in some seasons and mixed native shelterbelts supported higher numbers of some pest species of Coleoptera (e.g. Elateridae). These findings, considered in combination with the vegetation characteristics in Section 4.5.3, highlight potential trade-offs between timber production and invertebrate-related ecosystem services. Shelterbelts designed for timber production may lack the plant species diversity and structural complexity required to support beneficial invertebrates such as pollinators and some pest-predators. However, results suggest these trade-offs may be reduced if some development of plant species diversity and structure is achieved through tree/shrub species selection. Identification of plant species which are effective at attracting specific beneficial invertebrate taxa is also an active area of research (e.g. Pfiffner et al., 2009; Winkler et al., 2009). Inclusion of such plant species could enhance provision of services and further reduce trade-offs between invertebrate-related services and timber production.

Findings from this study improve our understanding of the effects of shelterbelt condition on composition of local invertebrate communities. Understanding of these effects could be further expanded by considering both arboreal and soil communities which contribute to biodiversity and ecosystem service provision but were beyond the scope of this study. Additionally, the scale of the impact of shelterbelts in our study may have been underestimated as true 'controls' (i.e. sites within large open paddocks without vegetation or other features) were difficult to find. Although every effort was made to maximise the distance of the controls from shelterbelts and other vegetation, control samples may have been influenced by the presence of such features within their vicinity. There would also be value in expanding or replicating this study to incorporate other configurations of agroforestry such as block plantings, riparian plantings, or retained remnant vegetation. These larger and more 'continuous' configurations may confer fewer negative edge effects and therefore provide higher quality habitat for invertebrates. Comparisons of insect communities between different configurations, as well as examination of effects of tree species selection within each configuration, would greatly improve our understanding of effects of agroforestry asset condition on ecosystem service provision at fine scales.

In considering implications for farmers seeking to enhance invertebrate-related ecosystem services, it is also important to acknowledge that this study does not account for potential indirect effects of shelterbelts on invertebrate communities. Some invertebrate taxa may not use the interior of shelterbelts, but still benefit from their presence. For example, pollinators in adjacent paddocks may benefit from reduced wind speeds (Hennessy et al., 2020), and predators may benefit from shelterbelt-dependent prey dispersing into adjacent areas (Huang et al., 2011) or specialise on the ecotone boundary (Wimp et al., 2011). Beyond the scale of a single paddock, some invertebrate taxa may also benefit from the cumulative effect of having many shelterbelts in the landscape, through improved connectivity (Diekötter et al., 2008) or increased availability/diversity of food resources (Wilson et al., 2021). Further, this study did not examine community differences at shelterbelt edges, where invertebrate abundance may be influenced by availability of a different suite of resources or environmental conditions (Bowie et al., 2014). There would be value in exploring how shelterbelts affect distribution of invertebrates across adjacent paddocks, or across the wider landscape.

4.6 Conclusions

Agroforestry systems provide a range of ecosystem services in agricultural landscape and as highlighted here, have the potential to effectively support invertebrate communities and enhance provision of invertebrate-related ecosystem services. The impact of shelterbelt condition on invertebrate communities at fine scales demonstrates the need to consider tree species selection in the design of future agroforestry systems.

Tree species selection in shelterbelts in the Tasmanian Midlands, a low-complexity agricultural landscape, affected local abundance of invertebrate taxa including those which play functional roles as either pollinators, pests, or pest-predators. Amongst the three shelterbelt species types examined in this study, trade-offs between timber production and invertebrate-related services were identified, with mixed native shelterbelts shown to be more effective at supporting pollinators and pest-predators compared to pines. However, it was also found that shelterbelts offer potential invertebrate conservation benefits regardless of tree species. These findings strengthen our understanding of the effects of shelterbelt condition on composition of local invertebrate communities, thereby improving our ability to predict provision of invertebrate-related ecosystem services in agroforestry systems. Particularly when linked to structural characteristics of shelterbelt vegetation, findings from this study can assist farmers in selecting shelterbelt tree species which best serve their objectives.

While this study focused on direct effects of shelterbelts on invertebrate communities (i.e. habitat suitability), there would be value in expanding this work to explore indirect effects of shelterbelts on invertebrate communities within adjacent paddocks, for example through wind speed reduction. This is the focus of a companion paper by the same authors, which uses an expanded version of this dataset to explore how abundance of different functional groups of invertebrates varies depending on distance from shelterbelt edges.

Chapter 5: Patterns of invertebrate abundance in pastures adjacent to shelterbelts: consequences for ecosystem services

This chapter is in preparation as:

Marais, Z.E., Baker, T.P., Hunt, M.A., and Barmuta, L.A. (in preparation). Patterns of invertebrate abundance in pastures adjacent to shelterbelts: consequences for ecosystem services.

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5.1 Abstract

Ecosystem services provided by invertebrates (e.g. pollination, pest control, and decomposition) are critical to production in many agricultural systems and valued highly by farmers. Agroforestry assets such as shelterbelts are known to support invertebrate communities in agricultural landscapes through provision of refuge, food resources, breeding/nesting sites, and microclimate regulation. However, relatively little is known about how agroforestry enhances the provision of agriculturally-important ecosystem services on farms by influencing distribution of various functional invertebrate groups at fine scales (i.e. the farm or paddock scale). In addition, to effectively understand the value of different types of agroforestry systems, gaps in our understanding of how agroforestry assets of varying condition affect distribution of functionally-important invertebrates at fine scales must be addressed. This study compared invertebrate communities (functional composition, total abundance, and abundance by functional group) at varying distances into open pasture on the leeward side of three common shelterbelt types (*Pinus radiata*, *Eucalyptus* sp., and mixed native) in the Midlands region of Tasmania, Australia over spring (2018), summer (2019), and autumn (2019). Findings from this study demonstrate that shelterbelt agroforestry systems have the potential to enhance paddock-scale provision of invertebrate-related ecosystem services, namely pollination and, to a lesser extent, pest control. Patterns of insect distribution in paddocks adjacent to shelterbelts suggest that wind speed is the primary driver for enhanced pollination potential, and also a potential driver of pest control, although habitat suitability of shelterbelt vegetation may influence provision of both of these services in some seasons. Although shelterbelts provide habitat for decomposers and some minor pests, results indicate that shelterbelts do not influence distribution of these functional groups in adjacent pasture. Overall, findings suggest that while shelterbelt condition (driven by choice of shelterbelt tree species) may affect the functional composition of invertebrate communities within shelterbelts and at their edges, it does not affect composition of communities in adjacent open pasture. Understanding fine-scale distribution of invertebrates in shelterbelt systems improves our ability to predict provision of invertebrate-related ecosystem services. This will enable better communication of the value of agroforestry to farmers and provide scale-appropriate information to inform their decision-making.

5.2 Introduction

Agroforestry systems play an important role in current and future multifunctional agricultural landscapes due to their capacity to enhance provision of multiple ecosystem services which deliver public and private benefits at various scales (Asbjornsen et al., 2014; Baker et al., 2018; Jose, 2009; Smith et al., 2012). To better understand and communicate the value of agroforestry systems, efforts are underway to measure ecosystem services provided by different types of agroforestry across a range of scales. In most industrialised agricultural landscapes, including those in Australia, natural resource management decisions are generally made at the scale of individual farming businesses. Focusing on fine-scale (i.e. farm or paddock-scale) provision of ecosystem services that confer productivity benefits is therefore useful in building a business case for agroforestry adoption and informing agroforestry design decisions (Marais et al., 2019). Although invertebrate-related ecosystem services (e.g. pollination, pest management) are often valued highly by farmers (Chapter 6), effects of agroforestry on provision of these services at fine scales remains understudied. Field studies to assess the distribution of functionally-important invertebrates within agroforestry systems are currently lacking. Such studies are needed in order to measure and value the contribution of agroforestry assets (e.g. shelterbelts) to provision of invertebrate-related ecosystem services at fine scales. Further, understanding how variation in agroforestry asset condition (e.g. tree species selection) affects invertebrate distribution (and therefore service provision) can assist in informing optimal design of agroforestry systems.

Invertebrates play a paradoxical role in agriculture, providing both critical ecosystem services and costly disservices. While some species can cause significant damage to pasture and/or crops when present in high numbers, others provide services upon which many agricultural systems depend. Wild pollinators, for example, provide crucial pollination services to agriculture, with approximately 70% of the world's most important food crops depending at least partly on animal (mainly insect) pollination (Klein et al., 2007; Ollerton et al., 2011). Pollinators provide value to crop production by increasing yield or improving quality, with the extent of benefit or reliance on pollination varying between crops and varieties (e.g. Hudewenz et al., 2014). The often overlooked but essential services of organic matter decomposition and nutrient cycling are provided by a wide range of invertebrate taxa, including dung beetles (Waterhouse, 1974). Integrated pest management is recognised as an effective means to manage pest pressure in many systems, with invertebrate predators and parasitoids largely responsible for control of pests in 33% of cultivated agricultural systems (Hawkins et al., 1999). Supporting pest-predators on farms can provide value by reducing loss of yield for little or no cost (Losey & Vaughan, 2006), with additional environmental benefits associated with reducing requirements for environmentally-harmful pesticides (Chagnon et al., 2015). Agricultural practices that are known to support invertebrates, including agroforestry (e.g. Varah et al., 2013), play an important role in sustaining and potentially enhancing provision of these ecosystem services.

Shelterbelts are a common type of agroforestry and provide multiple farm-scale benefits including wind speed reduction, timber production, carbon sequestration, and biodiversity (Baker et al., 2021b; England et al., 2020). Although productivity benefits provided through wind speed reduction are relatively well-understood (Baker et al., 2018), few studies have explored trade-offs and synergies in co-benefits provided by shelterbelts. In particular, there is a lack of empirical evidence to support measurement and valuation of invertebrate-related benefits. In addition to supporting invertebrate communities that differ to those in adjacent open pasture (Chapter 4), there is some evidence to suggest that shelterbelts influence the abundance and diversity of invertebrates in adjacent paddocks at fine scales (e.g. Thomas et al., 1991; Tsitsilas et al., 2006). However, distribution patterns of functionally-important invertebrates have not been widely demonstrated in shelterbelt systems.

There are several mechanisms through which shelterbelts can affect distribution of functionally-important invertebrates and associated provision of ecosystem services at the paddock scale. Shelterbelts may provide pollen/nectar resources, improve availability/diversity of prey, alter the microclimate to make conditions more hospitable, provide a buffer/refuge from disturbance, or reduce wind speed which may influence dispersal/foraging (Bentrup et al., 2019; Gurr et al., 2017). Although shelterbelt vegetation characteristics are drivers of invertebrate community composition within shelterbelts (Chapter 4), indirect drivers such as microclimate alteration and wind speed reduction could have greater influence on distribution of invertebrates in adjacent paddocks. Shelterbelts can offer crucial opportunities for invertebrate thermal regulation, either directly (providing shaded or sheltered habitat niches) or indirectly (altering air movement and temperature in adjacent areas) (Papanikolaou et al., 2017). Indeed there is evidence that some groups, including predatory beetles (e.g. *Carabidae*), rely on features such as shelterbelts for over-wintering (Thomas et al., 1991). Wind speed has also been shown to affect foraging and dispersal of some flying insects, including honey bees (*Apis mellifera*) (Hennessy et al., 2020) and bumblebees (*Bombus* sp.) (Rao et al., 2019). Most studies exploring effects of wind speed have either been undertaken in artificial environments (e.g. Lewis, 1967) or in systems containing one particular type of shelterbelt (e.g. Dix et al., 1997), which limits potential to explore interactive effects of habitat provision, microclimate, and wind speed. There is a need for more detailed exploration of these different drivers and their influence on distribution of a broad range of invertebrate functional groups.

Shelterbelt systems vary in terms of tree species selection, configuration (their position in relation to other forms of agriculture on the farm), and extent (area of shelterbelts as a proportion of total property area). Shelterbelt tree species selection has minor effects on the composition of invertebrate communities within shelterbelts (Chapter 4), however the extent to which invertebrate distribution patterns adjacent to shelterbelts are affected by tree species selection has not been established. Species selection is a key consideration for farmers, particularly if they intend to harvest commercial wood products from shelterbelts. Understanding how tree species selection impacts paddock-scale provision of invertebrate-related services would enable consideration of trade-offs and synergies between a broader range of co-benefits in the design of shelterbelt agroforestry systems.

This study explores the effects of distance from shelterbelts and shelterbelt tree species selection on distribution of functionally-important invertebrates, using invertebrate samples from shelterbelt-pasture systems in the Midlands region of Tasmania, Australia. While Chapter 4 of this thesis examines invertebrate biodiversity within shelterbelts, this study focuses on the provision of invertebrate-related ecosystem services (e.g. pollination and pest control) in areas adjacent to shelterbelts. We aimed to answer the following questions:

1. Do invertebrate communities differ significantly in their functional composition depending on a. distance from shelterbelts and b. shelterbelt species type, over different seasons (spring, summer, and autumn)?
2. How is paddock-scale distribution of specific functional groups (e.g. major pollinators, predators, pests, and decomposers) affected by shelterbelt species type over different seasons?

5.3 Methods

5.3.1 Study sites

Nine study sites were selected in the Midlands agricultural region of Tasmania, Australia (Figure 5.1). Study sites included three shelterbelt species types (three sites per species type) that are common to temperate Australia (*Pinus radiata*, *Eucalyptus* sp., and mixed native). Although some sites were part of mixed farming systems, all sites consisted of a shelterbelt adjacent to managed pasture at the time of this study. All study sites were located within an area between 41°31′S-41°43′S and 146°50′E-147°10′E spanning elevations of 142-202 m a.s.l. This area experiences an annual temperature range of -4 to 32°C and receives an average annual rainfall of 400–700 mm.

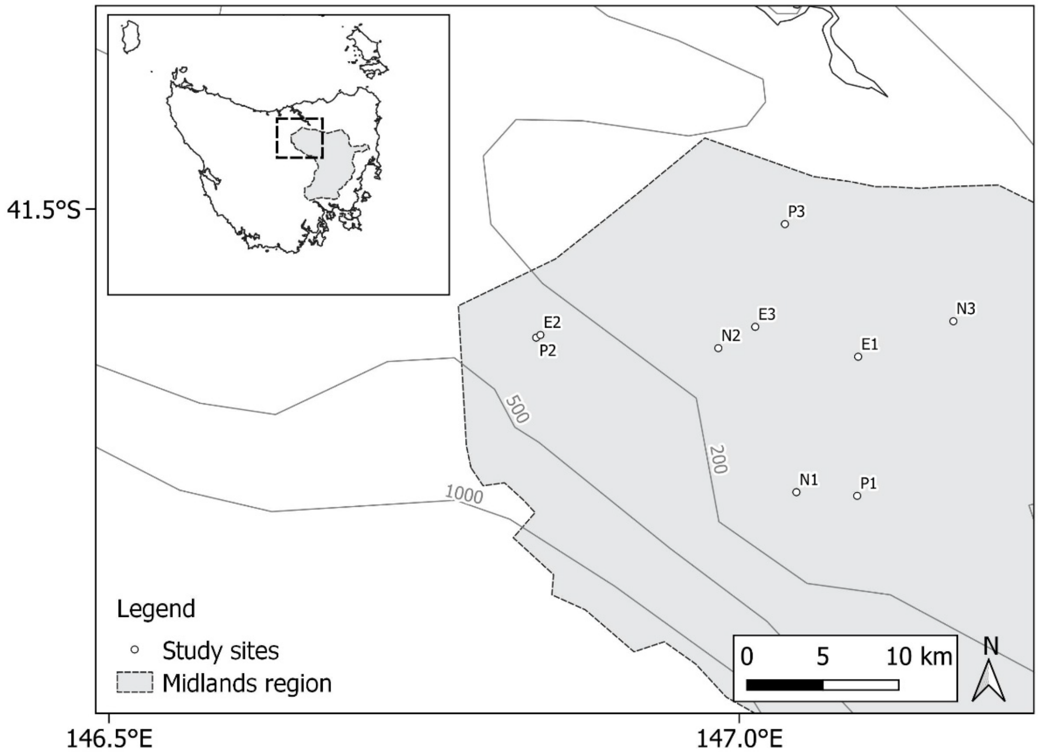

Figure 5.1: Location of study sites within the Midlands agricultural region of Tasmania (grey). 'E', 'P', and 'N' on site labels refer to *Eucalyptus* sp., *Pinus radiata*, and mixed native shelterbelt types, respectively. Topographic lines show height above sea level in metres.

With the exception of one *Eucalyptus* sp. site which consisted of predominantly *E. ovata* with a small amount of *E. pauciflora, Eucalyptus* sp. and *P. radiata* sites were planted as single-species shelterbelts (remaining *Eucalyptus* sp. sites were *Eucalyptus nitens*), typically for wood production. Mixed native shelterbelts, typically planted for biodiversity or carbon rather than wood production, varied in their composition but were all dominated by *Eucalyptus* sp. and *Acacia* sp. and contained a total of at least four Australian native shrub/tree species. Study sites were chosen that had a single shelterbelt adjacent to pasture, with most shelterbelts oriented approximately perpendicular to the prevailing wind direction. As winds in the study area originate mostly from the west and north, shelterbelts included in this study generally had a south-eastern aspect on the leeward side. All sites were planted 15-30 years prior to this study being undertaken. Exact ages of each site were recorded if known, otherwise ages were estimated using available historical aerial imagery. All shelterbelts were at least 200 m in length and 15-30 m wide, measured to the edge of the crown. Stock were excluded from all shelterbelts by fencing.

The Midlands region is a highly modified mixed agricultural landscape. All study sites were considered to have relatively low landscape complexity (<15 % cover of semi-natural forest or grassland within a 1000 m radius). Classifications were based on aerial imagery and publicly-available vegetation mapping (TASVEG 3.0), ground-truthed by visual inspection of study sites.

To limit the influence of other landscape features, sites were selected so that all invertebrate traps were at least 50 m from other integrated non-pasture vegetation (e.g. paddock trees, hedges), at least 100 m from native remnant vegetation patches, plantation forests, and water bodies (e.g. dams, creeks), and at least 200 m from flowering crops (although some flowers were present in the studied pasture). Stock were excluded from the studied pasture during sampling, however during the spring sampling period, stock exclusion at two sites was not possible. At these sites, traps within stocked areas were enclosed within a 1 x 1 m area using wire fencing with 12.5 x 15 cm mesh size. For the duration of the study, soils in adjacent pastures were not disturbed and pesticides were not applied.

5.3.2 Invertebrate sampling

At each site, three transects were established, extending from the centre of each shelterbelt to a distance of 10 tree heights (TH) on the leeward side of the dominant wind direction (Figure 5.2). Transect locations were randomly selected but were at least 50 m apart from each other and at least 50 m from the ends of each shelterbelt. On each transect, there were five sample points (15 sample points per site), including one within the shelterbelt (0 TH) which was located at equal distances from the windward and leeward canopy edges of the shelterbelt. On the leeward side, sample points were located at 0.5 TH, 1.5 TH, 5 TH, and 10 TH away from the shelterbelt (Figure 5.2). Exact distances varied between sites, as the mean dominant height of shelterbelts ranged from 11 to 18 m. Tree height was used to determine distance increments to capture the potential influence of wind speed reduction (as wind speed reduction is a function of shelterbelt height).

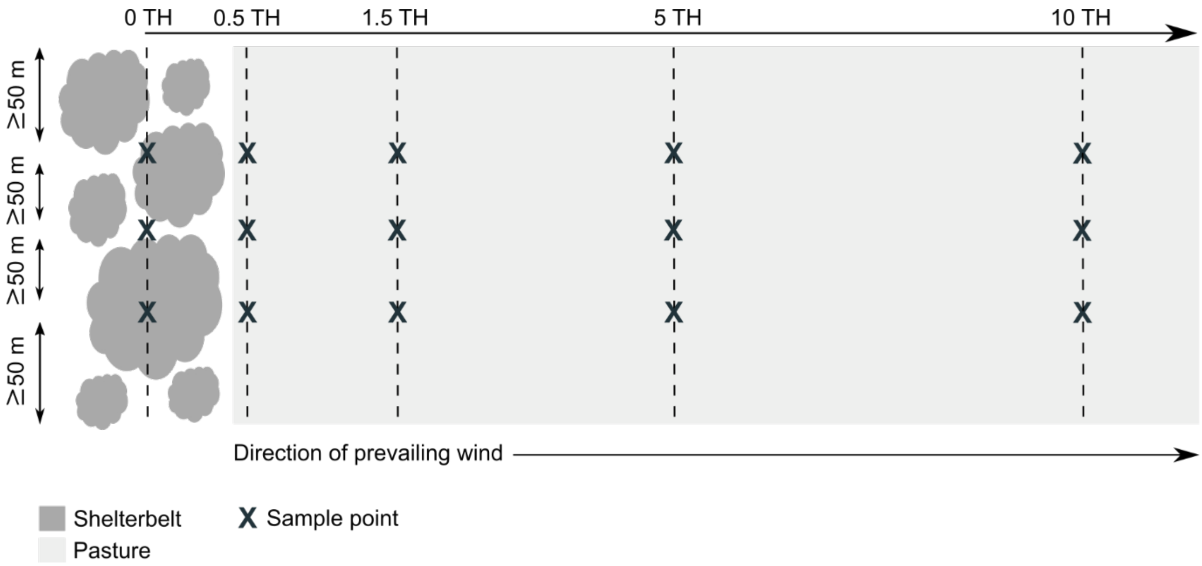

Figure 5.2: Study site configuration showing the location of the transects and sample points in relation to the shelterbelt.

At each sample point, two different types of trap were installed (Figure 5.3). Ground-active invertebrates were sampled using a single pitfall trap which consisted of a 350 mL plastic cup, placed in a hole so that the lip of the cup was flush with the soil surface. The cups were filled with 100 mL of preservative (1:2 propylene glycol-water) with a small amount of detergent added to reduce surface tension. To protect pitfall traps from rain and disturbance by animals, a 17 cm diameter plastic plate was suspended approximately 5 cm above each trap using three wooden skewers. Low-flying invertebrates were sampled using a single window flight intercept trap, placed 30 cm away from the pitfall trap. Each trap consisted of a white 11 L plastic bucket with two intersecting 28 x 30 cm transparent Perspex panels secured over the rim of the bucket, resulting in samples being collected 30-60 cm above ground level. The buckets were filled with 500 mL of the same preservative solution used in the pitfall traps.

Figure 5.3: Photograph showing the window flight intercept (left) and pitfall traps (right) in the field.

Samples were collected over three consecutive seasons: spring (October 2018), summer (January 2019), and autumn (April 2019). Sampling across all four seasons was not possible as the study involved large numbers of samples per site and capacity for processing samples was limited. Although shelterbelts may provide refuge for invertebrates during winter, winter was excluded to prioritise sampling of taxa in groups of particular functional interest (e.g. pollinators) which are generally more abundant and active in warmer months. In each season, traps were operated at all sites concurrently for seven days. During the summer sampling period, intercept traps were topped-up with additional preservative due to high rates of evaporation.

After collection, samples were poured through a 340 µm stainless steel mesh sieve and transferred to a sorting tray with 70% ethanol solution. Invertebrates were sorted to order and sub-order groups (Appendix D), with reference to CSIRO (1991) and the Tasmanian Institute of Agriculture Insect Collection. Sub-order groupings at the family, genus, or species level were aligned with feeding guilds (herbivores, predators, parasitoids, pollen/nectar feeders, and detritivores) based on the predominant feeding behaviour of adults, where known (see references in Appendix D). Sub-order groupings informed designation to one of the following functional groups: generalist predator, pest predator, decomposer (including dung processors), major/minor crop/pasture (field) pest, stock pest, tree pest, or major/minor pollinator. Counts of Acari, Collembola, and Thysanoptera were excluded from analyses, as sampling methods were not considered appropriate for obtaining reliable counts for these taxa. All invertebrates other than Acari, Collembola, and Thysanoptera were removed using forceps and stored in 70% ethanol after sorting.

5.5.3 Data analysis

All statistical analyses were performed within R V4.0.2 (R Core Team, 2020). Analysis involved two stages: model-based multivariate analysis to determine effects of distance (from shelterbelts) and shelterbelt species type on functional composition of invertebrate communities, followed by data visualisation to explore effects of distance and shelterbelt type on abundance of specific functional groups. For all analyses, trap types were combined to compare functional composition of communities comprised of both ground-active and low-flying invertebrates.

The package 'mvabund' (Wang et al., 2012) was used to test the effects of distance and shelterbelt type on composition of invertebrate communities at the functional group level using sample level data, with each sampling period (season) analysed separately. This approach employs a generalised linear model (GLM) framework to analyse multivariate abundance data, fitting a separate GLM to each group, and better accounts for mean-variance relationships in count data than distance-based approaches (Warton et al., 2012). In all analyses of multivariate data, a negative binomial distribution was specified as this rectified overdispersion that occurred when Poisson GLMs were used. Functional groups which occurred at only two or fewer study sites were excluded from analyses.

The ‘manyglm’ function in ‘mvabund’ was used to fit the model, with distance and shelterbelt type as the predictor variables. Distance was specified as an orthogonal polynomial term, as linear specification yielded routinely higher Akaike information criterion (AIC) values. Likelihood ratio tests were then applied via the ‘anova’ function to test the effect of distance, shelterbelt type, and their interaction, on functional composition. *P*-values were calculated from 999 resampling iterations using the PIT-trap technique. From the fitted model in ‘mvabund’, adjusted univariate *p*-values were also calculated for individual functional groups to test their response to shelterbelt type and distance. To visualise differences in functional composition, the ‘gllvm’ package (Niku et al., 2019) was used to plot ordinations of invertebrate count data.

For individual functional groups shown to be significantly affected by distance, data were examined visually to explore the nature of these effects. Mean abundance of invertebrates (total, and by functional group) was calculated at the site level, using sample level data with a fourth root transformation applied. For each season and shelterbelt type, mean invertebrate abundance (total, and by functional group) was plotted against distance as scatter plot with a loess smoothing line applied using the ‘ggplot2’ package (Wickham et al., 2016). For these plots, distance was presented on a $\log_{10}$ axis to improve visualisation of relationships between mean abundance and distance.

5.4 Results

5.4.1 Effects of distance and shelterbelt type on functional composition of invertebrate communities

Model-based multivariate analyses, testing effects of distance and shelterbelt type on functional composition of invertebrate communities across all sample points (within shelterbelts and across adjacent paddocks), showed that functional composition of invertebrate communities was not significantly affected by the type of shelterbelt in any of the three sampling periods (seasons). There was a significant interaction between shelterbelt type and distance ($p < 0.010$) in all seasons, indicating that the effects of distance from shelterbelts on functional composition of invertebrate communities depended on shelterbelt type, although the relationships with distance within each shelterbelt type varied seasonally (Figure 5.4).

Ordinations showed that functional composition varied more between shelterbelt types at distances within and close to shelterbelts (0 – 0.5 TH from shelterbelt edges). At distances further away (i.e. open pasture) the differences were smaller, although there were some outliers at 10 TH (Figure 5.4). This suggests that while shelterbelt type may affect functional composition of communities within shelterbelts and at their edges, it does not affect community composition in adjacent open pasture. Communities at distances furthest away from shelterbelts, i.e. at 5 and 10 TH from shelterbelts, were more similar to one another, suggesting greater homogeneity of invertebrate communities in open pasture than within or next to shelterbelts.

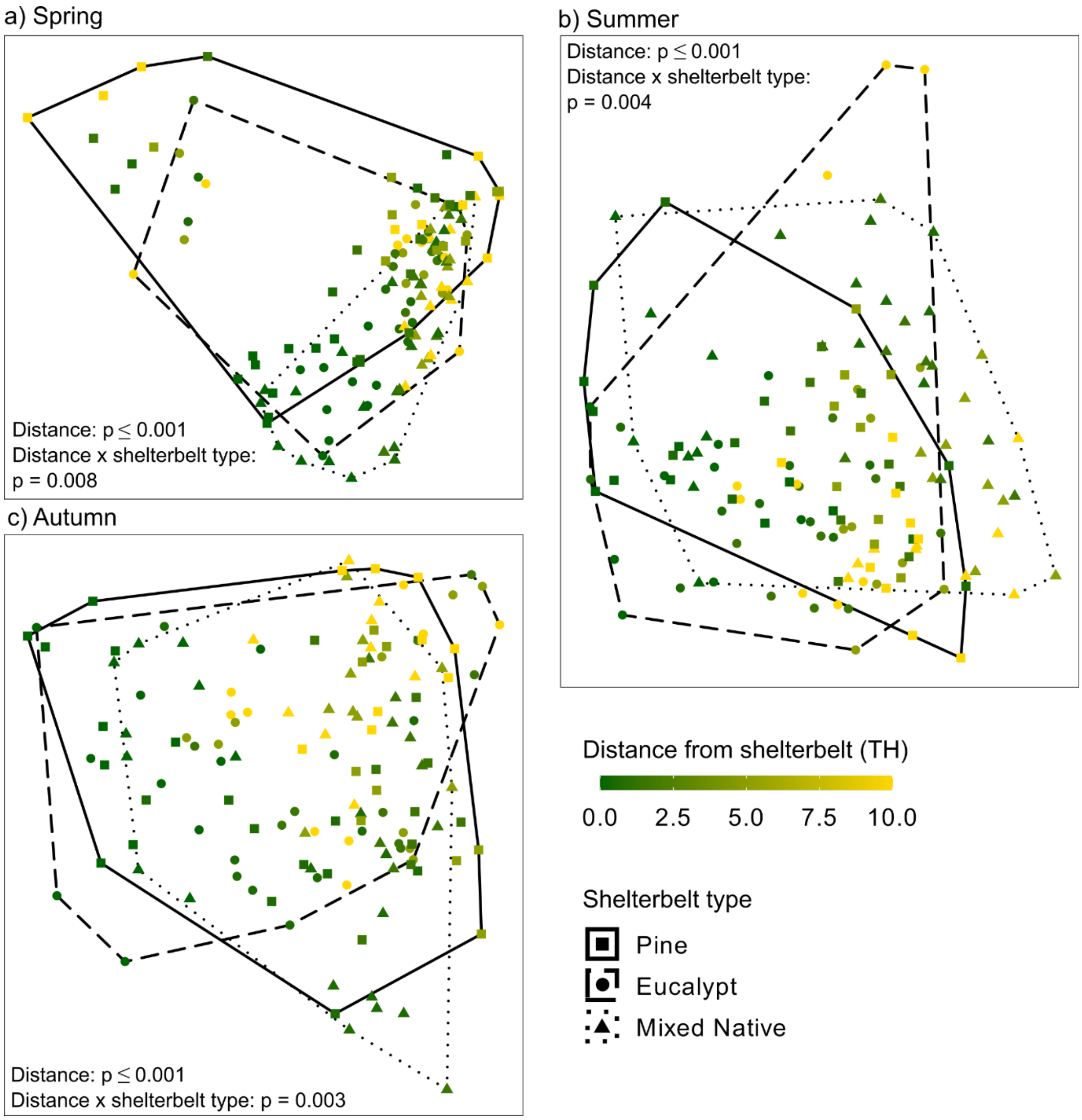

Figure 5.4: Latent variable model-based ordination of invertebrate abundance (ground-active and low-flying invertebrates) at a functional level, showing differences in functional composition of invertebrate communities between shelterbelt types and different distances from shelterbelts, for (a) spring, (b) summer, and (c) autumn. *P* values are presented for effects of 'distance' (from shelterbelts) and the interaction between 'distance' and 'shelterbelt type' on functional composition. *P* values for the effect of 'shelterbelt type' are not presented as this effect was non-significant ($p > 0.05$) for all seasons.

Univariate tests showed that significant effects of distance were common across all functional groups (Appendix D), with particularly strong effects on pollinators, field pests, and detritivores in all seasons (Table 5.1). In general, shelterbelt type did not significantly alter the distance response of individual functional groups, although there was a significant interaction between distance and shelterbelt type for generalist predators in summer ($p = 0.010$) and for minor field pests in autumn ($p = 0.009$). Effects of shelterbelt type were not significant for any functional groups.

Table 5.1: Results of univariate analyses of deviance for each invertebrate functional group, by season. Effects (adjusted p values) are presented for predictor variables 'shelterbelt type' (d.f. = 2), 'distance' (from shelterbelts) (d.f. = 2), and the interaction between 'distance' and 'shelterbelt type' (d.f. = 4) $^{*}p < 0.05$, $^{**}p < 0.01$, $^{***}p < 0.001$. Functional groups which were not significantly affected by any predictor variables ($p > 0.05$) are excluded.

	Shelterbelt type		**Distance**		**Distance x shelterbelt type**	
Functional group	**Deviance**	***P*-value**	**Deviance**	***P*-value**	**Deviance**	***P*-value**
Spring						
Major pollinator	7.056	0.879	82.497	≤0.001***	8.079	0.200
Minor pollinator	5.272	0.904	25.927	≤0.001***	12.275	0.167
Major field pest	0.712	0.974	19.229	0.003**	9.850	0.200
Minor field pest	1.842	0.974	29.571	≤0.001***	4.615	0.574
Tree pest	36.900	0.192	12.747	0.036*	11.585	0.200
Detritivore	9.642	0.879	59.135	≤0.001***	4.910	0.574
Summer						
Major pollinator	12.519	0.977	223.836	≤0.001***	25.926	0.062
Minor pollinator	10.278	0.990	30.018	0.024*	16.489	0.343
Minor field pest	27.533	0.783	36.016	0.006**	4.792	0.879
Other	20.761	0.898	45.180	0.003**	25.511	0.062
Tree pest	36.020	0.548	36.749	0.006**	0.753	0.950
Generalist predator	1.982	0.990	1.525	0.879	45.350	0.010*
Detritivore	10.472	0.990	58.583	≤0.001***	10.373	0.593
Autumn						
Major pollinator	4.478	1.000	163.920	≤0.001***	7.373	0.713
Minor pollinator	4.031	1.000	49.210	0.002**	19.821	0.119
Major field pest	1.222	1.000	33.449	0.002**	1.875	0.987
Minor field pest	25.110	0.712	42.032	0.002**	28.927	0.009**
Other	9.688	1.000	50.160	0.002**	19.068	0.119
Generalist predator	10.358	1.000	19.767	0.017*	19.772	0.119
Detritivore	1.508	1.000	22.449	0.010*	0.850	0.987

5.4.2 Relationships between distance from shelterbelts and total invertebrate abundance

Plots of mean total invertebrate abundance against distance (from shelterbelts) showed minimal influence of distance, with modest differences between seasons (Figure 5.5). In spring, there were no clear effects of distance on total abundance. In summer, total abundance was highest at shelterbelt edges (0.5 TH) for pine and mixed native shelterbelts. In autumn, total abundance was lowest within shelterbelts for all shelterbelt types, with abundance increasing at 0.5 – 1.5 TH from shelterbelt edges before tapering off at 5 and 10 TH. In general, effects of distance on total abundance appeared slightly stronger for pine and mixed native shelterbelts compared to eucalypts.

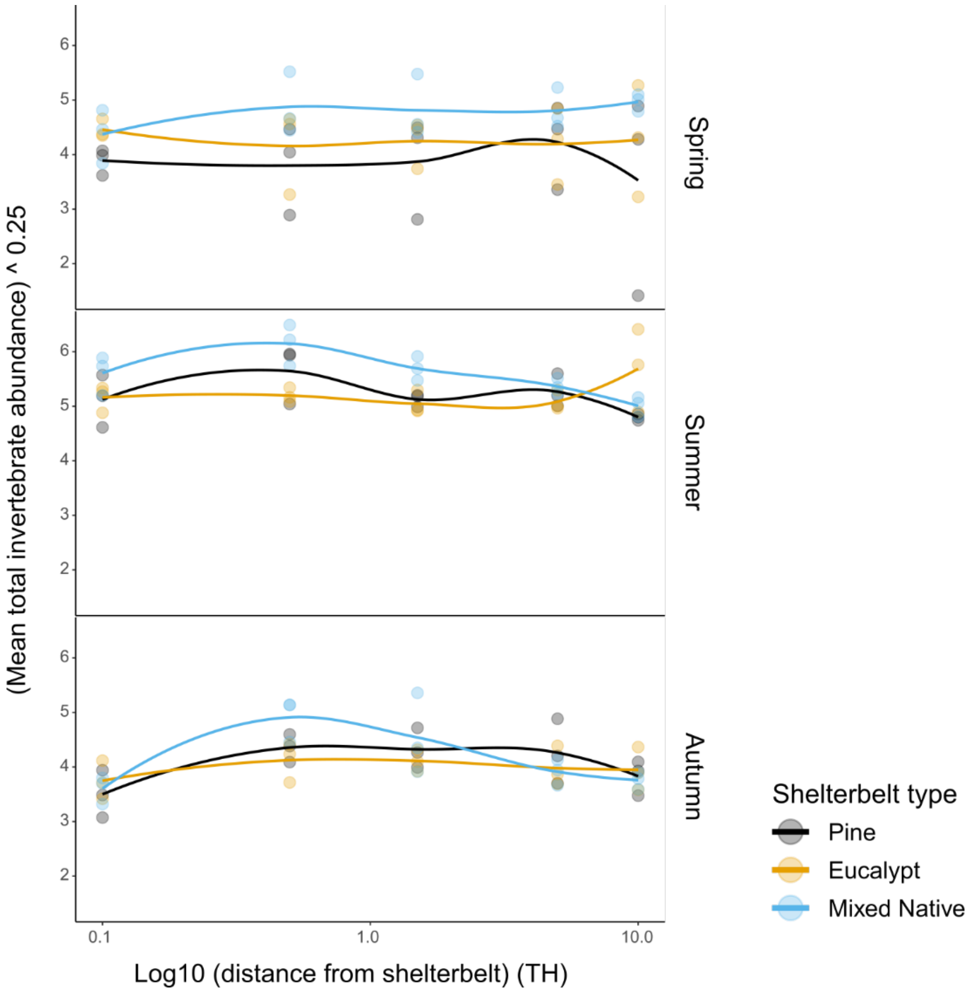

Figure 5.5: Plots of mean total invertebrate abundance (site level means with fourth root transformation applied) against distance from shelterbelts (TH, $\log_{10}$ scaled axis) for each shelterbelt type, for spring, summer, and autumn.

5.4.3 Relationships between distance from shelterbelts and abundance of functional groups

Plots of mean abundance of major and minor invertebrate pollinators against distance (from shelterbelts) showed that effects of distance on pollinator abundance were largely consistent across shelterbelt types and seasons (Figure 5.6). In the case of major pollinators (*Apis mellifera* and *Bombus terrestris*), abundance was very low within shelterbelts and peaked at distances between 1.5 and 5 TH from shelterbelt edges. For pine, and to a lesser extent mixed native shelterbelts, abundance of major pollinators declined slightly beyond 5 TH. While the nature of the relationship between distance and abundance of major pollinators was consistent across seasons, the scale of the response was much larger in summer compared to other seasons (i.e. > 100 individuals present at 10 TH in summer, compared to < 30 in spring/autumn.

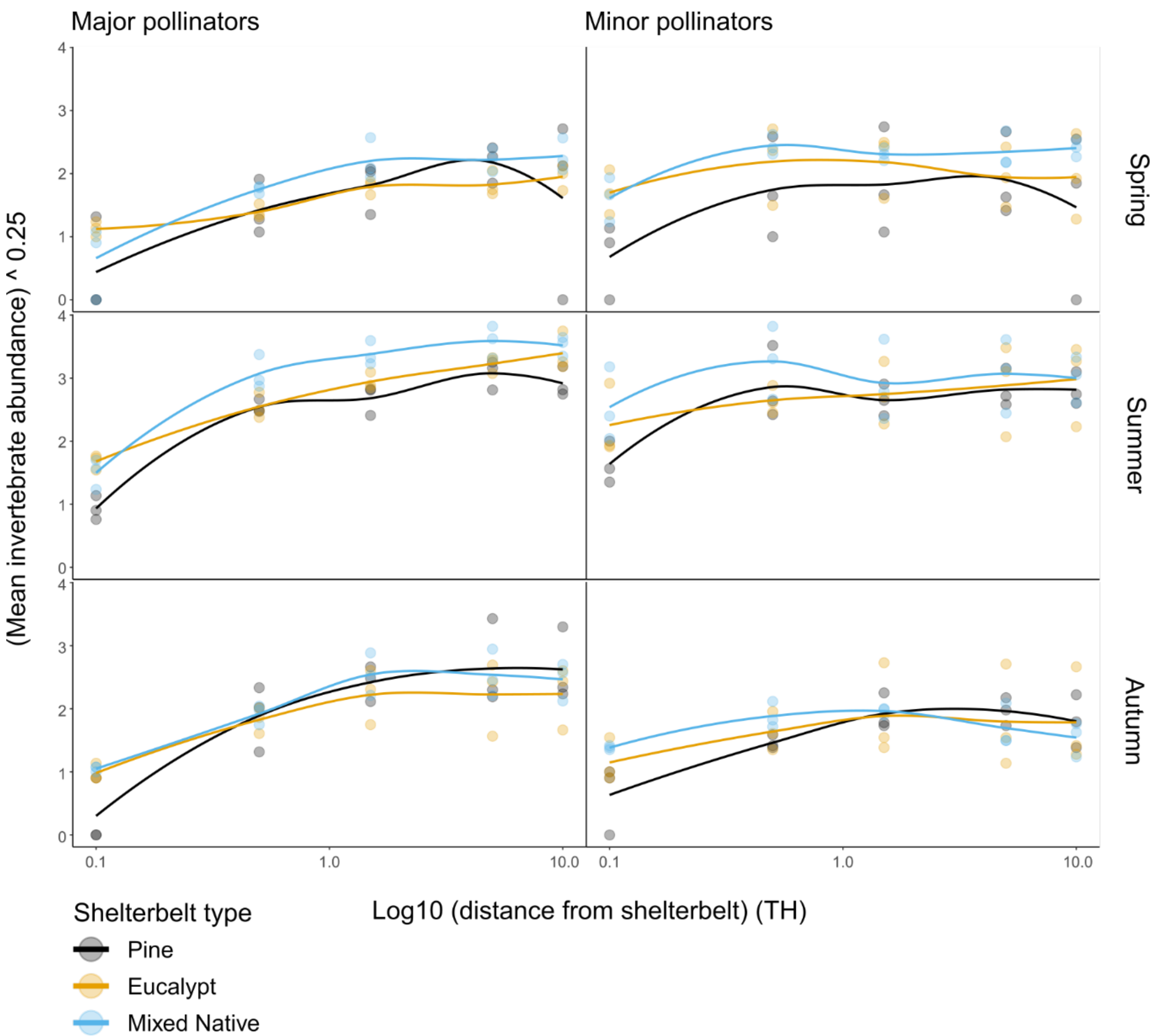

Figure 5.6: Plots of mean abundance of major and minor pollinators (site level means with fourth root transformation applied) against distance from shelterbelts (TH, log_{10} scaled axis) for each shelterbelt type, for spring, summer, and autumn.

For minor pollinators, abundance within shelterbelts was higher in spring and summer compared to autumn. In spring and summer, particularly for mixed native shelterbelts, abundance of minor pollinators peaked at shelterbelt edges (0.5 TH) before levelling off. As with major pollinators, there was a slight decline in abundance of minor pollinators beyond 5 TH in some seasons, particularly for pine and mixed native shelterbelts.

Plots of mean abundance of generalist predator invertebrates against distance showed that effects of distance on abundance of generalist predators were minimal, with little consistency across shelterbelt types and seasons (Figure 5.7). However, for mixed native shelterbelts, generalist predators were slightly more abundant 0.5 – 1.5 TH from shelterbelts compared to open pasture, particularly in spring and autumn.

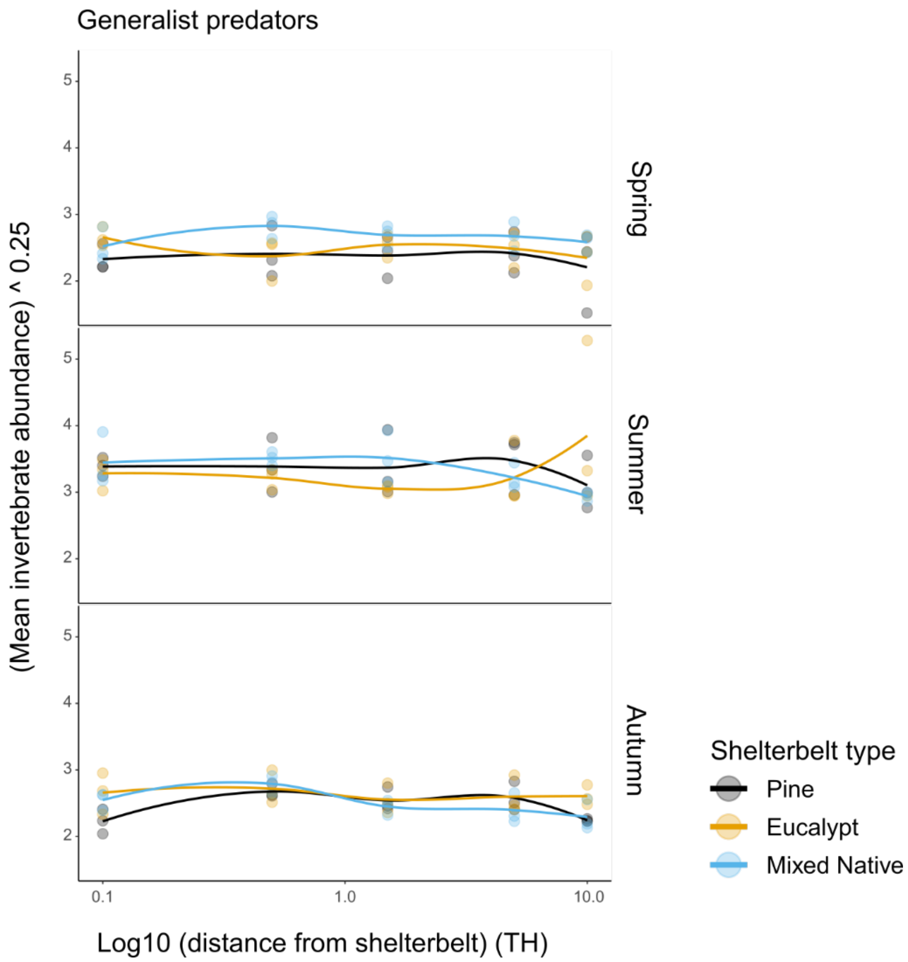

Figure 5.7: Plots of mean abundance of generalist predators (site level means with fourth root transformation applied) against distance from shelterbelts (TH, $\log_{10}$ scaled axis) for each shelterbelt type, for spring, summer, and autumn.

Plots of mean abundance of minor field pests and tree pests showed that effects of distance on abundance of pests varied considerably between seasons and shelterbelt types (Figure 5.8). In summer and autumn, minor field pests were present in high numbers within and at the edges of pine and eucalypt shelterbelts. Otherwise, minor field pests were generally more abundant in pasture (1.5 – 10 TH). In the case of tree pests, abundance was generally low and varied considerably between sites. Abundance of tree pests generally decreased with increasing distance from shelterbelt edges and was higher within and next to mixed native shelterbelts compared to other shelterbelt types.

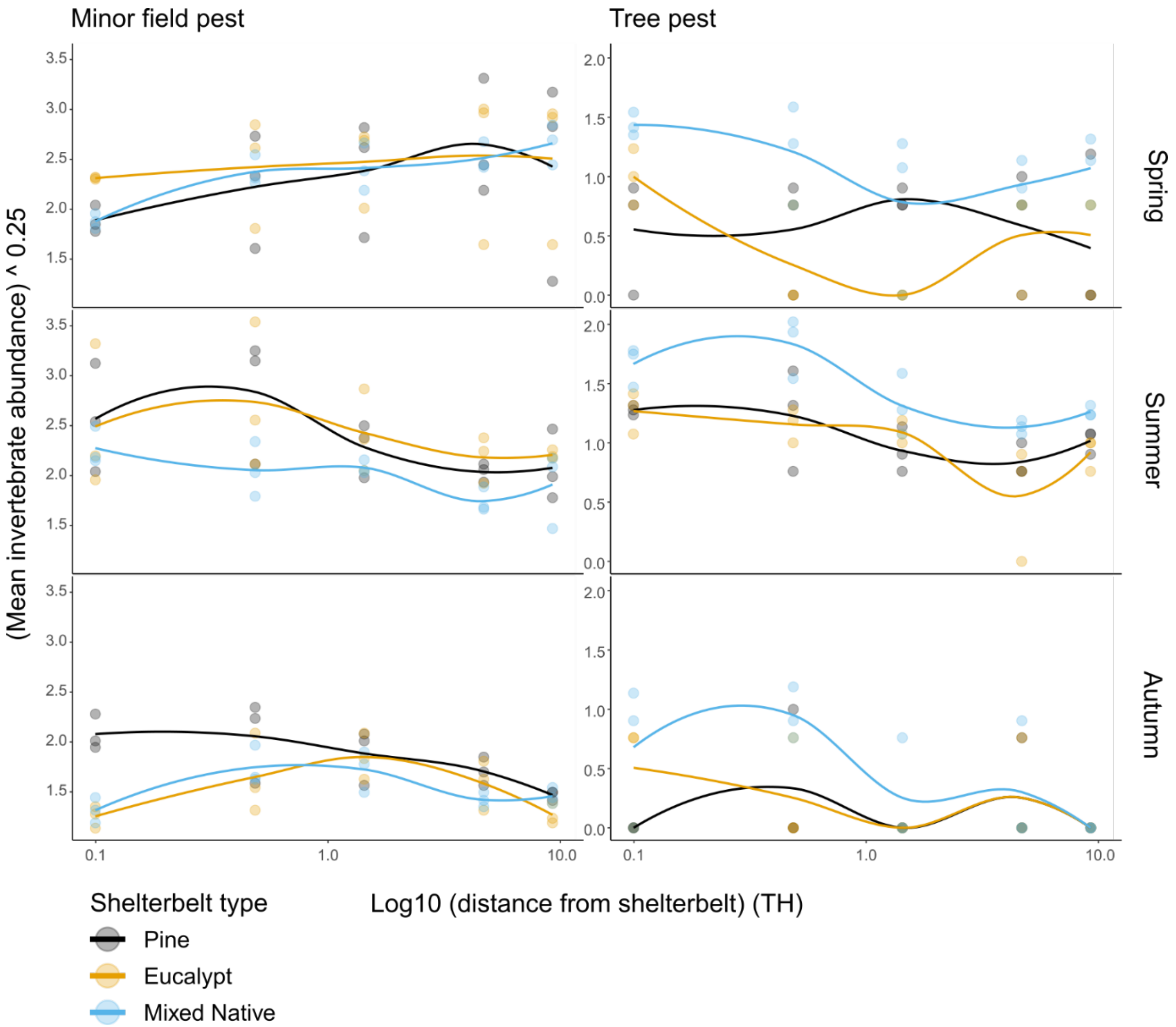

Figure 5.8: Plots of mean abundance of tree pests and minor field pests (site level means with fourth root transformation applied) against distance from shelterbelts (TH, $\log_{10}$ scaled axis) for each shelterbelt type, for spring, summer, and autumn.

Plots of mean abundance of decomposer invertebrates showed that decomposers were more abundant within shelterbelts compared to all other distances, across all seasons and shelterbelt types (Figure 5.9). This suggests a strong habitat preference for shelterbelts over pasture, rather than an effect of distance from shelterbelts.

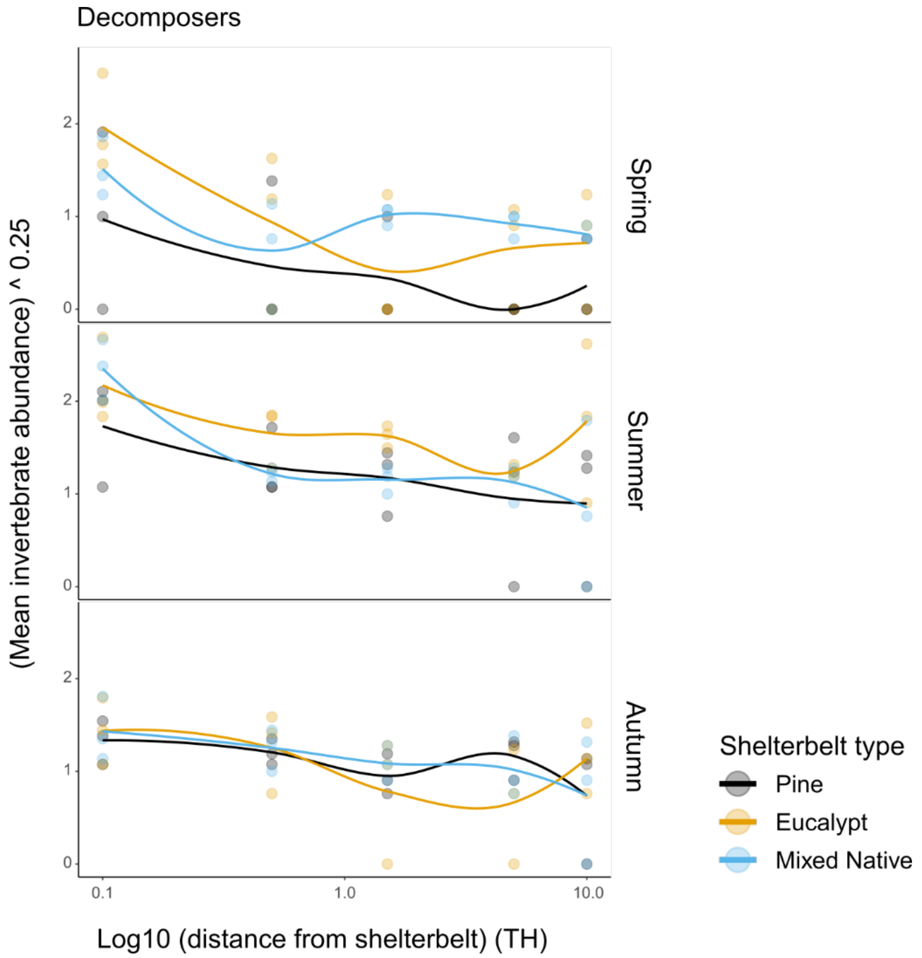

Figure 5.9: Plots of mean abundance of decomposers (site level means with fourth root transformation applied) against distance from shelterbelts (TH, $\log_{10}$ scaled axis) for each shelterbelt type, for spring, summer, and autumn.

5.5 Discussion

This study examined how shelterbelts and shelterbelt condition affect the distribution of functionally-important invertebrates at the paddock scale. Distance from shelterbelts was found to affect functional composition of invertebrate communities in adjacent pasture across all seasons and was a consistent driver of abundance of most individual functional groups tested. Individual invertebrate functional groups showed distinct patterns in response to distance, and abundance patterns of some groups varied between shelterbelt types and seasons. This suggests that groups respond to distance and shelterbelt condition in different ways depending on drivers that most influence their behaviour or dispersal (e.g. wind speed reduction). These findings strengthen our understanding of the effects of shelterbelts on fine-scale distribution of functionally-important invertebrates, thereby improving our ability to predict provision of invertebrate-related ecosystem services at the farm or paddock scale.

5.5.1 Potential drivers of effects of distance from shelterbelts on functional composition

Shelterbelts may influence the distribution and behaviour of invertebrates in agricultural landscapes by providing habitat resources (e.g. food, shelter, space, alternative prey), altering microclimate, or by reducing wind speed in adjacent areas (Gurr et al., 2017; Pasek, 1988). The significant effect of distance on functional composition of invertebrate communities observed in this study is likely to be the outcome of a combination of these factors.

In the case of some functional groups, observed effects of distance on abundance are likely to reflect the stark contrast in habitat between shelterbelt interiors and open pasture, rather than a continuous gradient of environmental change. Shelterbelts provide greater structural complexity in their vegetation and groundcover, higher diversity of flowering plants, and less disturbance of ground surface, compared to open pasture. Open pastures, while offering less structural complexity, contain habitat characteristics (e.g. dense grass/herb cover) that may appeal to certain invertebrate taxa. These habitat contrasts are likely to have driven distance effects in this study that were more dichotomous than continuous, for example the higher abundance of decomposers in shelterbelts (Figure 5.9), lack of major pollinators within shelterbelts (Figure 5.6), and higher abundance of minor field pests in open pasture (Figure 5.8).

Wind speed reduction has been shown to affect the behaviour of a wide range of invertebrate taxa, particularly flying invertebrates such as bees (e.g. Hennessy et al., 2020), beetles (e.g. Fadamiro, 1996), and flies (e.g. Vale, 1983). Results from this study suggest that effects of distance from shelterbelts on abundance of some invertebrate groups are driven by wind speed reduction, particularly in the case of pollinators, which are typically flying insects susceptible to wind. Abundance of pollinators was consistently found to be higher in the zone between 0.5 and 1.5 TH, with abundance dropping off after 5 TH. This zone of high pollinator abundance aligns with the area of greatest wind speed reduction, as shown by Cleugh (2002) (Figure 5.10). Generalist predators, a high proportion of which were small parasitoid wasps, were also found in higher numbers in this area (although not as consistently), suggesting that their distribution may also be partly influenced by wind speed reduction. These results match previous studies showing greater concentration of these taxa in areas with reduced wind speed (e.g. Lewis, 1965; Lewis & Smith, 1969), although this area is under-researched. If wind speed were a key driver of invertebrate distribution in shelterbelt systems, we may expect differences in strength of the distance effect between different shelterbelt types. In this study, we would have expected pine and mixed native shelterbelts to exhibit stronger distance effects compared to eucalypts, as they provide more effective wind speed reduction due to their lower porosity (Chapter 3). Although plots of both total invertebrate and pollinator abundance against distance showed a small trend towards effects of distance being stronger for pine and mixed native shelterbelts compared to eucalypts, this trend was not pronounced - potentially due to variation in porosity between sites and the low number of replicates of shelterbelt type. Sampling periods in this study were also relatively short (7 days) which would have limited opportunities to capture windy conditions. Direct testing of effects of wind speed on abundance of a broad range of invertebrates in shelterbelt systems would be helpful in confirming the role of wind speed as a key driver of invertebrate distribution.

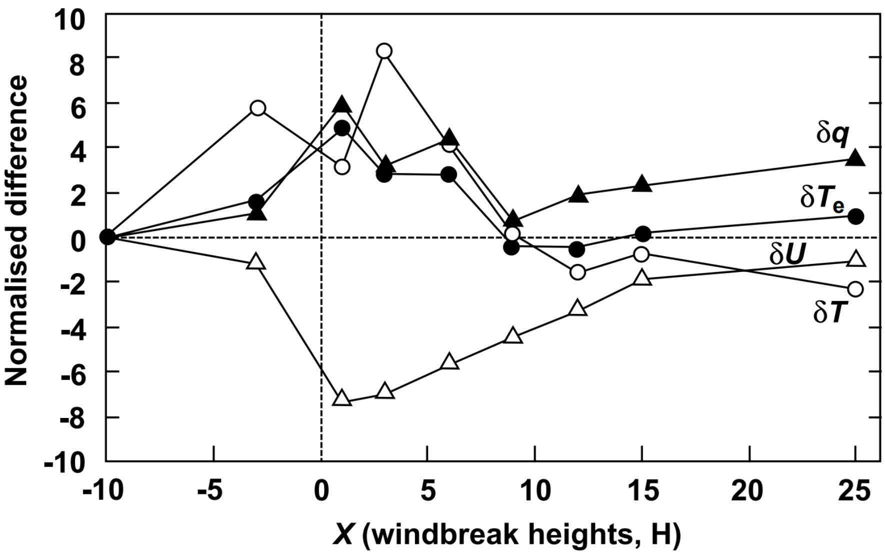

Figure 5.10: Spatial variation of diurnally averaged wind speed (δU, clear triangles), air temperature (δT, clear circles), equivalent temperature (δT_e, solid circles), and humidity deficit (δq, solid triangles). Reproduced from Cleugh (2002).

As shown by Cleugh (2002), shelterbelts also alter temperature and humidity in adjacent areas (Figure 5.10), with effects being greatest in the zone of 1 – 7 TH. These conditions are linked to changes in vegetation growth (e.g. Baker et al., 2021b) and are also likely to affect distribution of livestock across the paddock. Multiple environmental variables, including microclimate, could have contributed to effects of distance on distribution of invertebrates observed in this study. For example, combined influences of edge-related habitat features, stock congregation, and shade provided in the immediate lee of shelterbelts are likely to have driven the slightly higher abundance of some functional groups (e.g. minor pollinators, predators) observed at 0.5 TH. To disentangle the effects of multiple environmental variables, future field studies could aim to measure microclimate variables directly or completely exclude variables such as stock. Effects of distance on distribution of invertebrates may also vary depending on the time of day and the orientation or aspect of the shelterbelt. While all shelterbelts in this study had a similar orientation, broader application of the findings from this study would need to consider shelterbelt orientation and its critical influence on wind speed reduction and shading.

Findings from this study highlight the effects of shelterbelts on fine scale distribution of invertebrates. However, the size and configuration of available study sites led to some limitations in the study design which may have resulted in the underestimation of the magnitude of the effect. For example, it was not possible to consistently sample invertebrates in areas representing baseline conditions, i.e. by extending transects beyond 10 TH or by sampling on the windward side (refer to Figure 5.10), due to the size of the paddocks and the proximity of features such as roads and other vegetation. As a result we did not detect functional composition in areas unaffected by shelterbelts. Further, direct measurement of wind speed and other microclimate variables at sample points was not possible due to resource limitations. Future studies addressing the interactions between factors such as wind speed reduction, microclimate alteration, and habitat provision would further improve our understanding of the effects of shelterbelts on invertebrate distribution at fine scales. Seasonal variation in distance effects observed in this study could also guide design of more targeted studies, focusing on times within the growing season when invertebrate-related services would deliver the greatest level of production benefit/cost.

5.5.2 Influence of shelterbelt species type on distance effects

Significant interactions detected through univariate analyses indicate that effects of distance from shelterbelts on functional composition of invertebrate communities depends on shelterbelt type. This may be due in part to differences in the capacity of each shelterbelt type to reduce wind speed or alter microclimate (Section 5.5.1), or to differing contrasts in habitat characteristics between each shelterbelt type and the open pasture. Vegetation structure in shelterbelts, particularly cover and diversity in lower strata, affects composition of invertebrate communities at fine scales, although previous studies suggest that these effects vary widely across different taxonomic groups (Chapter 4). Differences in shelterbelt vegetation (e.g. greater floral diversity) are likely to have driven the higher numbers of minor pollinators, tree pests, and generalist predators which were observed within and immediately adjacent to mixed native and (to a lesser extent) eucalypt shelterbelts, compared to pines. Pine shelterbelts did however support higher numbers of minor field pests (mostly aphids) in autumn. Although in some cases these differences extended to 0.5 – 1.5 TH, differences in abundance between shelterbelt types generally did not extend into open pasture. These findings suggest that while the choice of shelterbelt tree species may affect the functional composition of invertebrate communities at very fine scales (within shelterbelts and at their edges), shelterbelt type does not affect composition of communities across the broader paddock.

This study was undertaken at sites in the Tasmanian Midlands, a low complexity agricultural landscape (Section 5.3.1). The impact of factors such as shelterbelt habitat suitability and microclimate on invertebrate distribution are likely to vary across gradients of landscape complexity (Chaplin-Kramer et al., 2013). As such, there would be value in replicating this study in landscapes of differing complexity to expand our understanding of how drivers of distribution interact across scales.

5.5.3 Consequences for ecosystem services and shelterbelt design

Although direct measurement of ecosystem service provision was beyond the scope of this study, results relating to the distribution of invertebrate functional groups have potential implications for fine-scale provision of ecosystem services such as pollination and pest control. Relationships between abundance (or density) of organisms and provision of ecosystem services are generally considered to be linear (e.g. Dedej & Delaplane, 2003; Gaston et al., 2018). However, the presence of an invertebrate pollinator, pest, or pest predator does not necessarily guarantee ecosystem service provision. Pollination, for example, also depends on the presence of the target crop plant and the behaviour of the pollinator (foraging frequency and intensity), which may be affected by factors such as the availability of nesting sites and alternative pollen/nectar resources in the landscape (Olsson et al., 2015). In this study, the abundance of invertebrates in each functional group is used as an imperfect proxy for potential ecosystem service provision, and consequences are discussed in the context of generalised ‘mixed farming’ systems. There is scope for future studies to expand on our findings through direct measurement of ecosystem service provision (e.g. suppression of specific pest species, or flower visitation) or associated benefits (e.g. fruit set, quality, or yield).

Results relating to distribution of major pollinators suggest that shelterbelts deliver potential pollination benefits primarily through wind speed reduction, with benefits concentrated in the zone of greatest reduction (1 – 7 TH). Shelterbelt tree species selection could affect potential for pollination in two ways: porosity of different shelterbelt types will affect wind speed reduction (Chapter 3) which could in turn affect pollinator abundance, and vegetation characteristics could affect habitat suitability for minor (mostly native) pollinators. Although effects of shelterbelt type on abundance of minor pollinators did not extend into open pasture, the capacity of mixed native and eucalypt shelterbelts to support higher numbers of these pollinators may confer benefits such as supplementary pollination at edges, or improved resilience of the farming system to risks associated with loss of major pollinators (e.g. colony collapse disorder in *Apis mellifera*) (Gill et al., 2016).

Results from this study suggest that shelterbelts do not significantly affect potential for pest damage in adjacent pastures, which supports the findings of previous related studies (e.g. Dix et al., 1997). While higher numbers of minor field pests were observed within and at the edges of pine and eucalypt shelterbelts in some seasons, these differences did not extend into open pasture. Although mixed native shelterbelts were found to support higher numbers of tree pests compared to pines and eucalypts, these invertebrates were observed in very low numbers throughout the study. While tree pests are generally not considered to confer an economic cost in mixed farming systems, design of shelterbelt agroforestry systems which contain both mixed native species and commercial timber species should consider potential for increased tree pest loads in areas immediately adjacent to mixed native species stands.

Observed relationships between distance from shelterbelts and abundance of predators were generally weak and inconsistent, suggesting that shelterbelts have a limited influence on potential for control of pests by insect predators in adjacent pastures. However, our results do suggest that pest control may be aided slightly by shelterbelts through a combination of wind speed reduction and habitat provision, with mixed native shelterbelts offering greater potential benefits compared to pines and eucalypts. Predator-prey interactions in agroecosystems are complex, and more targeted species-specific studies across a range of systems (e.g. arable, horticulture) may be required to uncover further effects of shelterbelts (and shelterbelt condition) on abundance of key insect predators and their prey (e.g. Dong et al., 2015). Addressing the scale limitation (i.e. designing future experiments so that they encompass areas unaffected by shelterbelts) would also assist in detecting these effects.

5.6 Conclusions

Overall, findings from this study suggest that shelterbelts have potential to confer productivity benefits by enhancing provision of invertebrate-related ecosystem services such as pollination in adjacent paddocks. Results also show that shelterbelt condition (i.e. tree species selection) has a limited influence on provision of services, indicating that the microclimatic effects of shelterbelts (e.g. wind speed reduction) may be the dominant drivers of invertebrate-related benefits. Shelterbelts also provide habitat for decomposers and pests, although distribution of these groups across adjacent pasture was not found to be affected by shelterbelts. Paddock-scale pollination benefits may be increased if effective wind speed reduction is achieved, with potentially greater benefits delivered by shelterbelts with higher diversity and complexity in their vegetation (i.e. mixed native species shelterbelts). Mixed native species shelterbelts may also confer minor advantages over pines and eucalypts in terms of reducing potential for damage to crops/pasture by pests (through support of generalist predators).

These findings suggest that there are synergies between some ecosystem services provided by shelterbelts (i.e. shelter and pollination), and trade-offs between others (i.e. timber production and pest control). Knowledge of these trade-offs can assist farmers in designing shelterbelt systems to achieve specific objectives. Trade-offs relating to biodiversity conservation are discussed in a companion paper (Chapter 4), which uses a subset of this data to explore effects of shelterbelt vegetation characteristics on local invertebrate community composition at the order or higher taxon level.

These findings strengthen our understanding of the effects of shelterbelts, and shelterbelt condition, on invertebrate distribution and ecosystem service provision at fine scales. Results from this study can be used to increase the accuracy of ecosystem service models, thereby improving our ability to communicate the relative costs and benefits of shelterbelt agroforestry systems at a range of scales. Findings from this study can also assist farmers in selecting shelterbelt tree species which best serve their objectives.

Chapter 6: Quantifying farmer preferences for agroforestry design attributes and ecosystem services

This chapter is in preparation as:

Marais, Z.E., Tinch, D., Tocock, M., Baker, T.P., and Hunt, M.A. (in preparation). Quantifying farmer preferences for agroforestry design attributes and ecosystem services.

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6.1 Abstract

There is growing international interest in the use of agroforestry to address production and environmental challenges in temperate agricultural landscapes by enhancing provision of ecosystem services. An agroforestry system's capacity to provide ecosystem services at various scales depends on its design. Knowledge of farmer preferences for agroforestry design, and how these relate to potential demand for benefits delivered by ecosystem services, is crucial in developing effective strategies to maximise agroforestry adoption and improve decision-making. This study explores farmer preferences for agroforestry design using data from a discrete choice experiment undertaken in temperate agricultural regions of Australia. Marginal willingness to pay estimates were obtained for: species composition, configuration, and extent of tree cover. To test the hypothesis that agroforestry design preferences will align with demand for ecosystem services, participants were also asked to rank the relative importance of a range of ecosystem services. Results revealed strong preferences for lower-cost agroforestry alternatives, mixed native species compositions over single species pine, and shelterbelt/woodlot configurations over paddock trees. Design preferences were found to align with demand for ecosystem services, with biodiversity and shelter-related services ranked most highly. These results can inform development and application of extension and policy efforts aimed at increasing agroforestry adoption within the study area and beyond. Findings also suggest a need for further research to substantiate links between agroforestry design attributes and provision of ecosystem services.

6.2 Introduction

Agroforestry systems enhance the provision of ecosystem services that deliver a range of private and public benefits at multiple scales. For example, agroforestry systems regulate microclimate to improve agricultural productivity outcomes at the farm scale (Baker et al., in press; Baker et al., 2021b), while also contributing to climate change mitigation, biodiversity conservation, and salinity management at the landscape scale (George et al., 2012; Townsend et al., 2012). Despite growing evidence of the benefits of agroforestry systems (Asbjornsen et al., 2014; Jose, 2009; Smith et al., 2012), adoption of agroforestry remains constrained in some parts of the world including Australia (Stewart, 2009). Global recognition of the importance of agroforestry for future multi-functional agricultural landscapes has led to numerous initiatives aimed at encouraging adoption of agroforestry by farmers to increase delivery of associated private and public benefits e.g. (Place et al., 2013). Although many existing initiatives focus on tropical systems, there is growing interest in the expansion of agroforestry in temperate agricultural areas, particularly in landscapes where historical or ongoing deforestation threatens key ecosystem processes (Santiago-Freijanes et al., 2018).

Interest in the expansion of temperate agroforestry - as a means by which to establish forest resources, improve farm productivity, and enhance ecosystem service provision - extends to Australia. Factors affecting adoption of farm afforestation and restoration in Australia are largely understood (Pannell et al., 2006; Schirmer & Bull, 2014). In the case of agroforestry, studies on adoption have focused on the analysis of economic viability (Donaghy et al., 2010; Flugge & Abadi, 2006), social drivers (Vanclay, 2004), policy impediments (Race & Curtis, 2007), and more recently, the values and social norms that influence perceptions of agroforestry (Fleming et al., 2019). Although these studies provide useful insight, further strengthening our understanding of agroforestry demand will assist in the development of effective strategies to increase agroforestry adoption. Specifically, gaps remain in our understanding of how potential adopters are likely to approach decisions relating to agroforestry design. The aim of this study is to explore farmer preferences for agroforestry design and ecosystem services in temperate regions, using data from a discrete choice experiment (DCE) undertaken in Australia.

There are many different ways in which trees can be managed alongside other forms of agriculture to create agroforestry systems. The tree component of an agroforestry system will commonly take the form of shelterbelts, trees in riparian zones, integrated remnant vegetation, woodlots, widely-spaced individual trees (paddock trees), or tree rows within pasture or crops. These configurations may be further varied by the choice of tree species, management techniques, or the extent of tree cover as a proportion of the property area. Different agroforestry designs will vary in cost (establishment, maintenance, and opportunity costs) and will offer different types and amounts of ecosystem services at various scales. For example, paddock trees may provide shade for livestock at the farm scale, whereas other configurations may provide better outcomes for management of local flood risk (Lunka & Patil, 2016). Single species compositions planted and managed using forestry techniques may provide opportunities for wood production, whereas mixed species compositions may be better for supporting biodiversity (Chapter 3). The extent of tree cover can also influence levels of ecosystem service provision (Townsend et al., 2012), and the costs associated with different design options may affect their perceived relative advantage (Pannell et al., 2006).

Discrete choice experiments are used to test how the preferences of farmers vary when they are presented with different agroforestry alternatives (Lamour & Subervie, 2020; Permadi et al., 2017). This enables analysis of preferences for individual attributes of agroforestry systems, as well as estimation of the value of attributes for which real market data are generally not available. Agroforestry alternatives may be differentiated by attributes such as the configuration and species composition of agroforestry plantings, extent of tree cover, requirements for labour or expertise, policy or extension structures, or total cost. By quantifying the relative importance and value of specific decision-making criteria, DCEs inform a predictive understanding of agroforestry demand and decision-making (Mercer & Snook, 2004). This understanding is critical to the effective design and implementation of strategies to increase potential for agroforestry adoption.

An example of a strategy that could encourage adoption is the application of ecosystem service valuation to demonstrate potential farm-scale benefits of agroforestry (England et al., 2020; Marais et al., 2019; Quandt et al., 2019). This work assumes that farmers will seek to design agroforestry systems that maximise provision of specific ecosystem services that align with their enterprise objectives. Using available evidence linking design attributes with provision of specific ecosystem services, the strength of this premise can be explored by examining farmer preferences for agroforestry design attributes alongside expectations for ecosystem service delivery.

In the context of agroforestry systems common in temperate agricultural areas of Australia, we use a DCE to explore the following research questions:

1. What are farmer preferences and values for key agroforestry system design attributes (species composition, configuration, extent, and cost)?
2. Do preferences for agroforestry design attributes align with demand for delivery of key ecosystem services?

Finally, we discuss how our findings might inform strategies for increasing adoption of agroforestry in temperate regions.

6.3 Methods

6.3.1 Discrete choice experiments

Discrete choice experiments (DCEs) are one of a suite of stated preference techniques that can be used to elicit an individual's preferences relating to a particular good or service. Drawing on theories of value (Lancaster, 1966) and random utility (Luce, 1959; McFadden, 1974), DCEs assume that individuals derive utility from the different attributes of the good rather than the good itself (Lancaster, 1966). As such, they can be used to estimate the relative importance and value of different attributes of the good or service, e.g. the relative importance to farmers of different attributes of agroforestry systems.

Discrete choice experiments are typically delivered as a survey in which respondents are presented with scenarios or 'choice sets' comprised of two or more alternatives. The good or service that is the subject of the experiment is treated as a set of different attributes with a restricted set of variations (levels) for each attribute. Attribute levels are varied across each alternative, and respondents are asked to choose between the alternatives in each choice set. These choices can then be analysed to determine the relative utility of each attribute. If cost is included as an attribute, marginal willingness to pay (WTP) for changes to particular attributes can also be calculated.

6.3.2 Data collection

Data for the DCE were collected using an online survey, which was designed and distributed via the 'surveymonkey' platform. Ethics approval for the survey was granted by the Tasmanian Social Science Human Research Ethics Committee (H0018228). All respondents were required to answer informed consent questions, screening, and introductory questions before commencing a series of 15 choice sets (Section 6.3.3). The choice sets were followed by a series of questions about reasons for choices, the importance of ecosystem services and disservices, and previous experience with agroforestry. The survey was piloted in December 2019 and sample responses were collected between March and August 2020. Results from the pilot were used to improve the efficiency of the design (see Section 6.3.3) but were not included in data analysis.

The purpose of this study was to explore farmer preferences for agroforestry system design attributes in the context of temperate Australian agroforestry systems. Although initially focused on the State of Tasmania, sampling was extended to other States with similar conditions and potential for temperate agroforestry expansion. Responses were received from farmers in Victoria, Tasmania, New South Wales, and Western Australia. Respondents were required to be 18 years old or older, and to identify as a farmer (owner or manager).

The survey was distributed via email to farmers within the researchers' networks, and via e-newsletters and social media to the memberships of regional farming groups, agricultural industry research and development groups, and State government agriculture bodies. In Tasmania, a radio interview was also conducted with the researchers to further advertise the survey. As an incentive for participation, a link to a prize draw for a gift voucher was provided upon completion of the survey.

6.3.3 Study design

Choice sets

Each respondent was presented with an identical series of 15 choice sets. NGENE 1.01.02 (ChoiceMetrics, 2014) was used to generate a Bayesian efficient design for a multinomial logit (MNL) model using parameter estimates from the pilot study (D-error = 0.02423) (Scarpa & Rose, 2008). As farmers, respondents were familiar with the choice attributes, which removed the need for large amounts of upfront information in the survey and in turn allowed for the relatively high number of choice sets in the design (15). The average time taken to complete the survey was 18 minutes and six seconds.

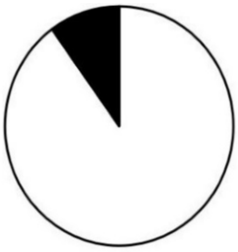
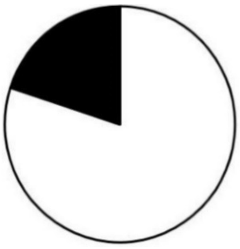
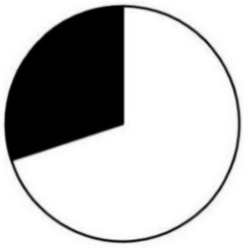
Choice sets were comprised of four alternatives consisting of a combination of four attributes: species composition, configuration, extent, and cost (Table 6.1). Attributes and levels were specified in consultation with agroforestry practitioners and researchers to ensure that they represented realistic hypothetical scenarios for agroforestry design in temperate Australia.

Table 6.1: Agroforestry design attributes and attribute levels used to generate the 15 choice sets (including those for the 'do nothing' alternative)

Attribute	**Levels**	**'Do nothing' alternative**
Tree species composition	*Pinus sp.* (single species) (1) *Eucalyptus sp.* (single species) (2) Mixed native species (multiple species) (3)	No trees
Configuration of trees	Shelterbelt (1) Woodlot (2) Paddock trees (3)	No trees
Extent of trees	10%, 20%, or 30% of 10 ha block	0%
Total cost per hectare ($)	10,400, 10,800, 11,200, 11,600, or 12,000	10,000

Respondents were presented with a hypothetical scenario in which they were asked to decide which of four alternative 10 ha blocks of land they would purchase following compulsory acquisition of the equivalent area of their own property, with compensation of $12,000 per hectare. Respondents were required to select only one alternative in each choice set. Each choice set included a zero-cost 'do-nothing' alternative as one of the four alternatives, presented as '*Option D: I would purchase land without trees at $10,000 per hectare rather than any of these options*'. Farmers were therefore faced with the decision to trade up to a total of $20,000 of their compensation (the 'do-nothing' scenario) for agroforestry assets consisting of different attributes, at different costs. Images depicting levels for each attribute were displayed alongside descriptive text. An example choice set is shown in Table 6.2.

Table 6.2: An example of one of the 15 choice sets used in the survey. Options could contain any combination of levels for configuration, species composition, extent, and cost as described in Table 6.1.

	Option A	**Option B**	**Option C**
Configuration	Shelterbelt 	Woodlot 	Paddock trees 
Species composition	Pine (single species) 	Eucalypt (single species) 	Mixed native (multiple species) 
Extent (of 10ha block)	 10%	 20%	 30%
Total cost per hectare ($10,000 plus cost of trees)	**$$** $10,800	**$** $10,400	**$$$** $11,200

Please select one:

(a) Option A
(b) Option B
(c) Option C
(d) Option D: I would purchase land without trees at $10,000 per hectare rather than any of these options

Follow-up questions

Following completion of the choice sets, respondents were asked whether they selected the 'do nothing' scenario (Option D) in any of the 15 choice sets (yes or no). If they answered 'yes', skip logic directed them to a question which asked them to select reasons for having made this choice. Respondents were also asked to select which attribute (configuration, species composition, extent, or cost) they perceived as being the 'most important' in their decision-making (compulsory question), as well as any that they considered 'not important' (optional question).

Towards the end of the survey, respondents were asked to rank ecosystem services and disservices according to how important they were to them in making decisions about trees on their farms (see Table 6.8 for full list of services and disservices). Each service or disservice was ranked individually on the following weighted scale: 'very important' (2), 'quite important' (1), 'neither important nor unimportant' (0), or 'unimportant' (-1). Weighted averages were calculated from these rankings using the following equation, where *w* represents the weight of the answer choice and *x* represents the response count for the answer choice:

$$weighted\ average = \frac{x_1w_1 + x_2w_2 + x_3w_3 \dots x_nw_n}{Total\ number\ of\ responses}$$

The final questions in the survey related to the respondents' previous experience with agroforestry. Respondents were first asked whether they had ever planted trees on their farm (yes or no), and whether they currently have trees on their farm (yes or no). If they answered 'yes' to the latter, skip logic directed them to a question about the extent of their farm currently covered by trees, the configuration of trees on their farm, whether any of their trees were suitable for production of commercial timber, whether any of the trees existed primarily to support biodiversity, and lastly whether any of the trees were funded by external parties.

6.3.4 Data analysis

Model specification

A random parameters logit (RPL) model specification was employed to analyse results of the choice experiment in this study. Unlike more restrictive model specifications such as the multinomial logit (MNL) specification (Box 1), the RPL specification accounts for individual preference heterogeneity by allowing parameters of explanatory variables to vary randomly across individuals within the sample rather than being fixed.

In the RPL specification, for each choice set *s* that they face, the utility *U* received by respondent *n* from selecting alternative *j* can be expressed as:

$$U_{nsj} = \beta_n{}' x_{nsj} + \varepsilon_{nsj} \quad \text{where } V_{nsj} = \beta_n{}' x_{nsj} \ \text{ and } \beta_n = \beta + \tau v_n$$

Unobserved heterogeneity is represented by τv_n, and β is the coefficient vector that is unobserved for each respondent (Revelt & Train, 1998). The probability of respondent *n* selecting alternative *j* can therefore be expressed as:

$$\text{Prob}(choice_n = j) = \frac{\exp\,(V_{nsj})}{\sum_{j=1}^{Jn} \exp\,(V_{nsj})}$$

To capture potential unobserved random effects, an error component (EC) parameter was added to the RPL model, creating an RPL-EC model. Incorporation of an EC parameter is one means of addressing biases associated with 'status quo' alternatives which are prevalent in DCEs with environmental applications (Scarpa et al., 2005). Distribution simulations for the RPL-EC model specification were based on 2500 draws. Draws were obtained using the Modified Latin Hypercube Sampling technique (Hess et al., 2006).

All parameters were specified as independently, normally distributed. Species composition and configuration variables were effects-coded (one level was chosen as the base level and two effects-coded variables were created for the other two levels). To explore potential non-linearity in the response of utility to increasing extent of trees as a proportion of property area, a quadratic specification for extent ($extent^2$) was included in the RPL-EC model.

All models presented in this study were estimated using PythonBiogeme 2.6a (Bierlaire, 2016), with code adapted from Rose and Zhang (2017).

Willingness to pay (WTP)

Once a model is estimated with coefficients in the utility, i.e. in 'preference space', WTP may be calculated by dividing the utility coefficient for a given attribute by the inverse cost coefficient. Alternatively, the model may be re-specified so that for each attribute the parameters are the marginal WTP rather than the utility coefficient (known as 'WTP space').

Estimation directly in WTP space can potentially improve the accuracy of the model, as it avoids occurrence of over-inflation of WTP values and allows for greater control over the distributions of WTP (Scarpa et al., 2008). For this reason, in addition to estimating utility coefficients in preference space, WTP for each non-monetary attribute in this study was estimated directly in WTP space. Although WTP values do not relate to a specific land area, given the choice scenario involved a hypothetical 10 ha land purchase these values are indicative of farmers' relative WTP for each attribute over 10 ha of land. Results are presented on a per-hectare basis to facilitate interpretation of the results.

Box 1: Multinomial logit and random parameters logit model specifications

The standard model for analysis of choice experiments is the multinomial logit (MNL) specification. In the MNL specification, error terms are assumed to be independently and identically distributed so that the probabilities (P_{nsj}) of alternative j being preferred over alternative i can be expressed as:

$$P(Y_i = 1, V_{ns}) = Prob(\varepsilon_{nsj} - \varepsilon_{nsi} > V_{nsj} - V_{nsi}, \forall i \neq j)$$

$$P_{nsj} = \frac{exp^{V(nsj)}}{\sum_j exp^{V(nsj)}}, j = 1, \ldots . J$$

The assumption of independent and identically distributed error terms in the MNL specification results in a property of independence from irrelevant alternatives (IIA). This stipulates that the relative probabilities of two alternatives being chosen are unaffected by the presence or absence of additional alternatives (Luce, 1959). Although the MNL specification is a common starting point for choice experiment analysis, the IIA assumption rarely holds, particularly if alternatives are similar. If the IIA assumption is violated, alternative models which allow for relaxation of the IIA assumption may be used.

The random parameters logit (RPL) model specification, also known as mixed multinomial logit (MMNL), is one such alternative to the MNL specification which allows for relaxation of the IIA assumption. The RPL specification accounts for induvial preference heterogeneity by allowing parameters of explanatory variables to vary randomly across individuals within the sample rather than being fixed.

Initial analysis for this study was undertaken using the MNL specification. However, it was found that the assumption of IIA was violated. To accommodate individual preference heterogeneity across the sample, alternative models which allow for relaxation of the IIA assumption were explored. Of the alternative models investigated, the RPL model was found to be the most appropriate for the analysis undertaken in this study.

6.4 Results

6.4.1 Sample information

A total of 76 complete responses were received. Given the relatively high number of choice cards completed by each individual in this study (15), this sample size resulted in a total number of observations (N = 1140) comparable to other DCEs targeting farmers (e.g. Schulz et al., 2014) and met requirements for statistical validity of the model, as tested in the pilot study. Compared to characteristics of the national (Australian) population and agricultural industry (Table 6.3), the sample was broadly representative although it did differ in some respects potentially related to inherit differences in temperate agricultural systems.

The majority of responses were received from Tasmania (33), New South Wales (23), and Victoria (15), with only 5 responses received from Western Australia. Tasmanian farmers may be overrepresented in the sample (Table 6.3) due to the researchers being based in Tasmania and having greater success in distributing the survey locally. Tasmania, Victoria, New South Wales, and Western Australia have the largest areas of commercial plantations in Australia (Montreal Process Implementation Group, 2018) and have historically had the largest areas of farm forestry plantations (URS Forestry, 2008), although recent data on the latter is scarce. As such, these States represent key areas of potential temperate agroforestry adoption and the geographic distribution of respondents was therefore considered appropriate for the purpose of this study.

Although the median age of respondents (57.5 years) was higher than the national median age (38 years), it was representative of the 2018-19 average age of farmers in Australia (58 years) (Australian Bureau of Statistics, 2020b). The majority of respondents indicated that their farming enterprise was either ‘mixed livestock/cropping’ (47.4%) or ‘livestock’ (35.5%), which is largely representative of Australian agriculture (Table 6.3). Mean farm property size in the sample was 2006 ha, which was roughly half the 2018-19 national average (4331ha) and significantly less than that of Western Australia (9633 ha), but larger than average property sizes in Tasmania (588 ha), Victoria (524 ha), and New South Wales (1938 ha) (Australian Bureau of Statistics, 2020a).

Table 6.3: Sample characteristics compared to national (Australian) population and agriculture industry characteristics (Australian Bureau of Statistics, 2020a, 2020b, 2020c, 2020d)

	Sample	Australia
State		*Proportion of total national employment in agriculture, forestry, or fishing (2018-2019):*
Tasmania	43.4%	4.7%
New South Wales	31.6%	26.8%
Victoria	19.7%	22.2%
Western Australia	5.3%	12.2%
Median age	57.5	38 (2019)
Age group		*Proportion of total population (2019):*
18-29 years	3.9%	16.9%
30-44 years	22.4%	21.1%
45-59 years	28.9%	19.7%
60+ years	44.7%	19.6%
Mean farm size	2006 ha	4291 ha (2018-2019)
Farm size		Data not available
1-50 ha	14.5%	
51-200 ha	17.1%	
201-1000 ha	34.2%	
1001-5000 ha	27.6%	
5001-20000 ha	6.6%	
Enterprise type		*Proportion of total number of agricultural businesses (2018-2019):*
Mixed livestock/cropping	47.4%	Data not available
Livestock	35.5%	Sheep/lamb: 35.9% Cattle: 54.0% Other livestock: 14.5%
Dairy	1.3%	6.4%
Cropping (excl. horticulture)	4.0%	45.0%
Horticulture (excl. grapes)	5.3%	12.6%
Viticulture	4.0%	4.1%
Other	2.6%	Data not available

Seventy-four of the seventy-six respondents (97.4%) stated that they had previously planted trees on their farm and all respondents stated that they had trees on their farm at the time of the survey. 61.8% indicated that the extent of tree cover on their farm was between 5% and 20% (Figure 6.1). Fewer than half of the respondents (36.8%) indicated that the trees on their farm were suitable for production of commercial timber, and 68.4% indicated that at least some of the trees on their farm exist primarily to support biodiversity. 64.5% of respondents indicated that they had received some funding from external parties to plant the trees on their farm. All options of tree configurations were commonly present on farms, although most respondents indicated that their farms contained either mixed native species shelterbelts, paddock trees, or remnant patches (Figure 6.2).

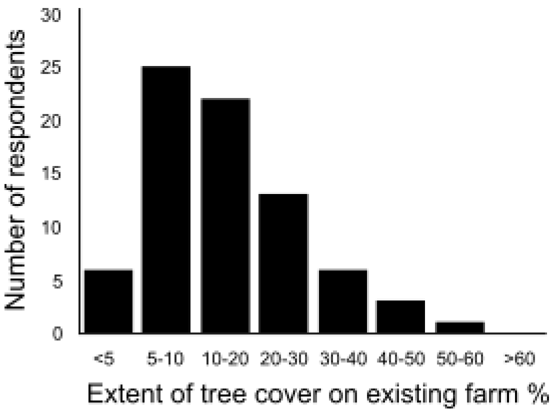


Figure 6.1: Number of respondents who selected each available category for extent of existing tree cover as a percentage of the total area of their farm (single selection)

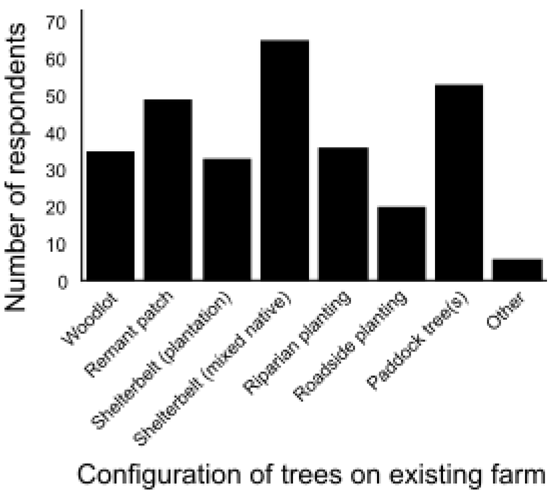


Figure 6.2: Number of respondents who selected each available option for configuration of existing trees on their farm (multiple selections allowed)

6.4.2 Choice experiment results

All attributes tested in the choice experiment (species composition, configuration, extent, and cost) were shown to significantly affect farmers' agroforestry design choices. There were significant differences between all levels of the effects-coded attributes (configuration and species composition), with the exception of there being no significant difference in the relative utility of woodlots compared to shelterbelts (Table 6.4). Willingness to pay (WTP) estimates showed that farmers were willing to pay significantly more for some attribute levels over others (e.g. mixed native species compositions over pines) (Table 6.5).

Table 6.4: Random parameters logit with error component (RPL-EC) results. Regression coefficients representing marginal utility (estimated in preference space) and willingness to pay (WTP) (estimated in WTP space) are presented alongside significance levels: ***$p < 0.01$, **$p < 0.05$, *$p < 0.10$, with standard error in parentheses. ASC refers to Alternative Specific Constant.

	Utility (preference space)		**WTP (WTP space)**	
Attribute	**Mean**	**Standard deviation**	**Mean**	**Standard deviation**
ASC (Option A)	2.29*** (0.80)		38.81*** (11.22)	
ASC (Option B)	2.14** (0.79)		36.87*** (10.78)	
ASC (Option C)	2.19** (0.82)		39.84*** (12.03)	
Configuration				
Paddock trees vs. shelterbelt	-0.81*** (0.28)	1.32*** (0.23)	-13.39*** (4.24)	32.52*** (10.75)
Woodlot vs. shelterbelt	-0.18 (0.23)	1.22*** (0.26)	-6.09 (4.48)	23.92** (8.70)
Species composition				
Mixed native vs. pine	2.06*** (0.32)	2.20*** (0.51)	50.65*** (17.19)	41.93*** (12.25)
Eucalypt vs. pine	0.47** (0.20)	0.73*** (0.23)	6.66** (2.54)	16.65** (6.52)
Cost	-3.08*** (0.33)	-0.97*** (0.17)	-2.62*** (0.28)	-0.88*** (0.25)
Extent	1.35** (0.53)	1.28*** (0.21)	24.00* (13.15)	28.88*** (8.56)
Extent^2	-0.45*** (0.13)		-7.90** (3.76)	
Error component	3.86*** (1.25)		69.73*** (21.00)	
Adjusted ρ^2	0.41		0.41	
Final log likelihood	-940.73		-940.67	

Table 6.5: Values representing marginal willingness to pay (WTP), on a per-hectare basis, for each non-monetary design attribute (coefficients estimated in WTP space, multiplied by 10 for correct interpretation) are presented with 95% confidence intervals (CI).

Attribute	**WTP (per hectare)**	**95% CI**
Configuration		
Paddock trees vs. shelterbelt	-\$133.87	-\$217.01 to -\$50.73
Woodlot vs. shelterbelt	-\$60.91	-\$148.66 to \$26.84
Species composition		
Mixed native vs. pine	\$506.50	\$169.67 to \$843.33
Eucalypt vs. pine	\$66.57	\$16.81 to \$116.33
Extent	\$239.99	-\$17.69 to \$497.67
Extent^2	-\$79.04	-\$152.72 to -\$5.36

The RPL-EC model resulted in significant standard deviations for all parameters ($p < 0.01$) (Table 6.4), indicating preference heterogeneity in the sample and confirming violation of the independence from irrelevant alternatives (IIA) assumption in the multinomial logit (MNL) model specification. The error component (EC) parameter was shown to be significant in the final RPL-EC model ($p < 0.01$) (Table 6.4). This suggests that the model estimation is strengthened by treatment of the EC as a separate parameter, possibly because it accounts for biases associated with the status quo or 'do-nothing' option i.e. effects on choices to select (or not select) the status quo that cannot be wholly explained by attribute values.

Regression coefficients in the 'Utility' column of Table 6.4 can be interpreted as marginal utility values, i.e. the additional satisfaction or benefit (utility) that a respondent derives from consuming an additional unit of the attribute. The coefficients on each of the Alternative Specific Constant (ASC) parameters represent the unobserved heterogeneity that leads respondents to be more likely to select the agroforestry options (A, B, or C), relative to the 'do-nothing' option (Option D). All three ASC coefficients are significant ($p \leq 0.01$) and positive, which can be interpreted as a preference for agroforestry alternatives over a scenario in which no trees are present on the land that is being purchased. Of a total of 76 respondents, 39 chose the 'do nothing' scenario (Option D) at least once, and the most frequently-selected reason for this choice was that 'available options for tree composition/configuration were not suitable' (Table 6.6).

Table 6.6: Reasons selected for choosing Option D ('do nothing' scenario) at least once in the 15 choice sets (multiple selections allowed)

Reason	Number of respondents
Available options for tree composition/configuration were not suitable for me	36
I thought that the other options were too expensive	6
Trees restrict my farming practices	4
I think that there are enough trees on my farm already	3
If I had money available, I would not choose to spend it on trees	1
I do not want to maintain trees and/or fences into the future	1
I prefer the look of land that does not contain trees	0
I do not think farmers should have to spend their own money on trees	0
I do not think that trees provide me with any benefits	0
Other	6
Total number of respondents who answered this question	39

Both configuration and species composition were shown to affect farmers' decisions relating to agroforestry design, with particularly strong preferences observed for species composition. The effects coding of these variables allows for comparison of relative utility between specific levels.

The paddock tree configuration was shown to have a significant negative impact on relative utility compared to the shelterbelt configuration ($p < 0.01$). Although results suggest that the woodlot configuration has a negative impact on relative utility compared to the shelterbelt configuration, the difference between these levels was not significant ($p = 0.42$). These results indicate that farmers prefer shelterbelts over paddock trees but have no significant difference in preference for shelterbelts or woodlots. Results for WTP indicate that farmers require compensation of \$133.87 (95% CI: \$217.01 to \$50.73) per hectare, for the paddock tree configuration compared to shelterbelts. Estimated WTP for the woodlot configuration compared to shelterbelts was non-significant, suggesting that farmers are indifferent regarding the choice between these two configurations.

The mixed native (multiple species) composition was shown to have a significant positive relative difference in utility compared to the pine (single species, softwood) composition ($p < 0.01$). The size of the coefficient for this parameter was also relatively high (Table 6.4), indicating that this difference in relative utility is large compared to other attributes. The eucalypt (single species, hardwood) composition was also shown to have a significant positive relative difference in utility compared to the pine composition ($p = 0.02$), although the size of the coefficient in this case was smaller. These results indicate that farmers strongly prefer mixed native species compositions over pine compositions and have a slight preference for eucalypt compositions over pine compositions. Results for WTP indicate that farmers are willing to pay \$506.50 (95% CI: \$169.67 to \$843.33) more, per hectare, for mixed native species compositions compared to pines but only \$66.57 (95% CI: \$16.81 to \$116.33) more for eucalypts compared to pines.

The coefficients for the extent and extent^2 parameters provide information about farmer preferences for varying extents of tree cover as a proportion of farm area. In preference space, the coefficient for extent is positive and significant ($p = 0.01$), whereas the coefficient for the quadratic specification, extent^2, is negative and significant ($p < 0.01$) (Table 6.4). These results indicate that within the range of tree cover extents included in this study (0-30%), larger extents have higher utility until a certain point, beyond which utility declines. The preference for agroforestry alternatives involving tree cover of 10% or greater, relative to the 'do-nothing' scenario of 0% cover, suggests that the turning point for the utility of tree cover extent is greater than 10%. In other words, farmers prefer tree cover extents greater than 10%, but an optimal extent of tree cover may exist somewhere between 10-30%. Estimates of WTP for the extent parameter were positive but not highly significant ($p = 0.07$) and the 95% CI (-\$17.69 to \$497.67) overlapped with \$0, indicating that some farmers are not willing to pay to increase tree cover extent by 10%. The significant negative coefficient for extent^2 when estimated in WTP space supports the theory that there is an optimal extent of tree cover for farmers.

The coefficient on the cost parameter is negative and significant ($p < 0.01$), indicating that cost significantly influences farmers' decisions relating to agroforestry design and that lower costs have higher utility. The size of the coefficient on the mean cost parameter (-3.08) is larger than those of all other attributes (Table 6.4), suggesting that cost has the strongest influence on decision-making of the attributes tested in this study.

As outlined in Section 6.3.3, respondents were asked to select which attributes they perceived as being the 'most important' as well as attributes that were 'not important' in their decision-making, after having completed the choice sets. Species composition was selected by 48 out of the 76 respondents as being the 'most important' attribute, and cost was chosen most frequently as being 'not important' (Table 6.7). Although some of the results from these direct questions align with the choice experiment results (e.g. the relatively strong preferences for species composition), others are contradictory (e.g. the low influence of cost).

Table 6.7: Attributes perceived to be 'most important' (single selection) and 'not important' (multiple selections allowed) by respondents. Numbers of respondent who selected each option are shown.

Attribute	**'Most important' (compulsory question)**	**'Not important' (optional question)**
Configuration	17	2
Species composition	48	2
Extent	9	8
Cost	2	45
Total number of respondents	76	55

6.4.3 Ecosystem service ranking

The results of the ecosystem service ranking questions (Section 6.3.3) show that the top five most important ecosystem services were ‘habitat for beneficial insects’, ‘shelter (or shade) for livestock’, ‘visual amenity’, ‘wind erosion control’, and ‘habitat for native wildlife’ (Table 6.8). ‘Fire risk’ achieved the highest weighted average ranking out of the disservices listed (0.74). When asked to list any services or disservices not included that were also of importance, respondent answers included: screening from noise and pollution, contamination of produce by leaves/branches, non-timber forest products (e.g. fruit and nuts), firewood, and salinity control. There was variation in individual preferences for all ecosystem services and disservices (i.e. rankings ranged from unimportant to very important), with the exception of ‘habitat for beneficial insects’ which was not regarded as ‘unimportant’ by any respondents.

Table 6.8: Weighted averages for ranking of ecosystem services and disservices (-1 = unimportant, 0 = neither important nor unimportant, 1 = important, 2 = very important)

Ecosystem service	**Weighted average ranking**
Habitat for beneficial insects (e.g. pollinators)	1.57
Shelter (or shade) for livestock	1.49
Visual amenity	1.28
Wind erosion control	1.21
Habitat for native wildlife	1.20
Surface water erosion control	1.14
Carbon sequestration (storage)	0.97
Nutrient/sediment pollution control	0.93
Groundwater regulation	0.88
Recreation (e.g. bird watching, walking, hunting)	0.57
Shelter for crops	0.51
On-farm air quality improvement	0.51
Reduction of spray drift	0.43
Timber production	0.07
Ecosystem disservice	
Fire risk	0.74
Competition with adjacent crops/pasture (e.g. shading)	0.65
Habitat for pests	0.63
Source of weeds	0.54

6.5 Discussion

Using a discrete choice experiment (DCE), an overall positive preference was shown for agroforestry alternatives over no-agroforestry scenarios, and agroforestry design attributes (cost, species composition, configuration, and extent) were shown to significantly influence farmer decisions. When comparing agroforestry design attributes, cost and species composition were shown to generally have greater influence on decision-making than configuration and extent. Farmers indicated that they would be willing to pay significantly more for mixed native species and eucalypt compositions compared to pines, as well as for shelterbelt or woodlot configurations compared to paddock trees. We discuss findings from the DCE in relation to sample characteristics (Table 6.3) and demand for ecosystem services. We then consider implications for temperate agroforestry extension and adoption, within the study area and more broadly.

6.5.1 Agroforestry design preferences

Results of the DCE showed that cost had the greatest influence of all the attributes tested (Table 6.4). Cost is generally expected to influence, if not constrain, adoption of innovations in agriculture, as it affects the perceived advantage of the innovation relative to the status quo (Pannell et al., 2006). In the case of agroforestry, this expectation is supported by previous studies which have shown that adoption may be influenced by perceived economic viability (Fleming et al., 2019). However, as demonstrated by Fleming et al. (2019), perceptions of agroforestry are diverse and relate to underlying value systems, including those more concerned with the non-economic values of trees. In some cases, farmers may view the cost of agroforestry as insignificant when considering the value returned. When asked directly, most respondents in this study stated that cost was ‘not important’ in their decision-making (Table 6.7). Although this finding contradicts results from the DCE (which highlighted the importance of cost), this may be attributed to different perspectives on ‘importance’: although some farmers may not consider cost to be ‘important’ in their decision-making, cost may still influence their choices once the initial decision to spend money instead of ‘do nothing’ is made. In other words, we would still expect farmers to seek value for money when choosing between agroforestry alternatives, even if other design attributes were considered more ‘important’ than cost. The size and significance of the coefficient for cost in the DCE results demonstrates underlying preferences for lower-cost or ‘best value for money’ alternatives.

Sixty-three percent of respondents selected species composition as being the most important factor in their decision-making (Table 6.7), which was supported by DCE results showing strong preferences and high WTP for certain species compositions over others (Table 6.4). These results suggest that species composition is likely to be an important factor in farmer decision-making relating to agroforestry design. This is consistent with Schirmer and Bull (2014), who found that Australian landholders’ overall willingness to adopt afforestation was significantly related to design preferences, including a preference for planting native species. The three species composition types tested in this study have the potential to deliver different types and amounts of ecosystem services and benefits to farmers (Chapter 3). Observed preferences for certain species compositions may therefore relate to demand for specific ecosystem services such as biodiversity, timber production, shelter, and carbon sequestration (see Section 6.5.2). The strong preference for the mixed native species composition may also relate to familiarity or intrinsic preference, as most respondents indicated that they had mixed native species plantings on their farms already (Figure 6.2).

Extent of tree cover was shown to have a moderate level of influence on decision-making (Table 6.4), with results indicating that farmers may prefer larger extents over smaller extents within a reasonable range. This finding is broadly consistent with that of Polyakov et al. (2015), who used hedonic modelling to determine that native vegetation cover of 20-40% was optimal for maximising per-hectare land value of rural properties in central-northern Victoria. The preference for larger extents (up to a certain point) suggests that farmers may see value in increasing tree cover on their properties, although WTP estimates indicate that some farmers are not willing to pay more for this increase (Table 6.5). The existence of a threshold for extent may stem from consideration of productivity trade-offs, as farmers are likely to prefer to retain a certain proportion of arable or grazing land in order to maintain the economic viability of the farm, particularly if the agroforestry plantings do not contain harvestable products. The threshold of preferences for extent may also vary depending on farm property size (Polyakov et al., 2015), as larger properties may have greater flexibility and capacity to designate areas for agroforestry plantings, partly because they often have higher proportions of marginal land. The preference for increasing extent of tree cover observed in the DCE may therefore be partly attributed to the average farm size in the sample being larger than relevant State averages. Preferences relating to extent could also be influenced by previous experience, as most respondents indicated that existing extent of tree cover on their property was close to 20% (Figure 6.1).

Preferences relating to configuration were generally not as consistent across the sample as those relating to cost, species composition, or extent (Table 6.4). However, when asked directly, 22% of respondents stated that configuration was the most important factor in their decision-making (Table 6.7). Preferences for certain configurations could be influenced by the type of farming enterprise and the size or layout of the property, as different configurations may confer advantages or disadvantages in terms of access and farming practice. For example, continuous woodlots may be preferred in broadacre systems due the practical challenge of navigating machinery around paddock trees or shelterbelts, whereas integrated shelterbelts or paddock trees may be preferred in rotational grazing systems to provide shelter across multiple paddocks. Although the prevalence of use of centre pivot irrigation among respondents was not known, recent rapid uptake of these systems may have influenced configuration preferences as the presence of paddock trees restricts movement of pivot booms. Preferences for configuration may also relate to demand for ecosystem services, particularly biodiversity and the provision of shade or shelter for livestock or crops (see Section 6.5.2).

6.5.2 Relationship between design preferences and ecosystem service rankings

Biodiversity-related ecosystem services were ranked highly for importance, with provision of habitat for beneficial insects and habitat for native wildlife within the top five highest-ranked services. Two thirds of respondents indicated that some of the trees on their farm exist primarily to support biodiversity, suggesting that existing practices reflect the perceived importance of these services. The high ranking of biodiversity-related services may relate to a desire to restore native flora and fauna in agricultural areas. However, the top ranking of habitat for beneficial insects suggests that farmers may be more interested in biodiversity as it relates to productivity, for example through pollination or integrated pest management.

Compared to single species, mixed native species compositions are likely to offer higher-quality habitat for wildlife and beneficial insects due to their greater structural complexity and floristic diversity (Carr et al., 2000; McElhinny et al., 2006; Saunders & Luck, 2018). In addition to composition, the total amount (extent) of habitat within a fragmented landscape has also been shown to correlate positively with biodiversity outcomes (Gardiner et al., 2018). The strong preferences exhibited for the mixed native species composition, and for increasing extent of tree cover, suggest alignment between farmer design choices and the perceived importance of biodiversity-related ecosystem services.

In the case of configuration, alignment of design preferences with ecosystem service demand is less clear. Island biogeography principles may suggest that agroforestry configurations such as woodlots, which provide larger or more continuous patches of vegetation compared to isolated paddock trees, would be more effective in supporting biodiversity (MacArthur & Wilson, 1967). However, evidence suggests that the habitat value of paddock trees should not be discounted, due to their capacity to support specific taxa including birds and terrestrial invertebrates (Le Roux et al., 2015; Oliver et al., 2006). The observed preference for woodlots/shelterbelts over paddock trees may reflect the assumption that these configurations provide better habitat, or possibly lower awareness of the comparable benefits of widely-spaced configurations. However, it is likely that preferences for these configurations relate more closely to farming practicality or demand for other ecosystem services, particularly those related to shelter.

Shelter (or shade) for livestock was ranked second-highest for importance by respondents (Table 6.8). The high ranking of this service may relate to the high proportion of respondents (84%) who indicated that their enterprise involved a livestock component. In livestock systems where stock are vulnerable to extreme conditions caused by wind, wind speed reduction provided by agroforestry is likely to be valued highly by farmers. Although woodlots (and to a lesser extent paddock trees) do offer potential shelter benefits (Baker et al., in press), correctly-oriented shelterbelt configurations have the potential to provide more effective shelter than woodlots or paddock trees per area planted. Preferences for shelterbelts/woodlots over paddock trees can therefore be considered to align with the perceived high importance of providing shelter and shade for livestock. Given the high ranking of both biodiversity and shelter-related services, it is also possible that the observed lack of difference in relative utility between woodlots and shelterbelts reflects perceived conflicts between configuration requirements for each of these services.

Aside from those relating to biodiversity and shelter, other highly-ranked services included visual amenity and erosion control (wind and surface water) (Table 6.8). Evidence linking the specific agroforestry design attributes tested in this study to provision of these services is sparse. In terms of erosion control, the relative capacity of the three tested species compositions to affect soil water infiltration and hence reduce surface water run-off has, to our knowledge, not be tested. There is some evidence to suggest that ‘clumped’ agroforestry configurations are more effective at improving infiltration compared to widely-spaced configurations, although these findings appear to be contingent on stock being excluded from the clumped configurations, resulting in reduced soil compaction (Lunka & Patil, 2016). Nonetheless, the preference for shelterbelt/woodlot configurations over paddock trees in this study aligns with the perceived importance of this service. While the relative visual amenity of different configurations and species compositions is likely to be highly subjective, the design attribute preferences observed in this study broadly align with Grala et al. (2010) who found that straight rows of trees appealed most to farmers in terms of aesthetics, followed by groups of trees nested between fields and single trees. The same study also found that shelterbelts consisting of a mixture of trees and shrubs were preferred over mixed conifer/hardwood shelterbelts (Grala et al., 2010).

In addition to considering the relevance of highly-ranked ecosystem services in the context of sample characteristics and observed design preferences, it is also useful to consider services that received low rankings. Relatively low rankings for cropping-related services may be a product of cropping-only systems being under-represented in the sample compared to national figures (Table 6.3). The low ranking of timber production and the preference against the single species composition types may be partly explained by the relatively low proportion of respondents who indicated that their current farms contain commercial timber species (37.33%), as a lack of previous experience with forestry may reduce the appeal of timber-oriented agroforestry systems. Farmers may also be aware of the challenges involved in generating a meaningful return from small-scale production of timber on farms, particularly if they are unfamiliar with the market and if quality is affected by issues such as heavy branching in shelterbelts. Attitudes towards timber production on farms may also be negatively influenced by awareness of the failures of the forestry Managed Investment Schemes implemented across Australia in the 2000s (Brown et al., 2010). Disservices also received lower rankings of importance compared to services (Table 6.8) suggesting a positive perception of the advantages of agroforestry or possibly lower awareness of disservices, which are generally under-researched, compared to services (Saunders, 2020).

6.5.3 Implications for agroforestry adoption

The overall preference for agroforestry alternatives over the 'do nothing' scenario suggests positive appeal of agroforestry within the study area. This positive appeal could indicate that there is a willingness amongst farmers in temperate agricultural areas of Australia to consider agroforestry, and therefore potential to improve agroforestry adoption where conditions are suitable. However, results from this study also suggest that farmer perceptions of relative economic advantage are likely to play a key role in determining adoption outcomes. Agroforestry research and extension should therefore consider ways to reduce establishment costs, improve farm-scale return on investment, or better demonstrate existing levels of return. Promoting designs that offer tangible returns or provide ecosystem services that confer productivity benefits, may assist in this regard. Development of methods to value a wide range of agroforestry-related ecosystem services at the farm scale is also likely to form an important part of this strategy.

Results relating to farmer preferences for different combinations of agroforestry system characteristics can assist in the design of systems that are more likely to appeal to potential adopters. For example, given observed preferences for configuration and species composition, extension agencies whose primary objective is to encourage adoption of agroforestry may opt to focus on promoting mixed native species shelterbelts or woodlots to achieve the best results. Considering the relative influence of each attribute, research in agroforestry design could also be prioritised according to preferences, for example by exploring which configurations and extents of low-cost mixed native species plantings deliver the greatest benefits. Although a strong preference for mixed native species compositions was observed in this study, it is important to recognise that individual preferences varied across the sample. While there is value in using observed preferences of a cohort to guide extension and research, a range of flexible options should be considered and promoted as these preferences will not apply to all individuals within a given area.

Based on what is known about the capacity of different agroforestry systems to provide particular ecosystem services, preferences for agroforestry design were broadly aligned with demand for ecosystem services (e.g. biodiversity, shelter, erosion control, and visual amenity). This finding indicates that farmers have some awareness of which agroforestry designs are likely to provide particular ecosystem services. It also suggests that farm-scale measurement and valuation of ecosystem services is likely to be a useful tool in encouraging adoption of agroforestry. However, links between agroforestry design attributes and some ecosystem services and disservices are tenuous. Research is needed to improve our understanding of how design attributes, particularly those with a high level of influence on decision-making (e.g. species composition), affect the provision of in-demand services (e.g. shelter, biodiversity). To confirm alignment between preferences for ecosystem services and agroforestry design attributes, hybrid models could also be employed to quantify interactions between service rankings and attribute preferences.

There is scope for further research to expand on the findings of this study, particularly in relation to whether agroforestry design preferences or ecosystem service demand vary depending on farm property size or enterprise type. Previous studies have shown that property size has the potential to influence adoption of management practices (Baumgart-Getz et al., 2012), although case studies in temperate agroforestry are sparse. Although we expected that some preference heterogeneity within the sample would be attributable to property size or enterprise type, the nature and size of the sample did not allow for analysis of preferences in relation to these factors. Further research is needed to assess whether different enterprise types have strong preferences regarding particular agroforestry design attributes (e.g. configuration, extent), and whether these preferences vary depending on property size. Other factors such as the level of farm mechanisation, scale of production, and ownership arrangements may also influence preferences and could therefore be worth exploring. This would enable better targeting of extension and policy efforts aimed at encouraging agroforestry adoption in temperate regions.

6.6 Conclusions

Realising the potential for adoption of agroforestry in temperate regions requires us to understand farmer preferences relating to the design of agroforestry systems and the services that they provide. This study explored farmer preferences for temperate agroforestry design and ecosystem services, using data from a discrete choice experiment survey undertaken in temperate agricultural regions of Australia. Building on a substantial body of research on agroforestry adoption, the findings strengthen our understanding of farmer decision-making and demand in relation to temperate agroforestry.

Results of the choice experiment, including strong preferences and willingness to pay for specific agroforestry species compositions and configurations, are useful in guiding strategies for increasing adoption of agroforestry within temperate agricultural regions. To expand the relevance of these results and build a broader literature base, methods from this study could be applied to other temperate regions which hold similar potential for agroforestry expansion. This may also allow for analysis of the effects farm enterprise type and property size on design preferences, which would in turn enable better targeting of extension efforts aimed at increasing agroforestry adoption.

Some of the broader findings also provide direction for future research on agroforestry economics, for example the influence of cost on decision-making and the alignment between farmer preferences for agroforestry design and ecosystem service demand. These findings illustrate the importance of current and ongoing efforts to demonstrate the economic viability of agroforestry through various means, including by valuing ecosystem services provided by agroforestry at the farm scale. To improve knowledge and outcomes in this area, further evidence is needed to support links between design attributes and provision of key ecosystem services.

Findings from this study offer novel insights into farmer preferences for temperate agroforestry design and associated priorities for ecosystem service provision, within the study area and further afield. These insights improve capacity to increase adoption of agroforestry in temperate regions through effective research, policy, and extension. Evidence to support these initiatives could be strengthened by extending this research to investigate public preferences for agroforestry system design.

Chapter 7: General discussion

7.1 Overview

This thesis examined whether concepts of natural capital accounting (NCA) could be usefully and practically applied to improve farm-scale decision-making and encourage adoption of agroforestry. A key focus was understanding how existing NCA concepts, which are traditionally applied at landscapes scales (e.g. condition assessment and ecosystem service valuation), could be adapted for application to agroforestry systems at fine scales (i.e. farm or paddock scale). Current methods and tools for applying NCA were reviewed, ecosystem services provided by agroforestry systems were identified, and field experiments were conducted to quantify provision of relevant services at fine scales – with a particular focus on how fine-scale condition affects service provision.

The research chapters identified several gaps in the evidence base for fine-scale ecosystem service provision within agroforestry systems and provided methods to address them. Specifically, individual chapters in this thesis examined: whether existing NCA tools and concepts can be usefully applied in the context of farm-scale agroforestry, how shelterbelt vegetation structure and ecosystem service provision are affected by species choice and age, and how species choice in shelterbelts affects local invertebrate community composition and the distribution of functionally-important invertebrates in adjacent pasture. Finally, I examined how farmer preferences for ecosystem services and agroforestry design attributes influence farm-scale decision-making.

In this chapter, findings from all research chapters are synthesised and discussed to demonstrate that with some adaptation, NCA concepts can be practically and usefully applied to agroforestry systems at fine scales. Firstly, I discuss the need for novel approaches in the valuation of agroforestry assets in order to build an effective business case for private investment in agroforestry, and how the NCA framework could be adapted to suit this purpose. Drawing on findings from the review (Chapter 2) and experimental chapters (Chapters 3-6), I then discuss how adapting the framework for this purpose requires us to 'scale down' concepts of ecosystem service measurement and valuation (i.e. consider how they could be used at the farm or paddock scale, as opposed to coarser scales). Using shelterbelts as an example, I also discuss how agroforestry design choices impact asset 'condition' and ecosystem service provision, referring to experimental chapters to support the use of 'structural characterisation' as a practical way of assessing condition. Based on findings from all chapters, an example of paddock-scale valuation of a suite of ecosystem services provided by shelterbelt systems is provided. I then discuss implications of the key findings from this thesis for agroforestry policy and practice. Finally, suggestions are made for future research to improve the usefulness and practicality of NCA in agroforestry decision-making.

7.2 The natural capital accounting framework: a novel approach to valuation of agroforestry assets

Agroforestry enhances provision of multiple ecosystem services in agricultural landscapes at a range of scales (Jose, 2009). As demonstrated in the conceptual model presented in Chapter 2 (Figure 2.3), many ecosystem services (e.g. microclimate regulation, amenity, and timber production) deliver benefits at the farm or paddock scale by improving productivity, profitability, or property value (e.g. Baker et al., 2018; Polyakov et al., 2015). The value of other services, such as carbon sequestration, may extend far beyond the boundary of the farm (Flugge & Abadi, 2006). As the evidence base for ecosystem service provision in agroforestry systems across these scales continues to expand, there is growing recognition that agroforestry will form an important part of future multifunctional landscapes. Despite this promise, adoption of agroforestry remains constrained – particularly in temperate industrialised agricultural landscapes, including those in Australia (Black et al., 2000; Zomer et al., 2014).

While this unrealised potential is likely the result of a combination of factors, previous studies suggest that financial factors (i.e. establishment or opportunity costs, return on investment) have a relatively high level of influence on a farmer's acceptance of agroforestry (Fleming et al., 2019). Findings from Chapter 6 of this thesis confirmed the importance of financial factors in determining decisions to adopt agroforestry and highlighted the breadth of ecosystem services that farmers are likely to consider in those decisions. Traditional approaches to valuing farms and their assets have tended to either ignore or undervalue agroforestry assets, often focusing only on consumptive forms of value (i.e. the value of the timber) (e.g. Herbohn et al., 2009) and ignoring non-use values (e.g. the value of amenity, biodiversity, erosion control, or microclimate regulation). To maximise potential for agroforestry adoption, there is a need to demonstrate the capacity of agroforestry systems to deliver a wide range of benefits to farmers and/or their investors (Pannell, 1999). This will require novel approaches to valuation of agroforestry assets at appropriate scales.

Chapter 2 of this thesis outlined how the NCA framework, underpinned by concepts of ecosystem service measurement and valuation, provides the structure for a novel approach to valuation of agroforestry assets at the farm scale. This approach involves: considering agroforestry assets as one form of 'natural capital' on farms, identifying and measuring the wide range of ecosystem services which flow from that capital in the context of a particular farm, and capturing the multiple forms of value delivered by those services at specific scales (in this case, the farm or paddock scale). Being broader and more flexible than traditional valuation methods, evidence from this thesis suggests that this approach is likely to be more effective in justifying investment in agroforestry by a wide range of farmers and investors. As such, it serves as a potentially useful tool for increasing uptake of agroforestry.

7.3 Adapting ecosystem service valuation for application to agroforestry at fine scales

In applying NCA concepts it is important to first consider the scale of application, as scale has critical implications for measurement and valuation of ecosystem services. In many cases, the appropriate scale of application can be identified by considering the scale(s) at which management decisions and/or actions are taken. Since its inception, NCA has mostly been applied at regional or national scales to inform policy relating to land management or resource use (e.g. Goio et al., 2008; Haines-Young et al., 2006). In most western industrialised agricultural landscapes and certainly in Australia, management of natural capital on agricultural land (including investment in afforestation or agroforestry) is carried out at the scale of individual farming businesses. As such, Chapter 2 of this thesis concluded that in applying NCA concepts for the purpose of justifying private investment in agroforestry, the appropriate scale of application is the farm scale. Depending on the farm layout, the scale of application may be refined even further to that of a single paddock. As highlighted in Chapter 2, fine scale application requires some adaptation of existing NCA concepts and methods, particularly ecosystem service measurement and valuation. While coarse estimates of ecosystem service provision (and associated benefits) are often sufficient for regional/national application of NCA, such estimates are generally too inaccurate for the purpose of evaluating benefits to famers and/or farm investors.

The literature review presented in Chapter 2 found that, when scaling down NCA concepts for application to agroforestry at the farm or paddock scale, significant gaps emerge in the evidence base for ecosystem service provision (i.e. gaps in service identification, measurement, and valuation). Findings from Chapter 6 suggested that these gaps limit our potential to encourage agroforestry adoption, as the survey showed that farmers value a wide range of services, from biodiversity-related services to visual amenity. While services such as wind speed reduction are well understood (Cleugh, 2002), others, particularly cultural and biodiversity-related services, remain under-researched. Moreover, Chapter 2 found that further research is required to understand trade-offs and synergies between multiple ecosystem services. These findings align with other recent reviews calling for evidence of enterprise-specific agroforestry benefits at fine scales (e.g. Baker et al., 2018; England et al., 2020). Experimental chapters in this thesis addressed several of these gaps. Chapter 3 quantified paddock-scale provision of multiple ecosystem services by different types of shelterbelts over time, concluding that important trade-offs exist between wind speed reduction, wood production, carbon sequestration, and biodiversity. Chapters 4 and 5 predicted fine-scale provision of biodiversity-related services in shelterbelt systems, specifically those provided by invertebrate fauna (e.g. pollination and pest control). Chapters 3-5 also demonstrated that provision of key ecosystem services by different types of agroforestry assets can be measured or modelled over time at fine scales, thereby supporting fine-scale implementation of NCA in agroforestry systems.

While this thesis focused on the application of NCA concepts at the farm scale, it is important to recognise that farmers' decisions to adopt particular types of agroforestry will also have consequences for provision of ecosystem services beyond the farm boundary. For example, findings from Chapter 3 highlighted how agroforestry design choices such as species selection can significantly affect carbon sequestration, which has global consequences for climate change mitigation. Shelterbelts were also shown to play an important role in providing habitat for invertebrate communities (Chapter 4), reaffirming the role of agroforestry assets in conserving biodiversity in agricultural landscapes. In some cases, optimising delivery of farm-scale benefits may result in perverse outcomes for delivery of public benefits. One such example is the removal of remnant vegetation, which often provides critical habitat structure for native fauna (e.g. tree hollows), to accommodate linear shelterbelts containing fast-growing species which may confer greater productivity benefits (Manning & Lindenmayer, 2009). Although applying NCA at fine scales through the lens of individual farmers is useful and appropriate for certain purposes (e.g. building a business case for agroforestry adoption), broader consequences of encouraging adoption of particular types of agroforestry should be considered in order to protect public values. Using NCA to understand incentives at the farm scale can also be helpful in encouraging appropriate adoption of agroforestry.

7.4 Accounting for differences in the fine-scale condition of agroforestry assets

In addition to refining concepts of ecosystem service measurement and valuation, Chapter 2 identified another critical element of the NCA framework that requires adaptation for useful application at fine scales: measuring and tracking asset 'condition'. This became a central focus of this thesis.

Historically, NCA has been widely applied to monitor stocks of natural capital and associated flows of ecosystem services in 'natural' ecosystems such as forests or grasslands. The 'condition' of an asset in this context is often assessed relative to some form of baseline for that particular ecosystem - usually an idea of an unaltered, intact, or pre-colonisation/industrialisation state (Helm, 2019).

Biophysical metrics such as species-based biodiversity indicators, vegetation cover, connectivity, and soil/water quality are commonly used to assess the condition of natural capital assets in these contexts (Maes et al., 2020). Agroforestry assets differ from 'natural' systems in that a baseline condition is often lacking or irrelevant, as they are usually isolated within agricultural landscapes, actively managed (e.g. pruned/thinned/grazed) and may be carefully designed to meet particular objectives. Condition is nonetheless critical to measure at fine scales, as certain biophysical characteristics of agroforestry assets are known to significantly affect provision of key ecosystem services, e.g. the effect of shelterbelt tree height on wind speed reduction (Chapter 3). As identified in Chapter 2 of this thesis, established concepts of asset condition (i.e. those relevant to 'natural' systems at broad scales) therefore need to be adapted in order to be usefully applied to agroforestry at fine scales.

Understanding condition is critically important in establishing agroforestry systems, as farmers are faced with several design choices including the extent of tree/shrub cover, the configuration of trees relative to other elements of the farm, and the selection of tree species. These design choices determine the condition of agroforestry assets and therefore the types and amounts of ecosystem services, and associated benefits, that are delivered. Chapter 3 demonstrated that tree species selection in shelterbelts significantly affected provision of services such as wind speed reduction, wood production, carbon sequestration, and biodiversity. Another example of this can be seen in Chapters 4 and 5, which showed that tree species selection influenced the abundance of particular groups of invertebrates (e.g. native pollinators) within shelterbelts, and to a lesser extent in areas of pasture adjacent to them. These findings highlight the links between agroforestry asset design, condition, and the provision of services and benefits at fine scales.

Findings from the choice experiment in Chapter 6 demonstrated that farmers are motivated to make agroforestry design choices which maximise benefits that align with their objectives. Findings from Chapter 6 also showed that farmers have strong preferences for certain attributes of agroforestry design, particularly species selection. For the NCA framework to be useful in guiding agroforestry design decisions, it is therefore critical that asset condition is assessed in such a way that connects design choices (particularly species selection) to delivery of key services and associated benefits at the farm scale.

Recent work by the SEEA-EEA Revision Working Group 2 proposes that in order to be effective, NCA condition metrics should be quantitative, feasible to measure, and should relate to flows of services and benefits most relevant to the context of application (Czúcz et al., 2019). Results of the survey in Chapter 6 showed that farmers consider a wide range of ecosystem services to be important, although shelter and biodiversity-related ecosystem services were ranked most highly by the survey participants. When applying NCA to encourage adoption of agroforestry by farmers or to inform their decision-making, it is therefore important that condition metrics relate to a broad range of ecosystem services. Chapter 3 examined whether vegetation structure is a useful and practical condition metric in the application of NCA to agroforestry at fine scales. Results showed that tree species selection, a key design consideration for farmers, significantly affected a range of structural and floristic attributes in shelterbelts including height, porosity, stand basal area, and plant species diversity. Findings also demonstrated that these structural attributes are linked to fine-scale provision of not only shelter, but a range of other services such as wood production, carbon sequestration, and biodiversity. These findings suggest that vegetation structure provides a useful condition metric for agroforestry assets in being quantitative, feasible to measure, and related to a wide range of ecosystem services that are valued by farmers.

Chapters 4 and 5 expanded the evidence base for impacts of tree species selection on invertebrate-related ecosystem services, which was found to be lacking (Chapter 3). Findings from these chapters suggest that when applying NCA to agroforestry, as is the case with concepts of ecosystem service measurement and valuation (Section 7.3), condition assessment requires consideration of both fine-scale and landscape-scale factors. Results showed that although shelterbelts provide important habitat for certain invertebrate taxa and enhance potential for pollination through wind speed reduction, potential for provision of invertebrate-related ecosystem services in adjacent pastures is not strongly affected by differences in shelterbelt vegetation structure. Although fine-scale habitat features can affect biodiversity in agricultural landscapes, landscape-scale metrics such as connectivity and overall complexity have also been shown to have strong influence (Chaplin-Kramer et al., 2013). The low influence of shelterbelt vegetation structure on distribution of invertebrates in adjacent paddocks observed in Chapter 5 indicates that both fine scale and landscape scale factors are important in determining flows of biodiversity-related ecosystem services from agroforestry assets. While vegetation structure offers a practical solution to the issue of condition assessment, these findings suggest the need for consideration of additional, landscape-scale condition metrics in order to adequately predict flows of farm-scale services and benefits from agroforestry assets - particularly in the case of biodiversity-related services.

7.5 Example of paddock-scale agroforestry ecosystem service valuation

Findings from Chapter 6 of this thesis suggest that information demonstrating a return on investment through a range of values would be useful to farmers in informing agroforestry adoption and design decisions. The NCA framework accommodates various options for valuation of ecosystem services that flow from natural capital assets. In some cases, valuation may be forgone entirely, with the stock/condition of natural capital assets simply measured over time in biophysical terms (e.g. water volume or forest cover). In cases where values are assigned, the framework allows for inclusion of multiple forms of value, not only monetary. However, monetary values can, in many instances, be useful in serving as a common unit or ‘language’, enabling the value of services flowing from natural capital to be accounted for in the same way as other cash flows on the farm. The agricultural bioeconomic model, Imagine, has recently been expanded to enable monetary valuation of multiple ecosystem services provided by shelterbelts at the paddock scale (wood production, carbon sequestration, shelter, and amenity) (Mendham, 2018). Using these methods, the cumulative paddock-scale returns from goods and services flowing from a two-belt configuration of pine, eucalypt, and mixed native shelterbelts adjacent to pasture is estimated here for a 25-year period (Table 7.1).

Table 7.1: Returns accumulated over 25 years (discounted at a rate of 8%) from services of wood production, shelter for pasture and livestock, carbon sequestration, and amenity in a two-belt configuration of *Pinus radiata*, *Eucalyptus nitens*, and mixed native shelterbelts adjacent to pasture. Returns are calculated using the Imagine bioeconomic model as per methods detailed in the report by Mendham (2018). These methods value wood production through timber prices, shelter through production functions based on results of field experiments, carbon sequestration through the Australian Emissions Reduction Fund, and amenity through studies on the impact of trees on land prices. Model inputs (height, standing wood volume, extent of shelter benefit) are adjusted based on quantitative comparisons of different shelterbelt types presented in Table 3.1.

	Cumulative discounted returns (per paddock, at 25 years)				
Shelterbelt type	**Wood production**	**Shelter**	**Carbon sequestration**	**Amenity**	**Total**
Pinus radiata	$3,063.66	$6,870.61	$4,530.76	$798.54	$15,263.57
Eucalyptus sp.	$3,204.68	$5,961.77	$4,389.04	$798.54	$14,354.03
Mixed native species	$0	$5,490.32	$2,110.77	$798.54	$8,399.64

While the values in Table 7.1 reflect a simplified scenario, they are useful in demonstrating how monetary valuation of ecosystem services (in this case through summarising market values) can be used to compare the relative benefits of different types of agroforestry. The advantage of this approach is that it clearly demonstrates a return on investment through a range of both extractive and non-extractive values. Conceptual models of fine-scale ecosystem service provision (e.g. Figure 2.3) could be used to guide selection of key ecosystem services to include in valuation, and to identify relevant valuation pathways.

This approach also demonstrates how bioeconomic models such as Imagine may be further expanded to account for differences in the condition of agroforestry assets. Based on results from Chapter 3 of this thesis, model inputs were adjusted to account for differences in condition via vegetation structure, (e.g. adjusting the extent of shelter benefit to account for differences in optical porosity). While gaps remain in our understanding of how vegetation structure affects other services in agroforestry systems (e.g. amenity), there will be opportunities to continue adjusting and refining bioeconomic models as these gaps are addressed.

The key limitation of a monetary valuation approach is that it excludes ecosystem services that are either too difficult, or not appropriate to value in monetary terms (e.g. cultural significance, recreation, biodiversity). While techniques do exist to estimate values for these services (as covered in Chapter 2), their value to individual farmers is likely to be highly subjective. In contradiction to outputs from the Imagine model (Table 7.1), results from the discrete choice experiment (DCE) presented in Chapter 6 showed that farmers are willing to pay more for mixed native species shelterbelts, and to a lesser extent eucalypt shelterbelts, compared to pines. This suggests that market values, as summarised by the Imagine model, do not adequately capture the full range of ecosystem services that are valued by farmers.

Further, fine-scale valuation of ecosystem services could lead to misrepresentation of services that are more accurately predicted at broader scales. While it is possible to predict provision of some services based on fine-scale condition metrics (e.g. shelter), findings from Chapter 5 of this thesis suggest that prediction of other services (e.g. services related to biodiversity) also requires consideration of landscape-scale factors. It will be important to consider these nuances as valuation tools continue to develop and expand.

7.6 Implications for policy and practice

Widespread adoption of agroforestry has been proposed as a solution to multiple pressures including biodiversity loss, food/fibre security, land degradation, and climate change (Flugge & Abadi, 2006; Waldron et al., 2017). Ambitious global and national afforestation targets such as the Trillion Trees Initiative (WEF, 2020) and Australia's goal to plant one billion new plantation trees (DAWE, 2020) add further impetus to the expansion of agroforestry, as a means by which such programs can avoid potential negative impacts on food security and rural livelihoods (Reij & Winterbottom, 2015). While this growing demand for agroforestry provides opportunities at regional and local scales, it also presents challenges for policy-makers and practitioners. For example, the challenges of encouraging individual farmers to participate in meeting global or national afforestation targets through agroforestry, and deciding which types of trees to plant in which locations or configurations. NCA approaches outlined in this thesis could be useful in addressing these challenges.

Commitments to plant trees at grand scales can be useful in generating momentum for afforestation and are often critical to the development of national or regional policy. However, there is growing concern that large scale tree planting initiatives lack strategies to ensure long term success (Brancalion & Holl, 2020). While such initiatives are often designed to address broad scale issues such as climate change, successful establishment and maintenance of trees in agricultural landscapes rely heavily on investment and ongoing commitment from farmers. Fine scale application of NCA as described in this thesis could ensure that trees are selected and planted with due consideration of farm-scale objectives in addition to those of the broader public. Using ecosystem service measurement and valuation techniques from this thesis could encourage greater levels of initial participation and assist in ensuring long term delivery of intended outcomes.

National policies that provide direct incentives to farmers are a common means for encouraging uptake of sustainable agricultural practices, including agroforestry and restoration. In Australia, the latest example is the Agriculture Biodiversity Stewardship Package which will reward farmers for undertaking activities that support biodiversity, including planting vegetation to create or enhance habitat (DAWE, 2021a). There is also growing interest from the private sector, particularly finance and insurance industries, in creating incentives for uptake of agricultural practices that mitigate business risks (e.g. Food Agility CRC, 2018). While incentives for encouraging adoption of agroforestry may differ in their intent, they share a common need for robust methods to value the broad range of services provided by trees on farms. This thesis can contribute to development of such methods. For example, the conceptual models presented in Chapter 2 can be used to identify the various ways in which agroforestry assets address risks to farming businesses. Based on the quantitative results presented in Chapters 3-5 (e.g. relative potential for wind speed reduction based on height and porosity, or potential for pollination benefits based on wind speed reduction and floral diversity), policy makers or financial institutions may then refer to the structural attributes of agroforestry assets as the basis for administering incentives over time. Ecosystem service trade-offs identified in this thesis (Chapters 2, 3) (e.g. decreased biodiversity in high carbon shelterbelts) will also be critical to the implementation and success of policies with multiple objectives, such as the Carbon + Biodiversity Pilot proposed as part of the Agriculture Biodiversity Stewardship Package, which aims to integrate markets for carbon and biodiversity credits (DAWE, 2021b). Understanding these trade-offs could also inform development of agroforestry designs and management techniques that address multiple objectives (e.g. eucalypt plantations that incorporate diverse mid-storey and under-storey vegetation, or coppicing to maintain vegetation in these lower layers).

Valuation of agroforestry assets plays an important role in underpinning development of new markets for ecosystem services, similar to the established market for Australian carbon credit units. This thesis contributes novel methods (e.g. use of structure as a condition metric) and empirical evidence (e.g. enhancement of pollination potential through wind speed reduction) to support fine scale measurement of ecosystem services, thereby improving the accuracy of valuation. Markets for ecosystem services may provide opportunities for farmers to diversify their income and cover a greater portion of their agroforestry establishment costs. However, markets to coordinate exchange of goods and services that flow from natural capital should be developed with caution, not least because they can only value ecosystem services that we are able to price (Raworth, 2017). There is also the risk that assigning a monetary value to vegetation on farms will erode existing cultures of goodwill by creating an expectation of financial compensation. While markets and economic incentives are potentially useful for encouraging expansion of agroforestry, they should not replace other fundamental ways in which we currently approach conservation and restoration.

7.7 Future directions

In aiming to inform development of a farm-scale business case for agroforestry, this thesis focused on measurement of relatively tangible ecosystem goods and services: shelter, wood production, carbon sequestration, and biodiversity. While findings confirmed the importance of these services to farmers (Chapter 6), they also highlighted the importance of less tangible cultural ecosystem services such as visual amenity and recreational/cultural practice. Cultural services such as these are typically understudied in ecosystem service literature but are important factors in natural resource decision-making (Queiroz et al., 2017). There would be value in expanding on the work of this thesis by undertaking more targeted studies on cultural ecosystem services. For example, DCEs could be used to estimate preferences and willingness to pay for the visual amenity provided by trees at both the farm and landscape scale.

Experimental work in this thesis focused on shelterbelts, showing that shelterbelt condition (driven by tree species selection) affects provision of ecosystem services at fine scales. Although findings from Chapter 6 confirmed that shelterbelts are a popular configuration in temperate agroforestry, there are many alternative options for integrating trees into farms including riparian buffers, woodlots, and widely-spaced trees. To enable farmers to compare relative costs and benefits of a wider range of agroforestry options, there would be value in expanding the scope of reviews and conceptual models for ecosystem service provision (Figure 2.3) as well as quantitative studies on effects of condition on ecosystem service provision (such as those presented in Chapters 3-5), to cover other agroforestry configurations. While some studies explore service provision in other agroforestry types, e.g. Baker et al. (in press) who examined wind speed reduction in patches of remnant woodland on farms, there is a lack of studies directly comparing service provision between agroforestry configurations.

As suggested in Chapter 2 of this thesis, incorporating NCA concepts into existing agroforestry planning tools such as the Farm Forestry Toolbox (Warner, 2007), or into farm-scale ecological accounting systems such as the Ecological Balance Sheet (Ogilvy, 2015) may broaden their useability. As discussed in Section 7.5, work is also underway to incorporate valuation of a broader range of agroforestry ecosystem services into existing bioeconomic models such as Imagine (Abadi et al., 2003). Findings from this thesis, particularly quantitative examination of effects of species selection on structure and ecosystem service provision, will contribute to the ongoing expansion and refinement of these tools. Ultimately, the aim should be to provide farmers with affordable tools to accurately predict provision of a broad range of ecosystem services by different types of agroforestry, so that they may design agroforestry systems that best suit their objectives.

7.8 Conclusions

Overall, findings from this thesis demonstrated that with some adaptation and further research, natural capital accounting concepts can be practically and usefully applied to agroforestry systems at fine scales.

Novel approaches to valuation of agroforestry assets are needed to build an effective business case for private investment in agroforestry and to inform optimal design of agroforestry systems. Measurement and valuation of multiple ecosystem services, within the NCA framework, can be applied for these purposes. Critically, in applying NCA to agroforestry, it is necessary to adapt existing concepts (e.g. ecosystem service valuation) for application at finer scales (i.e. farm, or paddock scale). Focusing on farmers as the beneficiaries in ecosystem service valuation exercises is key, as agroforestry adoption and design decisions are typically made at the scale of individual farming businesses. While fine-scale valuation is important, broader scale costs/benefits should also be considered. This thesis contributes methods and data to support measurement and valuation of a range of ecosystem services in agroforestry systems at fine scales.

To evaluate agroforestry systems at fine scales, this thesis also highlighted the importance of adapting the concept of 'condition assessment' which is central to the NCA framework. The fine-scale condition of agroforestry assets determines ecosystem service provision at the farm scale and is influenced by design decisions such as tree species selection. Results from Chapter 3 showed that structural characterisation of agroforestry assets is a practical way to assess condition and inform agroforestry design. However, findings from Chapter 5 suggest that landscape-scale condition metrics also play an important role in determining provision of some biodiversity-related services.

This thesis demonstrated that NCA concepts can be used to highlight the broad values provided in different types of agroforestry systems at fine scales, thereby assisting in increasing adoption of agroforestry by farmers. The thesis also showed that application of NCA concepts to agroforestry can assist farmers in designing agroforestry systems which best suit their objectives. Further research is needed to address remaining gaps in the evidence base for fine-scale provision of ecosystem services, with particular focus on biodiversity and cultural services. Ongoing development of fine-scale valuation tools stands to improve decision-making and policy development, although monetary valuation of agroforestry assets should proceed with caution to avoid perverse outcomes.

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Appendix A

Table A1: Chapter 3 study site information. Site ID contains a letter denoting the age category (M: 15-30 years, J: 6-14 years, Y: 2-5 years) followed by a letter denoting the composition type (P: *P. radiata*, E: *Eucalyptus nitens*, N: mixed native).

Site ID	Age (years)	Belt/block edge	Number of rows	Width (m)	Annual rainfall (mm)	Elevation (m a.s.l)
MP1	~20	Belt	7	21	663	202
MP2	~30	Belt	3	18	531.8	190
MP3	23	Belt	6	25	517.9	192
MP4	17	Belt	5	20	547.9	164
MP5	~20	Belt	3	15	566.9	358
ME1	~20	Belt	5	16	663	196
ME2	~20	Belt	6	19	652.6	132
ME3	~20	Belt	4	15	621.9	177
ME4	~20	Belt	7	25	621.9	174
ME5	~15	Belt	4	18	613	144
MN1	21	Belt	5	15	525.4	161
MN2	17	Belt	4	16	566.9	366
MN3	18	Belt	5	15	525.4	158
MN4	~28	Belt	6	23	580.9	163
MN5	23	Belt	4	15	686.7	212
JP1	~14	Belt	3	16	491.3	152
JP2	9	Belt	6	21	491.3	165
JP3	13	Belt	5	23	547.9	181
JE1	14	Block edge	6	20	507.8	158
JE2	14	Belt	5	16	686.7	204
JE3	~13	Belt	5	18	621.9	169
JN1	13	Belt	6	20	471	161
JN2	13	Belt	5	22	566.9	373
JN3	8	Belt	3	15	621.9	185
YP1	5	Block edge	3	20	491.3	194
YP2	5	Belt	5	21	643.4	166
YP3	5	Belt	3	15	683.7	149

Site ID	Age (years)	Belt/block edge	Number of rows	Width (m)	Annual rainfall (mm)	Elevation (m a.s.l)
YE1	4	Belt	4	16	652.6	158
YE2	4	Block edge	4	20	621.9	155
YE3	3	Belt	5	21	683.7	149
YN1	3	Belt	8	23	491.3	159
YN2	3	Belt	5	17	525.4	186
YN3	3	Belt	8	25	507.8	161

Appendix B

Table B1: Classification of invertebrate taxa within orders Coleoptera, Hymenoptera, Diptera, and Hemiptera to functional sub-group. References: (a) (CSIRO, 1991) (b) (Bailey, 2007) (c) (Elliott & DeLittle, 1985) (d) (Saunders & Retra, 2015) (e) (Bellati et al., 2012) (f) (McQuillan et al., 2007) (g) (Hoffmann & Andersen, 2003) (h) (Andersen, 1997)

Order	Sub-group	Taxon	Reference(s)
Coleoptera	Chafer scarab	*Acrossidius tasmaniae*	(b) (f)
		Acrossidius pseudotasmaniae	
		Adoryphorus couloni	
		Scitala sericans	
		Sericesthis nigra	
		Sericesthis nigrolineata	
		Saulostomus villosus	
	Dung scarab	*Geotrupes spiniger*	(f)
		Onthophagus sp.	
		Euoniticellus fulvus	
	Pest scarab	*Heteronyx* sp.	(c)
		Diphucephala colaspidoides	
	Other scarab	All other *Scarabaeidae*	(a)
	Other pest beetle	*Isopteron* sp.	(c) (e)
		Adelium brevicorne	
		Celibe sp.	
		Pterohelaeus sp.	
		Cadmus australis	
		Paropsis charybdis	
		Paropsis porosa	
		Paropsis delittlei	
		Paropsisterna variicollis	
		Paropsisterna agricola	
		Peltoschema orphana	
		Paropsisterna bimaculate	
		Phoracantha semipunctata	
		Coptocerus rubriceps	
		Hesthesis cingulata	
		Ancita crocogaster	
		Probatodes plumula	
		Bostrichidae sp.	
	Weevil	*Curculionidae*	(a)
	Elaterid	*Elateridae*	(a)

Order	Sub-group	Taxon	Reference(s)
	Carabid	*Carabidae*	(a)
	Staphylinid	*Staphylinidae*	(a)
	Other predatory beetle	*Histeridae* *Coccinellidae* *Cleridae* *Cantharidae* *Trogossitidae* *Melyridae* *Lampyridae*	(a) (b)
	Pollinating beetle	*Cleridae* *Cantharidae* All other *Cerambycidae* *Mordellidae* *Buprestidae* *Lucanidae*	(a) (b) (d)
	Other beetle	All other Coleoptera	(a)
Hymenoptera	Bumblebee	*Bombus terrestris*	(a) (e)
	Honey bee	*Apis mellifora*	(a) (e)
	Native bee	*Exoneura* sp. *Colletidae* *Halictidae* *Megachilidae*	(a) (e)
	Predatory ant	*Myrmecia* sp.	(g) (h)
	Other ant	All other *Formicidae*	(a)
	Parasitic wasp	All Apocrita: 'Parasitica' *Tiphiidae* *Sphecidae*	(a) (b) (e) (f)
	Other wasp	All other Apocrita: 'Aculeata'	(a)
	Sawfly	*Symphata*	(a) (c)
Diptera	Pest fly	*Agromyzidae*	(a) (e)
	Pest blowfly	*Lucilia cuprina* *Lucilia sericata*	(a)
	Pollinating fly	All other *Calliphoridae* *Sarcophagidae* *Syrphidae* *Bombyliidae*	(a) (d) (e)

Order	Sub-group	Taxon	Reference(s)
		Lauxaniidae	
		Tabanidae	
		Rhiniidae	
		Stratiomyidae	
	Predatory fly	*Syrphidae*	(a) (b) (e)
		Tachinidae	
		Pipunculidae	
		Dolichopodidae	
		Empididae	
		Asilidae	
		Nemestrinidae	
		Bombyliidae	
	Other fly	All other Diptera	(a)
Hemiptera	Predatory Hemiptera	*Anthocoridae*	(a) (e) (f)
		Nabidae	
		Reduviidae	
		Geocoris sp.	
		Jalloides sp.	
		Cermatulus nasalis	
		Oechalia schellenbergii	
		Deraeocoris sp.	
		Setocoris sp.	
		Cyrtorhinus lividipennis	
	Aphid	*Aphididae*	(a) (b) (c)
	Psyllid	*Psyllidae*	(a) (c)
	Other pest Hemiptera	*Gelonus tasmanicus*	(c)
		Amorbus obscuricornis	
	Other Hemiptera	All other Hemiptera	(a)

Appendix C

Table C1. Results of analysis of variance for each level of invertebrate classification (variable), by season. All invertebrate counts were fourth root transformed for analysis. These tests included all four treatments (open pasture and *P. radiata*, *Eucalyptus sp.*, and mixed native shelterbelts). Dashes indicate cases where observations for that taxa/sub-group were too few for analysis (or zero observations).

	Treatment effect (p value) (d.f. = 3) **$^*p < 0.05$, $^{**}p < 0.01$, $^{***}p < 0.001$**		
Variable	**Spring**	**Summer**	**Autumn**
Total abundance			
Ground-active	0.862	0.234	0.837
Low-flying	0.017 *	0.563	0.710
By order			
Ground-active			
Araneae	0.864	0.053	0.035 *
Coleoptera	0.890	0.720	0.819
Diplopoda	0.548	0.451	0.442
Diptera	0.844	0.263	0.897
Hemiptera	0.747	0.555	0.647
Hymenoptera	0.426	0.043 *	0.094
Isopoda	0.060	0.016 *	-
Lepidoptera	0.322	0.136	0.439
Neuroptera	-	0.004 **	-
Orthoptera	-	0.131	0.027 *
Psocoptera	0.326	0.087	0.460
Scorpiones	0.564	0.601	0.222
Blattodea	0.561	0.691	-
Chilopoda	0.180	0.712	-
Odonata	-	-	-
Dermaptera	0.441	0.536	0.645
Low-flying			
Araneae	0.986	0.253	0.597
Coleoptera	0.037 *	0.016 *	0.262
Diplopoda	-	0.441	0.586

Variable	Treatment effect (p value) (d.f. = 3) *$p < 0.05$, **$p < 0.01$, ***$p < 0.001$ Spring	Summer	Autumn
Diptera	0.112	0.234	0.730
Hemiptera	0.083	0.210	0.006 **
Hymenoptera	0.031 *	0.029 *	0.003 **
Isopoda	-	-	0.596
Lepidoptera	0.710	0.738	0.645
Neuroptera	<0.001 ***	0.028 *	0.286
Orthoptera	-	0.242	0.018 *
Psocoptera	0.448	0.064	0.003 **
Scorpiones	-	-	-
Blattodea	-	0.414	0.441
Chilopoda	-	-	-
Odonata	-	0.441	-
Dermaptera	-	0.441	-
By sub-group			
Hymenoptera			
Bumblebee	0.789	0.038 *	0.005 **
Honey bee	0.098	<0.001 ***	<0.001 ***
Native bee	0.236	0.015 *	0.005 **
Other ant	0.074	0.104	0.153
Predatory ant	0.012 *	0.009 **	0.067
Parasitic wasp	0.110	0.565	0.147
Other wasp	-	0.797	0.796
Sawfly	-	-	-
Diptera			
Other fly	0.578	0.101	0.765
Pest fly	-	-	-
Pest blowfly	0.572	0.554	0.577
Pollinating fly	-	0.204	0.571
Predatory fly	-	0.498	0.632
Coleoptera			

	Treatment effect (*p* value) (d.f. = 3) ***p < 0.05, **p < 0.01, ***p < 0.001**		
Variable	**Spring**	**Summer**	**Autumn**
Other beetle	0.056	0.073	0.925
Chafer scarab	0.107	0.392	-
Dung scarab	-	0.145	0.586
Pest scarab	0.011 *	0.435	-
Other scarab	0.163	0.145	0.503
Other pest beetle	-	0.165	-
Weevil	0.834	0.312	0.500
Elaterid	0.159	0.019 *	-
Carabid	0.586	0.357	0.535
Staphylinid	0.524	0.062	0.214
Other predatory beetle	0.101	0.091	0.627
Pollinating beetle	0.071	0.003 **	-
Hemiptera			
Other Hemiptera	0.296	0.310	0.222
Other pest Hemiptera	-	-	-
Aphid	0.413	0.281	0.030 *
Psyllid	0.025 *	0.009 **	0.153
Predatory Hemiptera	0.269	0.178	-

Appendix D

Table D1. Classification of invertebrate taxa to functional groups. References: (a) (CSIRO, 1991) (b) (Bailey, 2007) (c) (Elliott & DeLittle, 1985) (d) (Saunders & Retra, 2015) (e) (Bellati et al., 2012) (f) (McQuillan et al., 2007) (g) (Hoffmann & Andersen, 2003) (h) (Andersen, 1997)

Order / Higher taxon	**Functional group**	**Taxon**	**Reference(s)**
Araneae	Generalist predator	All Araneae	(a)
Scorpiones	Generalist predator	All Scorpiones	(a)
Isopoda	Decomposer	All Isopoda	(a)
Diplopoda	Decomposer	All Diplopoda	(a)
Blattodea	Tree pest	*Porotermes adamsoni*	(c)
	Decomposer	All other Blattodea	(a)
Chilopoda	Generalist predator	All Chilopoda	(a)
Coleoptera	Major field pest	*Acrossidius tasmaniae* *Acrossidius pseudotasmaniae* *Adoryphorus couloni*	(b) (f)
	Minor field pest	*Scitala sericans* *Sericesthis nigra* *Sericesthis nigrolineata* *Saulostomus villosus* *Listronotus bonariensis* *Sitona discoideus* *Naupactus leucoloma* *Listroderes difficillis* *Desiantha diversipes* *Hapatesus hirtus* *Agrypnus* sp. *Conoderus* sp. *Isopteron* sp. *Adelium brevicorne* *Celibe* sp. *Pterohelaeus* sp.	(b) (f) (c) (e) (a)
	Decomposer	*Geotrupes spiniger* *Onthophagus* sp. *Euoniticellus fulvus*	(f)
	Tree pest	*Heteronyx* sp. *Diphucephala colaspidoides*	(c) (a)

Order / Higher taxon	Functional group	Taxon	Reference(s)
		Cadmus australis	
		Paropsis charybdis	
		Paropsis porosa	
		Paropsis delittlei	
		Paropsisterna variicollis	
		Paropsisterna agricola	
		Peltoschema orphana	
		Paropsisterna bimaculata	
		Phoracantha semipunctata	
		Coptocerus rubriceps	
		Hesthesis cingulate	
		Ancita crocogaster	
		Probatodes plumula	
		Bostrichidae sp.	
		Gonipterus scutellatus	
		Rhachiodes dentifer	
		Rhadinosomus lacordairei	
		Platypus subgranosus	
		Alterpus rubus	
		Scolytinae	
		Buprestidae	
	Pest predator	*Carabidae*	(a)
		Coccinellidae	
	Generalist predator	*Histeridae*	(a) (b)
		Staphylinidae	
		Coccinellidae	
		Cleridae	
		Cantharidae	
		Trogossitidae	
		Melyridae	
		Lampyridae	
		Cantharidae	
		Cleridae	
	Minor pollinator	*Cleridae*	(a) (b) (d)
		Coccinellidae	
		Cantharidae	
		All other *Cerambycidae*	
		Mordellidae	

Order / Higher taxon	Functional group	Taxon	Reference(s)
		Buprestidae *Lucanidae*	
	Other	All other Coleoptera	(a)
Hymenoptera	Major pollinator	*Bombus terrestris* *Apis mellifora*	(a) (e)
	Minor pollinator	*Exoneura* sp. *Colletidae* *Halictidae* *Megachilidae* *Tiphiidae* *Netelia producta* *Vespidae* *Tenthredinidae*	(a) (e) (d)
	Pest predator	All Apocrita: 'Parasitica' *Tiphiidae*	(a) (b) (e) (f)
	Generalist predator	*Sphecidae* *Vespidae* *Myrmecia* sp. All other Apocrita: 'Aculeata'	(a) (g) (h)
	Minor pest	*Bruchophagus* sp.	(a)
	Tree pest	*Symphata* *Sirex noctilio*	(a) (c)
	Other	All other Hymenoptera	(a)
Diptera	Stock pest	*Lucilia cuprina* *Lucilia sericata*	(a)
	Minor field pest	*Agromyzidae* *Boreoides* sp.	(a) (e)
	Minor pollinator	*Lucilia cuprina* *Lucilia sericata* All other *Calliphoridae* *Sarcophagidae* *Syrphidae* *Bombyliidae* *Lauxaniidae* *Tabanidae* *Rhiniidae*	(a) (d) (e)

Order / Higher taxon	Functional group	Taxon	Reference(s)
		Stratiomyidae	
	Pest predator	*Syrphidae* *Tachinidae* *Pipunculidae* *Dolichopodidae* *Empididae* *Asilidae* *Nemestrinidae* *Bombyliidae*	(a) (b) (e)
	Other	All other Diptera	(a)
Hemiptera	Pest predator	*Anthocoridae* *Nabidae* *Reduviidae* *Cermatulus nasalis* *Oechalia schellenbergii* *Deraeocoris* sp. *Setocoris* sp. *Cyrtorhinus lividipennis*	(a) (e) (f)
	Generalist predator	*Geocoris* sp. *Jalloides* sp.	(a) (e) (f)
	Minor field pest	*Aphididae*	(a) (b) (c)
	Tree pest	*Psyllidae* *Gelonus tasmanicus* *Amorbus obscuricornis*	(a) (c)
	Other	All other Hemiptera	(a)
Lepidoptera	Major field pest	*Oncopera rufobrunnea* *Oncopera intricata* *Plutella xylostella* *Pieris rapae* *Persectania ewingii* *Mythimna convecta* *Agrotis munda* *Agrotis infusa*	(f)
	Minor field pest	All other Lepidoptera	(a)
Neuroptera	Pest predator	All Neuroptera	(a)
Orthoptera	Major field pest	*Phaulacridium vittatum*	(b) (f)

Order / Higher taxon	**Functional group**	**Taxon**	**Reference(s)**
	Minor field pest	All other *Acrididae* *Teleogryllus commodus* *Gryllotalpa* sp.	(a) (b) (f)
	Other	All other Orthoptera	(a)
Dermaptera	Minor field pest	*Forficula auricularia*	(e)
	Generalist predator	*Labidura riparia (truncata)*	(e) (a)
	Other	All other Dermaptera	(a)
Psocoptera	Other	All Psocoptera	(a)
Mantodea	Generalist predator	All Mantodea	(a)
Odonata	Generalist predator	All Odonata	(a)