The role of biodiversity on pest control ecosystem services in UK apple orchards

Charlotte A. Selvey

A dissertation submitted for the degree of

Doctor of Philosophy

University College London

Centre for Biodiversity and Environment Research (CBER)
within the Department of Genetics, Evolution and Environment (GEE)
University College London

21th October 2021

Declaration

I, Charlotte Anne Selvey, confirm that the work presented in this thesis is my own. Where information has been derived from other sources, I confirm that this has been indicated in the thesis.

Charlotte Selvey

21st October 2021

Abstract

In this thesis I assess the ability of biodiversity to provide a functioning pest control ecosystem service to control moth pest species in UK apple orchards. I assess the ability of four types of farm management: organic, Linking Environment and Farming (LEAF), integrated pest management (IPM) and conventional, to measure the ability of pest predation from birds, and the impact that predation has on apple yields.

I firstly describe the history and the landscape of the study area, an overview of the methods used and the farming systems that the field study and experiments took place on in Chapter 2. In Chapter 3 I assess farmland biodiversity by monitoring birds and butterflies as indicator species of biodiversity, to understand if farm management impacts biodiversity levels. Biodiversity was highest on organic orchards, which supports the plethora of studies in the literature. Using this information of biodiversity levels on orchard management types, in Chapter 4 I investigate whether this biodiversity supports a pest control service, and to a natural pest control service compares to a synthetic alternate used on non-organic orchards, through using a sentinel prey experiment in field. Pest control services were greater on organic farms, and followed the same patterns as insectivorous bird abundance, species richness, diversity, and density. This chapter also compares moth pest levels to understand the pest pressures across farms, which harbour different pest control strategies and showed that moth pest levels were broadly similar across all farm management types. Finally, in Chapter 5 I compare the farm management options available to famers, both the natural pest control system and the synthetic control system, using economic valuation methods. Although a natural pest control service from birds is present on organic orchards (Chapter 4), the yield per hectare increased significantly on non-organic orchards (expect LEAF) but is found to be in-different to yield value per hectare of organic orchards in variable scenarios. Importantly, the synthetic alternative to a pest control service available from wild insectivorous birds was found to be an insignificant farm management variable that impacts apple yield and yield value on non-organic orchards.

Overall, I conclude that biodiversity can support a viable pest control ecosystem service from wild insectivorous birds on organic orchards, to a level comparable to the yield value of non-organic orchards that use high amounts of synthetic alternatives. There is a natural pest control value of birds to farmers on organic orchards that is not available on non-organic orchards.

Impact Statement

The insight discovered from this thesis is beneficial to the academic community as well as farming and business communities. There are two main discoveries: that a pest control service from birds on organic orchards *does* exist; and the value of this service per hectare is comparable to the synthetic alternative when increased insecticides are used on non-organic farms.

Through disseminating this information at orchard events in the UK and sharing in-depth information to the farmers involved in this study, it will give farmers an insight into the benefits organic farming has to biodiversity and the provision of a natural pest control service, in comparison to non-organic farming. Although non-organic farming does provide higher cider apple yields per hectare compared to organic - other than LEAF - yield *value* is the same as non-organic. I give here evidence that organic farming can provide a biodiversity-friendly farm management that can be as profitable per hectare as non-organic farming when insecticide use is high.

In the face of environmental pressures and pest resistance to chemicals, organic farming is a resilient system in which natural ecosystem processes are harboured, whilst providing stable yield value for organic growers in comparison to the more volatile markets of chemical insecticide use and reliance. Similar yield values on organic orchards to non-organic enables organic farmers to benefit from a more environmentally friendly way of farming, creating a landscape with less reliance on insecticides. Continued financial support for organic farming is essential to ensure organic farming continues to prosper and paths the way for insecticide-dependant farms to consider lower insecticide use that enhances yield value per hectare.

Disseminating insights from this thesis will be by means of scholarly journal articles, presentations within farming and the organic community, as well as collaborating with business such as my collaboration with PepsiCo; to encourage primary producers within global supply chains to farm in a way that enhances biodiversity and ecosystem services: with lower use of insecticides. The impact of such mass scale transition to organic-inspired farming will support global biodiversity and species worldwide that are currently threatened by the increasing intensification of agriculture.

Acknowledgments

I would firstly like to sincerely thank my supervisors: Ken Norris, for being so supportive throughout my PhD, especially pulling me past the finish line in the final year, I honestly don't think I could have completed this without your guidance, humour and encouragement; and the late Dame Georgina Mace for extra support and welcoming me into the CBER research group, especially after transitioning from Reading to UCL, and for always being so optimistic and supportive.

Secondly, I wish to send my gratitude to the 30 growers in my study who, without their support for my research, I would not have a study. Each grower was willing to give personal information, allow me to access farms and allow bird and butterfly surveys to take place, so thank you all. A thank you to all my fieldworkers who helped collect data on butterflies, birds, and sentinel prey – you made fieldwork less lonely and helped me cope with the extent of the study area of focus.

I would like to also thank everyone who dedicated their time and expertise to my research, particularly Fiona Spooner, who helped me hugely with finding my way around GIS and R.

I wish to express my heartfelt thanks to my closest friends and family around me, who put up with my lack of social ability and visits, who supported me and kept encouraging through extremely difficult times and who were always willing me to continue, as they knew it meant so much to me – it must have been tiring to watch the process, so thank you.

Finally, I would like to thank my husband, Ross Miller. You have been there throughout the entire process, felt the struggles as much as I have, managed to support me tirelessly throughout. I simply could not have done it without your motivation and encouragement.

Contents

List of Figures

List of Tables

Attribution

During this PhD I received advice on my research direction from my Supervisors Prof Ken Norris and the late Dame Georgina Mace. I received statistical analysis advice from Prof Ken Norris, Dr. Xavier Harrison and Fiona Spooner for general GIS and R assistance.

CHAPTER 1

Introduction

1.1 OVERVIEW

1.1.1 Biodiversity loss, food production and growing demands

The loss of biodiversity due to human-driven agricultural expansion, logging and development has caused major concerns for species declines, with extinction rates increased to similar levels of the last five global mass-extinction events over the last 500 million years (Barnosky et al., 2011; Tilman et al., 2017). Agricultural practices have been reported to occupy 40% of Earth's land surface (Garfinkel & Johnson, 2015; Mclaughlin, 2011), with even higher percentages in Europe, where the UK classifies 71% of its land as agriculture (DEFRA, 2014). Furthermore, it is estimated that 40% of the land that has already been converted to agriculture is predicted to transform low into high intensity agriculture (Munang et al., 2010; TEEB, 2008).

Pressures to increase agricultural production are further heightened by the global human population increase, set to reach nine billion by 2050 (UN, 2010), alongside higher food consumption (Cumming et al., 2014) and demand for a meat-rich diet, which both correlate with wealth (Tilman et al., 2001; 2002) as well as signifincaly increasing green house gases (Tilman & Clark, 2014). This growing and affluent population requires a land-greedy system. Approximately 30-50% of cereal production is used to feed livestock (FAO, 2006; Tscharntke et al., 2012), rather than the growing human population. With this in mind, under a business-as-usual model, species populations are expected to continue their decline as a further 11% of natural habitats are estimated to either be converted to agriculture or lost to infrastructure or climate change (TEEB, 2008).

Agriculture intensification aims to produce higher yields per unit, without the need for agriculture expansion by means of increasing chemical inputs, mechanisation of agricultural practises and innovations in crop developments such as insecticides usage, tilling and improving crop varieties (Bommarco et al., 2013; Tilman et al., 2011). However, the degradation of environmental conditions and biodiversity through agricultural intensification has been evident globally (Foley et al., 2005; Bommarco et al., 2013), with negative impacts documented to ecosystem services and their providers such as pollinators

(Potts et al., 2010); farmland birds in Europe and the UK (Donald et al., 2001; Butler et al., 2007), with more intense threats to wild bird populations in developing countries (Green et al., 2005). Similar declines in response to intensification to farming practices through increased pesticide use, tillage and reduced crop varieties, have also been reported to arthropods (Puech et al., 2014; Benton et al., 2002) and butterflies (Wilson et al., 1999).

Farmland bird populations have been declining in most European countries since 1970, due to the increase in cereal production, with intensification of farming practices and reductions in suitable bird habitat and food availability (Donald et al., 2001). From 1979 – 1999 British farmland birds declined dramatically; ten million breeding birds from ten farmland bird species disappeared from farmland in 20 years (Krebs et al., 1999), largely due to lack of invertebrates due to increased insecticide use (Fuller, 2000). Furthermore, long term historical studies have shown the positive correlation between bird numbers and farmland arthropod numbers, both negatively affected by farming intensity (Benton et al., 2002). Similar declines in populations of bees (Kremen et al., 2002), arthropods and flowering plants (Sotherton & Self, 2000) are documented due to agricultural intensification. The diversity of butterflies decreases due to unstable habitat areas (Steffan-dewenter & Tscharntke, 2000) and butterfly abundance declined 30% in a 16 year period in The Netherlands (Van Dyck et al., 2009). The conservation of biodiversity should not be limited to the 5% of remaining pristine habitat, instead, include these agricultural lands to assist in global targets of improving the status of biodiversity worldwide (Tscharntke et al., 2005; CBD, 2018).

The need for more sustainable farming methods to intensify food production has been coined "sustainable intensification", where increasing yields, without agricultural expansion, can continue whilst reducing environmental impacts and degradation (Godfray et al., 2010; Royal Society, 2009). A variety of farming methods, such as organic and integrated pest management (IPM) were developed (Stern, et al., 1959) with the aim to reduce the use of chemical pesticides in farming practices and protect agro-ecosystems and ecosystem services that support it (EC, 2017; EC, 2018).

1.1.2 Biodiversity and ecosystem services

Biodiversity is complex so the definition should reflect the multiple roles biodiversity has in relation to ecosystem services. Biodiversity is referred to as an ecosystem service itself, a good, and sometimes as a regulating process to provide other ecosystem services (Mace et al., 2012). Mace et al. (2012) advises, rather than try to re-define biodiversity, to use the overarching definition set out by the Convention on Biological Diversity:

"the variability among living organisms from all sources including, inter alia, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems" (CBD, 2018).

The Millennium Ecosystem Assessment (MA) identifies the support that biodiversity gives to ecosystem services of which society receives, including: the origins and yields of pharmaceuticals from ecosystems rich in species diversity, such as the Tropics (MA 2005a); regulating ecosystem services such as pollination, seed dispersal and carbon storage (MA 2005b); wild crop diversity for continued improvements in crop modifications (Jenkins et al., 2004) and resilience against agricultural pests (Mace et al., 2012; Cardinale et al., 2003).

The links between biodiversity and ecosystem services have been investigated to understand how our ecosystems function and provide the services we rely on. There is ample evidence suggesting that, directly or indirectly, biodiversity is correlated with, or has influences on, ecosystem functions and therefore affecting the services we receive (Cardinale et al., 2012; Isbell et al., 2011). Cardinale et al. (2006) provide a detailed analysis of 111 studies that manipulated biodiversity to identify the effects on ecosystems. Their conclusions found a correlation between species loss and loss of ecosystem functioning but the particular species supporting functioning or ecosystem services is unknown, thus a "precautionary approach" is suggested to "preserve as much biodiversity as possible" (Cardinale et al., 2006: 991). Newbold et al. (2013) suggest this link between biodiversity and the ecosystem functions and services should be investigated at local and landscape scales, such as catchment areas, rather than global. Working at country-level is an obvious scale to understand what biodiversity a country has, how much is needed to support human population, how much needs to be protected to support that population, and what policies should be implemented to do this. But to safeguard biodiversity, that supports ecosystem services important to people and agriculture, local action is necessary.

The ecosystem services approach has started to build momentum in highlighting the benefits we receive from nature, not just through the production of food but regulating services such as pollination, pest regulation and supporting services such as soil formation and nutrient cycling (MA, 2005b). However, Swift et al. (2004) discuss whether more localised ecosystem functions are jeopardised, especially in relation to agro-ecosystems, when plot diversity is reduced. Often, under experimental conditions, only a single species is necessary in delivering a specific ecosystem function (Swift et al., 2004). However, when the resilience of an ecosystem is threatened by climate change or invasive species, then higher biodiversity levels may allow ecological niches to be filled to continue to support ecosystem functions (Swift et al., 2004; Vandermeer et al., 1998).

The current risks to natural pest control ecosystem services are high and highlighted by the increased number of insecticides being used in the agricultural industry, demonstrating the lack of this service in these agro-ecosystems. Pesticides are responsible for loss of insect biodiversity and thus reduction in number of invertebrates and natural enemies in the landscape (Power, 2010; Stoate et al., 2009), for birds to predate upon. If broad spectrum insecticides are used this can have detrimental effects on non-target natural enemies (The National Research Council, 2010), including birds (Krebs et al., 1999). Although pesticide usage has increased since the 1960s, the percentage of crop loses have continued to increase in the last 40 years (Oerke, 2006). Furthermore, uniform monocultures are still highly susceptible to disease and insect attacks, due to disruption of biological pest control services through increasing pesticide use (Thrupp, 2000; Tscharntke et al., 2005).

1.1.3 Valuing ecosystem services

The economic valuation of biodiversity has been identified as an "important tool to illustrate the link between nature and human welfare" (Paul et al., 2020). It is used within ecology to increase the awareness of biodiversity in decision-making processes.

In environmental economics, ecosystem services are goods, from nature, which people value, that increase human welfare (Bateman et al., 2011). As biodiversity is now recognised as an essential but diminishing resource, it fits the definition of economics "the efficient use of scarce resources" lending itself to being analysed economically (Edwards & Abivardi, 1998: 240). The difference between *value* and *price* is important to distinguish, as what something is priced at may not reflect its value to society or to an individual. Bateman et al. (2010) give an insightful example of the value of walking through a woodland park, although the price of entering may be zero it is not of zero value, as many people choose to spend time there.

Silvertown (2015), outlines that the decision to value an ecosystem service is still in debate between conservationists, with one side announcing there is no choice but to value (Costanza et al., 1997), whilst others urge the pragmatic decision to value or not to value to be taken with caution (Kallis, Gómez-Baggethun, & Zografos, 2013). Often what we value becomes visible but conservationists, policy makers, and academics that oppose valuation believe that what is valued and quantified ultimately becomes 'for sale' (Hungate & Cardinale, 2017). It is important to consider the concerns surrounding the valuation of nature; there is a risk of authorities and institutions being given permission to control nature for the benefit of humans, rather than an authentic recognition that nature, itself, is valuable and priceless (Foster, 1997; McCauley, 2006). Whether nature should be valued or not is still debated between environmentalists today (Norton, 2017). Redford & Adams (2009) set out seven arguments against the valuation of ecosystem services, which highlight multiple scenarios where the value of ecosystems and nature may be undermined by economic theory, human demand, changing markets and the development of engineered ecosystem services that could undervalue intact ecosystems in favour of the most popular services, where the economic justification "outweighs non-economic justification for [biodiversity] conservation", that serves humans only in the short term (Redford & Adams, 2009: 785). Redford & Adams also make a valid statement that not all ecosystem services are of service in the short term, such as forest fires and floods, but are vital as part of global ecosystem processes yet cause ill-health and death. If these were to be incorporated into ecosystem service valuation, we could completely undervalue nature. An example of undervaluation comes from the valuation of native wild bees using forest fragments, which gave pollination services to coffee plantations in Costa Rica (Ricketts et al., 2004). The study calculated native bees to be worth $62,000 per year, however following the study the farm suffered from a crash in coffee prices, so they decided to plant pineapples. Pineapples do not rely on pollination, so the decision to expand agriculture into surrounding forest fragments was made easier, as the value of the forest fragments reduced by $62,000 when the farm crop changed from a pollinator dependant crop to a non-pollinated crop (McCaugley, 2006). Converting the remaining, low-valued, forest fragments to a more profitable crop is an easy economic decision for farmers to make (Silvertown, 2015). McCauley (2006) argues that if we speak to people's hearts, rather than their wallets, it may benefit longer-term nature conservation.

However, using only intrinsic values, such as protected areas - a traditional conservation practice for conserving biodiversity without holding an economic value – also has its pitfalls. Illegal hunting and agricultural practice expansions into protected areas cause further species population declines to levels that, if continued at the same rate, would increase the already high extinction rates (Packer et al., 2013; Tilman et al., 2017). Although the extent of protected areas and sustainable forests are increasing, these efforts have been inadequate on their own at halting biodiversity loss (Butchart et al., 2010).

To strengthen the protection of biodiversity, economic value - made clear in policy and decision making – could make up part of an imperative toolkit, alongside increased funding for habitat protection and integration of biodiversity into land-use planning and development decisions (Butchart et al., 2010; Wenny et al., 2011). A multi-disciplinary approach uses the collaboration of economists and ecologists to both explore outside the usual limits of each discipline in order to try to represent complex ecosystem functioning, and the services they provide, in terms of economic value (Barbier & Heal, 2005). Using a variety of methods to protect biodiversity is necessary for different locations and situations.

To value goods where market prices are non-existent or are "imperfect reflections" of market prices, a variety of valuation methods have been developed by economics to value ecosystem services (Bateman et al., 2010). Bateman et al. (2011) give an in-depth review on economic valuation methods available, including: Bateman, 2007; Bateman et al., 2002; Freeman, 2003; Heal et al., 2005; Pagiola et al., 2004. However the description in Barbier, (2007) synthesises Freeman (2003), Heal et al. (2005), and Pagiola et al. (2004) to describe each valuation method, how they are applied and which ecosystem services are valued through them (Barbier, 2007: 186, Table 2). Further reviews on valuation case studies, and their decisions to value certain ecosystem services in the interest of biodiversity, have been scrutinised in Kallis et al. (2013).

To further simplify Barbier (2007), in general, valuation methods can be split into three categories, according to the Joint Nature Conservation Committee (JNCC, 2013):

1. Stated preference methods
2. Revealed preference methods
3. Cost based approaches

These are split into direct (1) and indirect valuations (2 & 3), where indirect approaches do not rely on the answers to questionnaires or answers to questions about how much they would be willing to pay/accept for an environmental change, instead the indirect technique reveals the value of a service by relating the ecosystem service to a marketable good that the individual purchases (Pearce & Moran, 1994).

Stated preference methods provide surveys to ask individuals hypothetical questions to assess how people make choices between levels of environmental goods at different prices. This direct valuation approach reveals the extent of an individual's willingness to pay for certain ecological benefits and environmental goods (Bateman et al., 2011; JNCC, 2013; Pearce & Moran, 1994). Contingent valuation and choice modelling are the valuation methods to choose from within stated preference. Both are willingness-to-pay, survey-style methods, the former based on environmental change and the latter based on combination of attributes that are within environmental change where the participant ranks the combinations (JNCC, 2013). However, stated preference methods are based upon the general public opinion who may not be informed about biodiversity to make a decision (Hougner, Colding & Soderqvist, 2006).

Revealed preference methods use indirect valuation (Pearce & Moran, 1994), by providing real choices for people to understand what value they put on a particular ecosystem service. Examples of these methods include: hedonic pricing – where environmental characteristics are reflected in property value, used often within cities (Gómez-Baggethun & Barton, 2013), and travel cost methods – where the cost in travel to reach a recreational site becomes the proxy for the value of the site (JNCC, 2013; Bateman et al., 2010). However, often revealed preference is not practical as most species do not have a known market price (Daniels et al., 2017).

Cost based approaches are used to value ecosystem services that have either been part of the production or costs based on how much it would cost to replaced or restore them (JNCC, 2013; Pearce & Moran, 1994). For example:

- production function costs – isolates the effect of an ecosystem service to understand its contribution to overall production, usually demonstrated in regulating services of agriculture, where the ecosystem service can be indirectly inferred from the price of the market products (crops) (Barbier, 2007; Bateman et al., 2011; Freeman, 2003; Zhang & Swinton, 2012);
- replacement costs method, which evaluates the costs incurred by replacing an ecosystem service, if totally lost, to its original state, such as pollination by calculating hand pollination costs (Allsopp, de Lange, & Veldtman, 2008) or pest control through the use of insecticides (Power, 2010); and

- damage costs avoided – which calculates the costs avoided by not letting ecosystem services degrade, usually in respect to flood and storm protection (Bateman et al., 2011).

Replacement costs are often used in relation to ecosystem service valuation, especially regulating ecosystem services such as pollination, water purification and pest control. A good example of pollination valuation is by Allsopp (2008), who valued insect pollination with replacement costs of four different methods of hand pollination and pollen dusting techniques in the Western Cape, South Africa. Allsopp found that hand pollination costs are so high that it would lead to a market failure in the grape industry if local landscapes degraded enough to not supporting pollinators and the inputs of this industry would far outweigh the profitability, impacting the international competitiveness of the industry. Furthermore, different methods of replacement were valued very differently, depending on the cost of each replacement method, which shows how assumptions used during valuations strongly influence the end value (Allsopp et al., 2008).

However, many economists, including Bateman et al., (2011), Barbier (2007), Freeman (2003) and Heal et al. (2005) believe that replacement costs are not suitable for valuation of ecosystem services and should be used with caution, because the costs of replacement may not resemble the value they try to approximate. Replacement value methods often overestimate the value of a service because usually an output price (i.e., yield) is not included during analysis, where yield would increase with production costs, so costs incurred through the replacement is spread over larger units of output (Letourneau et al., 2015).

Although the above methods of valuation exist, and an increasing amount of papers discuss the value of ecosystems and biodiversity, few give quantitative estimates of the value of nature and out of that few the models are not convincing to be incorporated into policy or farm management (Hungate & Cardinale, 2017).

There is a need to simplify the valuation approaches outlined above through focusing on the final ecosystem services, rather than confusing ecosystem processes for ecosystem services, is a vital concept within the ecosystem service valuation frameworks that needs re-thinking to include the rational that to produce a final ecosystem service, like crop yield, human and natural services must work together to make them beneficial for human wellbeing (Bengtsson, 2015; Lele et al., 2013). This has not been addressed enough within academic literature, to date.

1.1.4 Business and biodiversity

Businesses can impact biodiversity through climate change induced by rising carbon dioxide levels (Vitousek et al., 1997) and extractive processes, such as mining, logging and agriculture (Tscharntke et al., 2005; Wood et al., 2000). However, it is less clear what dependencies businesses have on biodiversity and how risks can be minimised through identification of those dependencies. Although research highlights negative impacts caused by agriculture, the ways in which businesses can interact with their supply chains to minimise impacts has been less explored. Dyllick and Hockerts (2002) highlight this as an environmental sustainability issue but also an economic sustainability issue, which requires a business to meet the needs of current, and future, investors through sustainably managing resources.

Multi-national corporations have started to focus efforts on sustainable farming, creating global sustainability targets to reduce reputational damage and improve long-term contracts, relationships, and environmental sustainability throughout their supply chain, such as PepsiCo's sustainable farming initiative (Pepsico, 2010). Without basic ecosystem functions, increased synthetic inputs such as fertilisers, irrigation systems, and pesticides are required. Ultimately, biodiversity loss to agricultural land may not be sustainable; where increased pest resistance to chemicals present cases of decreased agricultural resilience (Luck et al., 2009; Lewis et al., 1997).

Little is known about the influence of biodiversity loss on yield and therefore profit or yield value (Chapter 5), and as the influence biodiversity has on ecosystem services that support crops is not highlighted as important within short-term five 'business' years, it is often overlooked by more pressing or tangible targets such as reducing water usage and carbon emissions (Pepsico, 2013).

This project focuses on gaps in the academic literature to answer questions, not only posed by academics and policy makers, but multinational corporations. Using PepsiCo as a case study and Copella apple juice as a case brand within my study area, I present here new research that investigates different farm management options available, their influence on biodiversity and ecosystem services associated with it and compare the value of those ecosystem services between farming options, to assist decision makers at both the farm, policy, and corporation level.

1.2 THESIS OVERVIEW

Biodiversity loss across the globe is not halting and the demands for providing more food for a growing population without reducing biodiversity has been a popular and important research field for decades. Yet, organic farming is declining in the UK, which is a known farming method that reduces biodiversity loss. Ecosystem services have been a biological tool that can highlight the benefits we receive from biodiversity. Although biodiversity is greater on organic orchards, the functional consequences of this are not well known. In this thesis, I focus on how a valuation method could better inform farmers to make economic and environmental decisions about farm management options that could both serve them financially and conserve biodiversity.

During this study I look at the provisioning of a pest control ecosystem service from birds on apple orchards and compare this service's value on organic orchards to the synthetic alternative used on non-organic farms. Apple orchards were chosen as the system based on ecological and practical reasoning. Ecologically, apple orchards are both varied in their management and are known to support wildlife and biodiversity on low-intensity, traditional orchards. Practically, the basis of this study was to use a PepsiCo supply chain, Copella apple juice, to bring relevance to large multinational businesses in their quest to understand the value of biodiversity.

In **Chapter 2** I give an overview of the study area's horticultural history and landscape, as well as an introduction to the methods used throughout the thesis. This background information about the study sites and study farms forms the basis of all data chapters that follow.

In **Chapter 3** I assess the biodiversity on four farm management types to understand the baseline levels that exist on farms, to use this information for later chapters. Birds were used as the primary biodiversity indicator species and results showed organic orchards hosted the highest biodiversity than other farm management types in the study.

In **Chapter 4** I use information from Chapter 3 to understand the level of pest control services biodiversity can provide. Sentinel prey experiments and pest moth abundance methods were undertaken to provide results that link biodiversity with increased pest control service from the functional group: insectivorous birds. Pest moth abundance was broadly similar across all farm types, which suggest two types of farming systems are at play: one where pests are controlled by a natural pest control system, and one controlled by a synthetic alternative.

In **Chapter 5** I compare these two types of farming systems, a natural pest control system of organic farms and one that relies on synthetic alternatives, in terms of their ability to provide the final ecosystem service of apple yield and yield value to farmers. Yield per tree and per hectare is assessed in each farm system and insecticide volumes and apple prices for non-organic and organic markets was used to decipher the yield value per hectare to farmers. Per tree, organic yields were the same or higher than non-organic farms and yield value was higher on organic farms than all non-organic. However, yield per hectare is higher on conventional and IPM farms in comparison to organic, but not to LEAF. Insecticide use changes the relation with yield value, the more insecticides used the lower the yield value on non-organic orchards, showing that the synthetic alternative to natural pest control is more volatile than organic farms at producing yield value.

I conclude the thesis with an overarching summary in **Chapter 6** where I discuss the implications of findings from each chapter and the relevance of the results to both the farming, academic and business communities. I discuss limitations experienced and directions for future research on the comparisons of a natural pest control service supported by biodiversity to the use of a man-made replacement. This research will provide significant contributions to our understanding of the value of biodiversity, filling in gaps between general biodiversity differences on farming options, the functioning ecosystem services provided and net yield profits to overall inform decision making in policy, business and at farm level.

1.3 REFERENCES

Allsopp, M. H., de Lange, W. J., & Veldtman, R. (2008). Valuing insect pollination services with cost of replacement. *PLoS ONE*, *3*(September). http://doi.org/10.1371/journal.pone.0003128

Barbier, E. . (2007). Valuing ecosystem services as productive inputs. *Economic Policy*, 177–299.

Barbier, E. B., & Heal, G. M. (2005). *Valuing Ecosystem Services*. *The Economists' Voice*.

Barnosky, A. D., Matzke, N., Tomiya, S., Wogan, G. O. U., Swartz, B., Quental, T. B., … Ferrer, E. A. (2011). Has the Earth's sixth mass extinction already arrived? *Nature*, *471*(7336), 51–57. http://doi.org/10.1038/nature09678

Bateman, I. J. (2007). Valuing preferences regarding environmental change. In J. Pretty, A. Ball, T. Benton, J. Guivant, D. Lee, D. Orr, … H. Ward (Eds.), *The SAGE handbook of environment and society* (pp. 155–171). London: Sage Publications.

Bateman, I. J., Carson, R. T., Day, B., Hanemann, W. M., Hanley, N., Hett, T., … Swanson, J. (2002). *Economic valuation with stated preference techniques: a manual*. Cheltenham: Edward Elgar Publishing.

Bateman, I. J., Mace, G. M., Fezzi, C., Atkinson, G., & Turner, K. (2010). Economic Analysis for Ecosystem Service Assessments. *Environmental and Resource Economics*, *48*(2), 177–218. http://doi.org/10.1007/s10640-010-9418-x

Bateman, I. J., Mace, G. M., Fezzi, C., Atkinson, G., & Turner, K. (2011). Economic analysis for ecosystem service assessments. *Environmental and Resource Economics*, *48*(2), 177–218. http://doi.org/10.1007/s10640-010-9418-x

Bengtsson, J. (2015). Biological control as an ecosystem service: partitioning contributions of nature and human inputs to yield. *Ecological Entomologymology*, *40*, 45–55. https://doi.org/10.1111/een.12247

Benton, T. I. M. G., Bryant, D. M., Cole, L., & Crick, H. Q. P. (2002). Linking agricultural practice to insect and bird populations: a historical study over three decades. *Journal of Applied Ecology*, *39*, 673–687.

Bommarco, R., Kleijn, D., & Potts, S. G. (2013). Ecological intensification: harnessing ecosystem services for food security. *Trends in Ecology & Evolution*, *28*(4), 230–8. http://doi.org/10.1016/j.tree.2012.10.012

Butchart, S. H. M., Walpole, M., Collen, B., van Strien, A., Scharlemann, J. P. W., Almond, R. E. a, … Watson, R. (2010). Global biodiversity: indicators of recent declines. *Science*

(New York, N.Y.), *328*(5982), 1164–8. http://doi.org/10.1126/science.1187512

Butler, S. J., Vickery, J. A., & Norris, K. (2007). Farmland Biodiversity and the Footprint of Agriculture. *Science*, *315*(January), 381–384.

Cardinale, B. J., Duffy, J. E., Gonzalez, A., Hooper, D. U., Perrings, C., Venail, P., … Naeem, S. (2012). Biodiversity loss and its impact on humanity. *Nature*, *486*(7401), 59–67. http://doi.org/10.1038/nature11148

Cardinale, B. J., Harvey, C. T., Gross, K., & Ives, A. R. (2003). Biodiversity and biocontrol: emergent impacts of a multi-enemy assemblage on pest suppression and crop yield in an agroecosystem. *Ecology Letters*, *6*(9), 857–865. http://doi.org/10.1046/j.1461-0248.2003.00508.x

Cardinale, B. J., Srivastava, D. S., Duffy, J. E., Wright, J. P., Downing, A. L., Sankaran, M., & Jouseau, C. (2006). Effects of biodiversity on the functioning of trophic groups and ecosystems. *Nature*, *443*(7114), 989–92. http://doi.org/10.1038/nature05202

CBD. (2018a). Aichi Biodiversity Targets. Retrieved January 18, 2018, from https://www.cbd.int/sp/targets/

CBD. (2018b). Article 2: Use of Terms. Retrieved July 5, 2018, from https://www.cbd.int/convention/articles/default.shtml?a=cbd-02

Commission, E. (2018). What is organic Farming? Retrieved July 4, 2018, from https://ec.europa.eu/agriculture/organic/organic-farming/what-is-organic-farming_en

Costanza, R., D’Arge, R., Groot, R. d., Farber, S., Grasso, M., Hannon, B., … van den Belt, M. (1997). The value of the world’s ecosystem services and natural capital. *Nature*, *387*(15th May), 253–260. http://doi.org/10.15713/ins.mmj.3

Cumming, G. S., Buerkert, A., Hoffmann, E. M., Schlecht, E., von Cramon-Taubadel, S., & Tscharntke, T. (2014). Implications of agricultural transitions and urbanization for ecosystem services. *Nature*, *515*(7525), 50–57. http://doi.org/10.1038/nature13945

Daniels, S., Witters, N., Beliën, T., Vrancken, K., Vangronsveld, J., & Van Passel, S. (2017). Monetary Valuation of Natural Predators for Biological Pest Control in Pear Production. *Ecological Economics*, *134*, 160–173. http://doi.org/10.1016/j.ecolecon.2016.12.029

DEFRA. (2014). *Agriculture in the United Kingdom*. Retrieved from https://www.gov.uk/government/uploads/system/uploads/attachment_data/file/315103/auk-2013-29may14.pdf

Donald, P. F., Green, R. E., & Heath, M. F. (2001). Agricultural intensification and the collapse of Europe’s farmland bird populations. *Proceedings. Biological Sciences / The Royal*

Society, *268*(1462), 25–29. http://doi.org/10.1098/rspb.2000.1325

Dyllick, T., & Hockerts, K. (2002). Beyond the business case for corporate sustinablity. *Business Strategy and the Environment*, *11*, 130–141.

EC. (2017). Integrated Pest Management.

Edwards, P. J., & Abivardi, C. (1998). The Value of Biodiversity: Where Ecology and Economy blend. *Biological Conservation*, *83*(3), 239–246.

FAO. (2006). *World agriculture: towards 2030 / 2050 - Interim Report*. Rome, Italy.

Foley, J. A., DeFries, R., Asner, G. P., Barford, C., Bonan, G., Carpenter, S. R., … Snyder, P. K. (2005). Global Consequences of Land Use. *Science*, *309*(5734), 570–574. http://doi.org/10.1126/science.1111772

Foster, J. (1997). *Valuing Nature? Economics, Ethics and Environment*. (J. Foster, Ed.). London and New York: Routledge.

Freeman III, A. M. (2003). *The measurement of environmental and resource values: theory and methods. In: Resources for the Future* (Second Ed). Washington, DC.

Fuller, R. J. (2000). *Relationships between recent changes in lowland British agriculture and farmland bird populations: an overview*. Thetford, Norfolk.

Garfinkel, M., & Johnson, M. (2015). Pest-removal services provided by birds on small organic farms in northern California. *Agriculture, Ecosystems & Environment*, *211*(2015), 24–31. http://doi.org/10.1016/j.agee.2015.04.023

Godfray, H. C. J., Beddington, J. R., Crute, I. R., Haddad, L., Lawrence, D., Muir, J. F., … Toulmin, C. (2010). Food Security: The challenge of Feeding 9 Billion people. *Science*, *327*(812). http://doi.org/10.1126/science.1185383

Gómez-Baggethun, E., & Barton, D. N. (2013). Classifying and valuing ecosystem services for urban planning. *Ecological Economics*, *86*(2013), 235–245. http://doi.org/10.1016/j.ecolecon.2012.08.019

Green, R. E., Cornell, S. J., Scharlemann, J. P. W., & Balmford, A. (2005). Farming and the Fate of Wild Nature. *Science*, *550*(307). http://doi.org/10.1126/science.1106049

Heal, G. M., Barbier, E. B., Boyle, K. J., Covich, A. P., Gloss, S. P., Hershner, C. H., … Shrader-Frechette, K. (2005). *Valuing ecosystem services: toward better environmental decision making.* Washington: The National Academies Press.

Hougner, C., Colding, J., & Soderqvist, T. (2006). Economic value of a seed dispersal service in the Stockholm National Urban Park. *Ecological Economics*, *59*, 364–374. http://doi.org/https://doi.org/10.1016/j.ecolecon.2005.11.007

Hungate, A. B., & Cardinale, J. B. (2017). Biodiversity: what value should we use ? *Frontiers in Ecology and the Environment*, *15*(6), 283. http://doi.org/10.1002/fee.1511

Isbell, F., Calcagno, V., Hector, A., Connolly, J., Harpole, W. S., Reich, P. B., … Loreau, M. (2011). High plant diversity is needed to maintain ecosystem services. *Nature*, *477*(7363), 199–202. http://doi.org/10.1038/nature10282

Jenkins, M., Scherr, S. J., & Inbar, M. (2004). Markets for Biodiversity Services: Potential roles and challenges. *Environment*, *46*(6), 32–42.

JNCC. (2013). Ecosystem Valuation. Retrieved July 26, 2018, from http://jncc.defra.gov.uk/page-6383-theme=textonly

Kallis, G., Gómez-Baggethun, E., & Zografos, C. (2013). To value or not to value? That is not the question. *Ecological Economics*, *94*(2013), 97–105. http://doi.org/10.1016/j.ecolecon.2013.07.002

Krebs, J. R., Wilson, J. D., Bradbury, R. B., & Siriwardena, G. M. (1999). The second Silent Spring? *Nature*, *400*, 611–612. http://doi.org/10.1038/23127

Kremen, C., Williams, N. M., & Thorp, R. W. (2002). Crop pollination from native bees at risk from agricultural intensification. *Proceedings of the National Academy of Sciences of the United States of America*, *99*(26), 16812–16816. http://doi.org/10.1073/pnas.262413599

Lele, S., Springate-Baginski, O., Lakerveld, R., Deb, D., & Dash, P. (2013). Ecosystem services: Origins, contributions, pitfalls, and alternatives. *Conservation and Society, 11*(4), 343–358. https://doi.org/10.4103/0972-4923.125752

Lewis, W. J., Lenteren, J. C. Van, Phatak, S. C., & Tumlinson, J. H. (1997). A total system approach to sustainable pest management. *Proceedings of the National Academy of Sciences of the United States of America*, *94*(November), 12243–12248.

Luck, G. W., Harrington, R., Harrison, P. A., Kremen, C., Berry, P. M., Bugter, R., … Zobel, M. (2009). Quantifying the Contribution of Organisms to the Provision of Ecosystem Services. *American Institute of Biological Sciences*, *59*(3), 223–235. http://doi.org/10.1025/bio.2009.59.3.7

Mace, G. M., Norris, K., & Fitter, A. H. (2012). Biodiversity and ecosystem services: a multilayered relationship. *Trends in Ecology & Evolution*, *27*(1), 19–26. http://doi.org/10.1016/j.tree.2011.08.006

McCauley, D. J. (2006). Selling out on nature. *Nature*, *443*(September), 27–28. http://doi.org/10.15713/ins.mmj.3

Mclaughlin, D. W. (2011). Land, Food, and Biodiversity. *Conservation Biology*, *25*(6), 1117–1120. http://doi.org/10.1111/j.1523-1739.2011.01768.x

Millenium Ecosystem Assessment. (2005a). *Ecosystems and Human Well-Being: Opportunities and challenges for Business and Industry*. Washington, DC.

Millenium Ecosystem Assessment. (2005b). *Millenium Ecosysetm Assesment - Biodiversity Regulation of Ecosystem Services*. Washington, DC.

Newbold, T., Scharlemann, J. P. W., Butchart, S. H. M., Sekercioglu, Ç. H., Alkemade, R., Booth, H., & Purves, D. W. (2013). Ecological traits affect the response of tropical forest bird species to land-use intensity. *Proc R Soc B*, *280*. http://doi.org/20122131

Norton, B. G. (2017). A Situational Understanding of Environmental Values and Evaluation. *Ecological Economics*, *138*, 242–248. http://doi.org/10.1016/j.ecolecon.2017.03.024

O'Brien, K. L., & Leichenko, R. M. (2000). Double exposure: Assessing the impacts of climate change within the context of economic globalization. *Global Environmental Change*, *10*, 221–232. http://doi.org/10.1016/S0959-3780(00)00021-2

Oerke, E.-C. (2006). Crop losses to pests. *The Journal of Agricultural Science*, *144,* 31–43. http://doi.org/10.1017/S0021859605005708

Packer, C., Loveridge, A., Canney, S., Caro, T., Garnett, S. T., Pfeifer, M., … Polasky, S. (2013). Conserving large carnivores: Dollars and fence. *Ecology Letters*, *16*(5), 635–641. http://doi.org/10.1111/ele.12091

Pagiola, S., von Ritter, K., & Bishop, J. (2004). *Assessing the Economic Value of Ecosystem Conservation. The World Bank Environment Department*. Retrieved from http://www.cbd.int/doc/case-studies/inc/cs-inc-iucn-nc-wb-en.pdf

Paul, C., Hanley, N., Meyer, S. T., Fürst, C., Weisser, W. W., & Knoke, T. (2020). On the functional relationship between biodiversity and economic value. *Science Advances, 6*(5). https://doi.org/10.1126/sciadv.aax7712

Pearce, D., & Moran, D. (1994). *The Economic Value of Biodiversity*. London: IUCN and Earthscan. Retrieved from http://books.google.com/books?id=RdH6DRZY0KIC

Pepsico. (2010). *Passionate about growing: PepsiCo UK sustainable farming report 2010*. UK & Ireland.

Pepsico. (2013). *Performance with Purpose. Sustainability Report 2013*. New York.

Potts, S. G., Biesmeijer, J. C., Kremen, C., Neumann, P., Schweiger, O., & Kunin, W. E. (2010). Global pollinator declines: trends, impacts and drivers. *Trends in Ecology & Evolution*, *25*(6), 345–53. http://doi.org/10.1016/j.tree.2010.01.007

Power, A. G. (2010). Ecosystem services and agriculture: tradeoffs and synergies. *Philosophical Transactions of the Royal Society of London. Series B, Biological Sciences*, *365*(1554), 2959–71. http://doi.org/10.1098/rstb.2010.0143

Puech, C., Baudry, J., Joannon, A., Poggi, S., & Aviron, S. (2014). organic vs. conventional farming dichotomy: Does it make sense for natural enemies? *Agriculture, Ecosystems & Environment, 194,* 48–57. http://doi.org/10.1016/j.agee.2014.05.002

Ricketts, T. H. H., Daily, G. C. C., Ehrlich, P. R. R., & Michener, C. D. D. (2004). Economic value of tropical forest to coffee production. *Proceedings of the National Academy of Sciences of the United States of America*, *101*(34), 12579–12582. http://doi.org/10.1073/pnas.0405147101

Robinson, R. A., & Sutherland, W. J. (2002). Post-war changes in arable farming and biodiversity in Great Britain. *Journal of Applied Ecology*, *39*, 157–176.

Royal Society. (2009). *Reaping the Benefits: Science and the sustainable intensification of global agriculture*. London. Retrieved from http://www.asbp.org.uk/uploads/documents/resources/Andrew Smith - CERAM ASBP 27th Feb.pdf

Silvertown, J. (2015). Have Ecosystem Services Been Oversold? *Trends in Ecology & Evolution*, *30*(11), 641–648. http://doi.org/10.1016/j.tree.2015.08.007

Sotherton, N. W., Self, M. J., Game, T., & Trust, C. (2000). *Changes in plant and arthropod biodiversity on lowland farmland : an overview*. *The Ecology and Conservation of Lowland Farmland Birds*.

Steffan-dewenter, I., & Tscharntke, T. (2000). Butterfly community structure in fragmented habitats. *Ecology Letters*, *3*, 449–456.

Stern, V. M., Smith, R. F., van den Bosch, R., & Hagen, K. S. (1959). The integration of chemical and biological control of the spotted alfalfa aphid: The integrated control concept. *Hilgardia*, *29*(2).

Stoate, C., Báldi, a, Beja, P., Boatman, N. D., Herzon, I., van Doorn, a, … Ramwell, C. (2009). Ecological impacts of early 21st century agricultural change in Europe--a review. *Journal of Environmental Management*, *91*(1), 22–46. http://doi.org/10.1016/j.jenvman.2009.07.005

Swift, M. J., Izac, a.-M. N., & van Noordwijk, M. (2004). Biodiversity and ecosystem services in agricultural landscapes—are we asking the right questions? *Agriculture, Ecosystems & Environment*, *104*(1), 113–134. http://doi.org/10.1016/j.agee.2004.01.013

TEEB. (2008). *The economics of ecosystems & biodiveristy: An interim report*. Cambridge, UK.

The National Research Council. (2010). *Strategic Planning for the Florida Citrus Industry:*

Addressing Citrus Greening Disease. Washington, D.C.: The National Academic Press. Retrieved from http://www.nap.edu/openbook.php?record_id=12880&page=96

Thrupp, L. A. (2000). Linking Agricultural Biodiversity and Food Security: The Valuable Role of Sustainable Agriculture. *Royal Institute of International Affairs*, *76*(2), 265–281.

Tilman, D., Balzer, C., Hill, J., & Befort, B. L. (2011). Global food demand and the sustainable intensification of agriculture. *PNAS*, *108*(50). http://doi.org/10.1073/pnas.1116437108

Tilman, D., & Clark, M. (2014). Global diets link environmental sustainability and human health. *Nature*, *515*(7528), 518–522. http://doi.org/10.1038/nature13959

Tilman, D., Clark, M., Williams, D. R., Kimmel, K., Polasky, S., & Packer, C. (2017). Future threats to biodiversity and pathways to their prevention. *Nature*, *546*(7656), 73–81. http://doi.org/10.1038/nature22900

Tilman, D., Fargione, J., Wolff, B., D'Antonio, C., Dobson, a, Howarth, R., … Swackhamer, D. (2001). Forecasting agriculturally driven global environmental change. *Science (New York, N.Y.)*, *292*(5515), 281–4. http://doi.org/10.1126/science.1057544

Tscharntke, T., Clough, Y., Wanger, T. C., Jackson, L., Motzke, I., Perfecto, I., … Whitbread, A. (2012). Global food security, biodiversity conservation and the future of agricultural intensification. *Biological Conservation*, *151*(1), 53–59. http://doi.org/10.1016/j.biocon.2012.01.068

Tscharntke, T., Klein, A. M., Kruess, A., Steffan-Dewenter, I., & Thies, C. (2005). Landscape perspectives on agricultural intensification and biodiversity - ecosystem service management. *Ecology Letters*, *8*(8), 857–874. http://doi.org/10.1111/j.1461-0248.2005.00782.x

UN. (2010). *World Population Prospects: The 2010 Revision. Vol. I: Comprehensive Tables*. New York.

Van Dyck, H., Van Strien, A. J., Maes, D., & Van Swaay, C. A. M. (2009). Declines in Common, Widespread Butterflies in a Landscape under Intense Human Use. *Conservation Biology*, *23*(4), 957–965. http://doi.org/10.1111/j.1523-1739.2009.01175.x

Vandermeer, J., van Noordwijk, M., Anderson, J., Ong, C., & Perfecto, I. (1998). Global change and multi-species agroecosystems: Concepts and issues. *Agriculture, Ecosystems & Environment*, *67*(1), 1–22. http://doi.org/10.1016/S0167-8809(97)00150-3

Vitousek, P. M., Mooney, H. A., Lubchenco, J., & Melillo, J. M. (1997). Human Domination of Earth's Ecosystems. *Science, 277*.

Wenny, D. G., DeVault, T. L., Johnson, M. D., Kelly, D., H. Sekercioglu, C., Tomback, D. F.,

& Whelan, C. J. (2011). The Need to Quantify Ecosystem Services Provided by Birds. *The Auk*, *128*(1), 1–14. http://doi.org/10.1525/auk.2011.10248

Wilson, J. D., Morris, A. J., Arroyo, B. E., Clark, S. C., & Bradbury, R. B. (1999). A review of the abundance and diversity of invertebrate and plant foods of granivorous birds in northern Europe in relation to agricultural change. *Agriculture, Ecosystems & Environment*, *75*(1–2), 13–30. http://doi.org/10.1016/S0167-8809(99)00064-X

Wood, A., Stedman-Edwards, P., & Mang, J. (2000). *The Root Causes of Biodiversity Loss*. Oxford, UK: Earthscan.

Zhang, W., & Swinton, S. M. (2012). Optimal control of soybean aphid in the presence of natural enemies and the implied value of their ecosystem services. *Journal of Environmental Management*, *96*(1), 7–16. http://doi.org/10.1016/j.jenvman.2011.10.008

CHAPTER 2

Methods

2.1 THE STUDY SYSTEM

Apple orchards were chosen as the system to test hypotheses set out in this thesis (Chapter 1), based on ecological and practical reasoning. Ecologically, apple orchards are both varied in their management and are known to support wildlife and biodiversity on low-intensity, traditional orchards (Natural England, 2010). Knowing their capacity to support biodiversity (Tuck et al., 2014), I was able to use organic orchards as a control group where no insecticides are used, and three increased-intensity management groups which used differing amounts of insecticides (IPM, conventional and LEAF). Practically, the basis of this study was to use a PepsiCo supply chain. The UK apple market is a major section of the supply chain for Copella, with approximately 29,000 tonnes of UK apples pressed to make Copella per year out of 53,000 needed (Pepsico, 2010). The rest of the apples come from a variety of locations including China, Poland, and Europe (PepsiCo, *Pers Comms,* 2014). With more than half of the primary production for Copella coming from the UK, a focus on biodiversity potential on UK apple orchards would inform a vast PepsiCo supply chain, with the potential to change PepsiCo farming practices in support of biodiversity.

This study took place in the county of Herefordshire and the surrounding bordering counties of Worcestershire and Gloucestershire, UK. The area is known as a primary apple and pear growing region of the country, previously also known for hops (Capper, R. 2015 *Personal Communication)*. Since the late 1600s apple orchards were common in Herefordshire's agricultural landscape and by 1644 the first pomology book was published by John Evelyn (Evelyn, 1664). The landscape characteristics of the study area fall under three National Character Area profiles, defined by Natural England as: South Herefordshire and over Severn, the Herefordshire lowlands, and the Herefordshire plateau (*figure 2.1*). Although the study sites span over three areas defined by different characteristics, all three area profiles are rural, with only Leominster, Hereford, Ross-on-Wye, and Newent included as larger settlements with small pressures on development. The three areas are dominated by well-wooded landscaped, including ancient woodlands and hedgerows with remnants of traditional orchards containing veteran trees important for UK wildlife, as well as nationally significant areas of lowland meadows, ancient parklands and permanent grasslands

(Natural England, 2013, 2014b, 2014a). The Herefordshire plateau is a strong hold for the rare noble chafer beetle and the golden eye lichen, re-discovered in 2007 (Natural England, 2014a). Fertile and high-grade Old Red Sandstone soils, with localised deposits of glacial drift, make for perfect agricultural soil in Herefordshire lowlands and South Herefordshire over Severn. This gives rise to more intensive agriculture, mainly mixed arable and fruit production (Natural England, 2013; 2014b), whereas the Herefordshire plateau has poorer shallow soils composed of Red Sandstone overlain with heavier loams and clays, making larger scale arable farming more dominant.

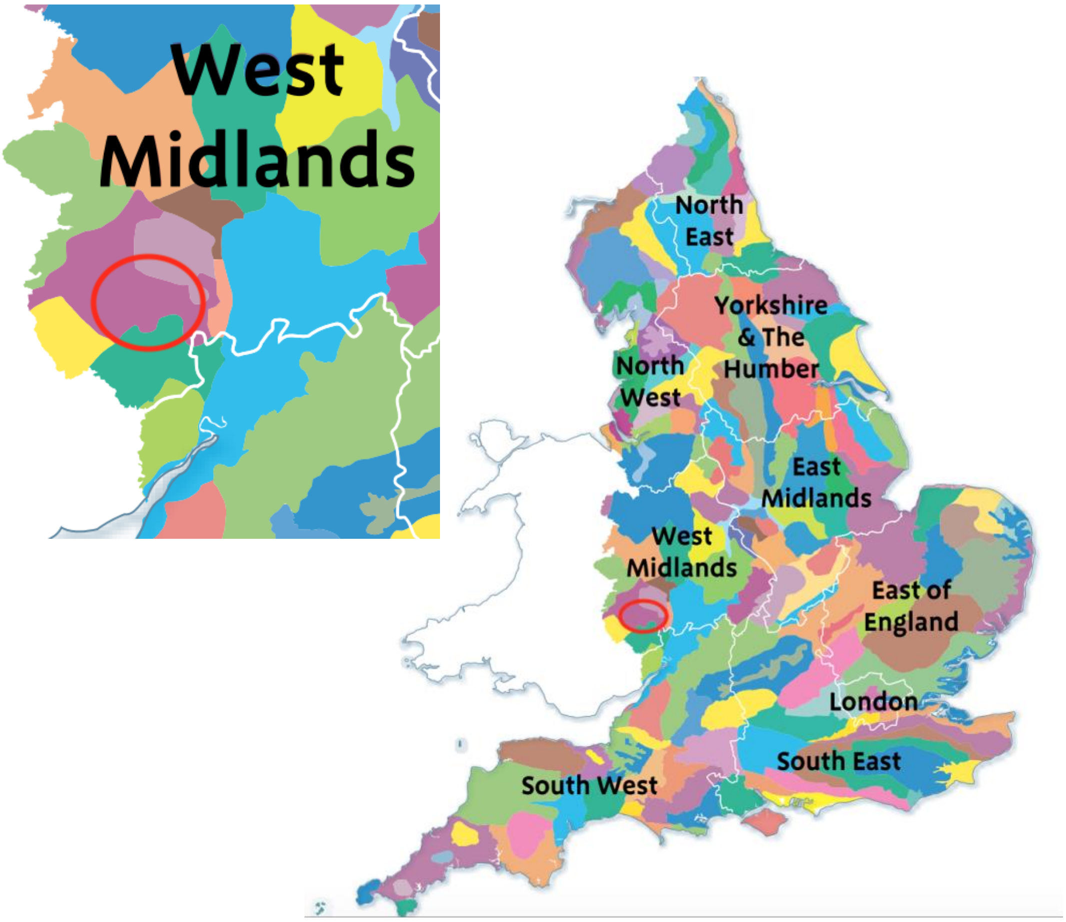


Figure 2.1 National character areas map (Natural England, 2013). The red circle illustrates the study area, which spans across the Herefordshire plateau (light pink), the Herefordshire lowlands (dark pink) and South Herefordshire over Severn (green).

This study was focussed on comparing types of farm managements where types of apples (dessert, cider, or juice) are managed differently, so connecting with a range of growers was essential. The area of Herefordshire had the second highest number of growers who grew fruit for Copella apple juice (a PepsiCo brand) but was the area with the highest LEAF growers who grew dessert apples and cider.

Farmers were contacted in late 2014, early 2015 to gauge interest in being involved as a study farm. I was kindly given a list of growers from LEAF Headquarters in Warwick and used their phone lines to contact LEAF Marque farms to ask if they would like to be a part of this study. To contact dessert growers who sold to Copella I was put in contact with an independent apple buyer who bought on behalf of PepsiCo. The buyer provided a list of growers in Herefordshire and four growers agreed to take part in the study. These growers were conventionally (3) and IPM (1) managed (explained in *section 2.2).* The remaining farms were contacted by word of mouth via the "snowballing" technique (Goodman, 1961). I found the contact details online for a handful of farmers via The Cider Route (2014), who pointed me in the direction to other growers who may be willing to take part. This continued until I found approximately equal numbers of farmers in each management category (*table 2.1*). The study started with 31 growers however, one farmer abandoned his farm after the first year, so this was taken out of the study.

All farms were located within a $60km^2$ area and the surrounding landscape of the study orchards was similar (*figure 2.2*). Some farms grew only one variety of apple whereas others grew multiple apple varieties and other crops, including plums, pears, and soft fruit. The orchard fields ranged between 0.6ha to 115ha in size. The management of each study orchard differed and were separated into four categories: conventional (7), IPM (7), LEAF (8) and organic (9).

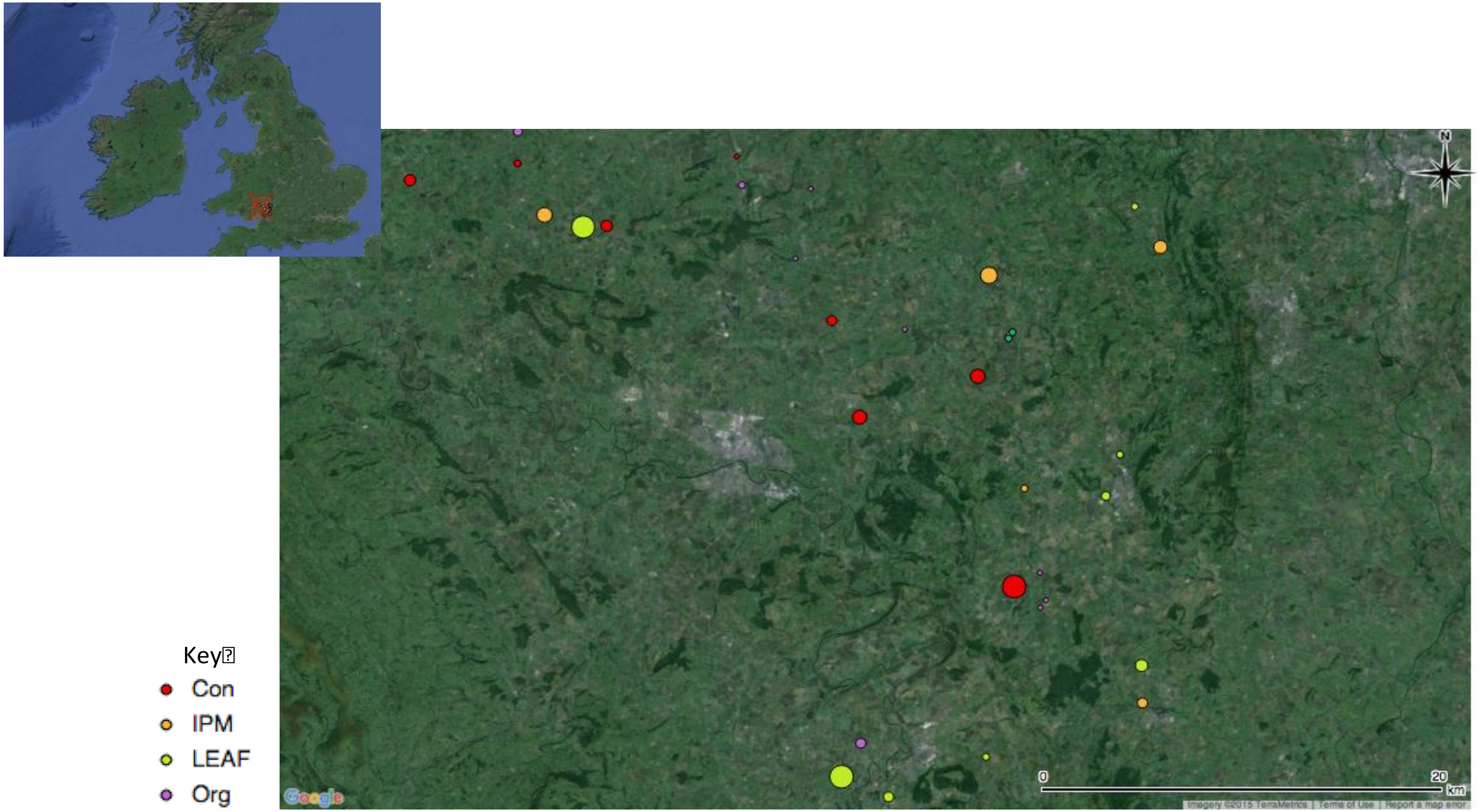


Figure 2.2 Field study sites and farm locations in Herefordshire and surrounding borders of Worcestershire, and Gloucestershire, UK. Each colour represents a different farm category and the larger the circle size, the larger the total orchard area.

2.2 FARMING SYSTEMS

In Europe, the key farming system that aims to support biodiversity and enhance ecosystem services, is through wildlife-friendly farming schemes, which can be government or market led (Hardman et al., 2016).

IPM is a European Commission approach (EC, 2017b), which advises against reliance on pesticides to control pests to combat the emergence of pesticide resistant species (Barzman et al., 2015). The aim of IPM is to use technologies that are deemed more sustainable and increase the use of bio-control alongside pest-specific insecticides (Birch, Begg & Squire, 2011). IPM advice the use of natural enemies and control timings of sprays to aim towards a low pesticide-input pest management (Hillocks, 2012), with guidelines from eight principles of IPM set out by the European Commission: (i) prevention and suppression, (ii) monitoring, (iii) decision making, (iv) non-chemical methods, (v) pesticide selection, (vi) reduced pesticide use, (vii) anti- resistant strategies, (viii) evaluation (Barzman et al., 2015). There are three levels of IPM that range from level 1 – including only timed sprays and pest thresholds to monitor pests before spraying, to level 3 which includes the reduction of broad spectrum insecticides, crop rotations, alternative bio-control methods, habitat management, and multi-crop interactions for optimise natural agro-ecosystem processes (Kogan, 1998). This study used mostly level 1 IPM farms, with only two that stated any management other than times sprays, which were natural enemy introduction.

The premise of IPM is to monitor pest levels through random field samples before applying chemical sprays, therefore it is important to understand that pest levels are not uniform across an agricultural field (Pearce & Zalucki, 2006), as farmers may overestimate or underestimate pests depending on the area surveyed. IPM principles arose in 2014 but are still not studied extensively, or holistically, in the literature where only one aspect of the IPM approach is usually focussed on (Stenberg, 2017).

LEAF farms are farms that have been certified as following the LEAF Marque standards. LEAF is a member of ISEAL, an independent body that accredits sustainable certification, and is one of six national associations under the European Initiative for Sustainable Development in Agriculture (EISA) (LEAF, 2016). LEAF have several "essential" practices their farmers must conform to, which form an Integrated Farm Management (IFM) approach that builds on the foundations of IPM. Some practices relate specifically to biodiversity, including the protection of woodland, hedgerows and coppices on the farm landscape and the provision of winter bird seed and habitats. One recommended practice is to obtain a minimum of 5% semi-natural habitat on site (LEAF, 2017).

The organic farm category in this study comprises of farms that are both Soil Association organic certified or farms that do not hold organic certification but use no chemical or copper inputs what-so-ever. Organic farms have a set of standards and principles to abide by to achieve Soil Association certification and enable the sale produce as certified organic. One of the general principles that is contested by zero-input farmers is that copper is permitted as an alternative to chemical fungicides (Soil Association, 2019: 9), which can persist in the soils and transferred into produce (Trewavas, 2001). The Co-FREE EU funded project (2016) was tasked with finding alternatives to copper use in organic farming. While it was found that copper cannot be phased out completely, especially on grapevines and fruit production, there were alternatives available including disease-resistance cultivars and alternative compounds from plant and microbial origins (European Commission, 2018). Organic farming is beneficial to species richness and abundance to multiple taxa (Bengtsson et al., 2005), and has been found to benefit biodiversity on a range of farming systems (Feber et al., 1997) from perennial systems in Japan (Katayama, 2016) to annual wheat growing farms in the UK (Hardman et al., 2016), across European and African regions (Schneider et al., 2014). A meta-analysis from Tuck et al. (2014) garnered data from 94 studies over the last 30 years and found a 30% increase in species richness on organic farms. The plethora of studies that support increased species richness on organic farms indicates that these findings are likely to be true for most organic farming compared to conventional.

Conventional farming practices are those based on pesticide advice from local agronomists without the use of natural enemies or pheromone traps. Due to demand for pristine apples with no trace of insect presence in supermarkets, conventional farming on dessert apples in the UK is intense. However, pest checks are advised by agronomists who visually check pest levels in the surrounding area and then advises farmers accordingly.

IPM and LEAF Marque farms have the same management techniques, using pesticides where necessary as well as natural enemies and pheromone traps but LEAF state, among other requirements for LEAF Marque certification, a minimum of 5% semi-natural habitat on site (LEAF, 2014). LEAF and IPM have been separated for analysis to understand if differences exist between certified and non-certified farms in terms of biodiversity.

Farming system type was assigned to individual farms through an initial farmer conversation and questionnaire (*Appendix A1*). Within the questionnaire farmers are asked if they are LEAF Marque farmers, trying to be LEAF Marque, organic certified through the Soil Association, or conduct any form of integrated pest management (IPM) on their farm and to what extent. It became apparent that most conventional farms used IPM to some extent, but during farmer questionnaires they did not mark themselves as IPM because they were unaware the management they use is classified as low level IPM. This point has been discussed in chapter discussions which compare farm management and within final limitations (*section 6.2*)

Furthermore, before conducting the questionnaire I assumed that farmers who used no chemicals would already be classified as certified organic. However, after discussing their farm management it was decided that input-free farms would also be classed as organic, including those that do not use copper as a fungicide (which is permitted through Soil Association organic). Therefore, during my analysis and results, I have not differentiated between organically certified farms and those that farm organically but without certification, these have been classified under the one term 'organic' for this thesis.

2.3 DATA COLLECTION

Primary data was collected on each farm during the years 2015 and 2016. Due to the different sizes of farms, the data collected was standardised according to how large the farm was. The data collected is split into three categories that links to the following chapters:

Biodiversity indicators – bird and butterfly community data were collected which included abundance, Shannon diversity and species richness. Species count information of both birds and butterflies were collected per orchard using point counts for birds and Pollard walks for butterflies as the surveying methods for Chapter 3.

Predation indicators – predation attempts and pest moth abundance were measured for Chapter 4. Predation attempts were measured using sentinel prey approach as the objective, repeatable measures of predation rates and pheromone traps were used for three species of moth pests that attracted male moths to the trap and count every 4-6 weeks.

Value indicators – farm financial data was collected through questionnaires to record insecticide application volume, insecticide costs and total saleable yield in 2015 and 2016. This information was used in Chapter 5 to answer questions surrounding yield valuation . Full questionnaires can be found in *Appendix A.*

2.3.1. Ethical code of conduct

Three pillars of ethics were followed to ensure information was collated sensitively and participants were informed throughout the study whilst knowing it was fully voluntary to be part of this research. These pillars were: *informed consent*, *beneficial to society*, and *confidential* (UCL Research Ethics Committee, 2019).

Farmers were informed of the study and what their role would be in the study from the first conversation. They were made aware that being part of the study was voluntary and no contract would tie them into providing information. Farmer participants were made aware that they could withdraw from the study with no repercussions if they chose to. Potential risks and benefits of the study were outlined during a conversation to make sure participant farmers had enough information about the study to make an informed decision to be included in the study.

The study research results will be presented within the farming community so to benefit participant farmers as well as the wider farming community, with this new research ongoing in the area where participants are living. The benefits to the overall community and information shared will be greatly valued by other farmers in the study area. There are no potential adverse effects or risks of the study on participants and this was highlighted during initial conversations.

During data collection, potentially sensitive information was given to the researcher by each farmer including farm address, financial information, insecticide application costs and yield data. Therefore, confidentiality was essential to protect the farm information from being used by the public. It was also important that farmers didn’t know other participants of the study. It is a close farming community so profit of an individual farm could be inferred if yield data and insecticide costs were publicly shared. All sensitive information was asked in the form of a voluntary email questionnaire and was completed by all participants.

2.3.2 Biodiversity indicators

This study has used bird abundance, species richness and diversity as the primary biodiversity indicators, to estimate the status in biodiversity. The bird data was collected over two years, 2015 and 2016. A secondary, more localised indicator was also measured – butterfly abundance, species richness and diversity. This butterfly data was collected in the spring and summer of 2016. There was an assumption here that an increase in abundance, species richness and diversity of birds and butterflies was the equivalent of an increase in biodiversity levels.

Butterflies can provide a proxy for plant diversity in the area, due to their dependency on particular food plants (Ehrlich & Raven, 1964). Birds are a good indicator of landscape quality and changes due to their susceptibility of habitat fragmentation and change; predator species may indicate changes at lower trophic levels, including changes in insect levels due to pesticide applications (Wilcove, 1985; Blair, 1999; Fuller, 2000).

Abundance of bird and butterfly species and their species richness is widely used as a proxy of biodiversity, including as UK government official measures (JNCC, 2014; Blair, 1999). Species richness is a particularly useful measure in terms of ecosystem functioning and ecosystem services; as species richness increases, so does the probability of filling ecological niches and exploiting a particular resource; thus fulfilling ecological functions (Tscharntke et al., 2008; Perfecto et al., 2004; Cadotte et al., 2011). Species richness alone, however, may sometimes lead to weighting common species higher than uncommon species, depending on the circumstances. Tscharntke et al. (2008) explain how agro-ecosystems with smaller fields and higher landscape heterogeneity have higher species richness, but these extra species may just be common and widespread with no adaptations to agricultural environments (Duelli, P 1992 in Tscharntke et al., 2008).

By solely focussing on maximising species through abundance and species richness, the dominance or "eveness" of species is missed (Büchs et al., 2003). Instead, Shannon Weiner Diversity Index accounts for this eveness and is often used as an additional measure of biodiversity along side species richness and abundance. Here, I use three measures of biodiversity: Shannon diversity, species richness and abundance to quantify biodiversity and the ecosystem service it provides.

Functional traits have been used in other studies as a measure of biodiversity as they determine how an organism might use resources, for example beak size can be measured to determine what food type avian species use (McGill et al., 2006; Cadotte et al., 2011). However, when focus is on measuring one particular ecosystem service, functional trait diversity may not be useful to determine the provision of ecosystem services through biodiversity. Instead, functional trait diversity is used to determine how ecological niches are filled in the community as a whole, rather than directly measuring an ecosystem service such as seed dispersal or pest regulation (Bregman et al., 2016).

2.3.3 Estimating bird biodiversity

2.3.3.1 Bird survey observers

Three experienced observers carried out orchard bird surveys for both biodiversity estimations and insectivore abundance. Observers were coded as SF, CP, and PL and all three observers were experienced in the British Trust for Ornithology (BTO) Breeding Bird surveys having their own survey sites to complete for BTO for at least two years in the local area. SF had the least experience on the BTO surveys having the two-year experience whereas CP and PL had over 10 years' experience. All three observers worked full time in ornithology in collaboration with agricultural or conservation research projects as the dedicated ornithologist. Additionally, all three ornithologists work in overseas birding tours as the priority bird guide. Their visual and audible skills were highly sort after lending them as very good options for this study.

Observer SF carried out all surveys in 2015, whereas CP and PL shared the responsibility in 2016. I ensured that both observers were not solely in charge of one type of farm management. They each had a mix of all four farm managements in their orchard distributions.

Observers were trained prior to the start of surveys in distance sampling, which all observers had already experienced. Observers were tested in field conditions on their hearing ability within an orchard using a pre-recorded tape of bird calls and played back at an unknown distance to the observer. All observers got all bird calls correct and estimated distances were inconsistent between them. This finding led to the use of a laser rangefinder

(Bushnell 202205 Sport 850 Vertical Laser Rangefinder) as a necessity during the study. Demonstrations using this model were given in field sites prior to the start of the study.

2.3.3.2 Bird sampling strategy

In order to survey each farm without bias the farms were overlaid with a grid of $100m^2$ in QGIS (QGIS Development Team, 2015). A plot is made up of a four-grid square which has one point at the centre of each (*figure 2.3*), termed "point counts" from here on. The point counts were chosen if they were 200m away from the neighbouring point and fell on an apple orchard. The maximum points per farm was 21 and the minimum was one. The number of point counts per farm was allocated depending on size of apple orchard area to ensure a proportional amount of point counts were undertaken. If more potential point locations were available than points allocated the most orchard-central point was chosen. Each point count was visited twice within the Breeding Bird Survey (BBS) time frame between April and June. Due to BBS time restrictions and dawn survey restrictions (BTO, JNCC, & RSPB, 2014), only 8-10 point counts were feasible per morning. This allowed for the largest farm to contain 21 point counts across the entire orchard area and all other farm point counts were proportional to this.

Ralph et al. (1995) and Sutherland et al. (2004) state that a bare minimum of 30 point counts per habitat is needed in order to perform analysis on rare birds as well as common birds. *Table 2.1* shows point counts per farm category in our study. Quinn et al. (2012) support this and use 16 point counts on each farm type. Other studies such as Petit & Petit (2003) consider the minimum count per habitat as 40 point counts. Considering the above studies, logistics, and time available on all farms in the study area, point counts in this study range between 30 – 120 per farm category, all proportionate to size of orchard area.

Table 2.1 Amount of point counts, number of farms and average orchard area per farm category

Farm Category	Point counts per category	No. of Farms	Average total orchard area (ha)
LEAF	120	8	41.04
IPM	80	7	36.03
Organic	30	9	8.02
Conventional	98	7	35.31

2.3.3.3 Bird field methods–distance sampling

The protocol for the bird surveys used was distance sampling. *Figure 2.3* shows the layout of an example orchard with at least 200m between each point count to ensure over counting of birds is minimised as radial bird surveys capped detections at 100m from the marked point count location.

Bird surveys were undertaken twice between April and June 2015, and twice between the same period in 2016, between dawn and three hours afterwards, advised due to the higher detectability of birds during this time (Sutherland et al., 2004). Audial and visual bird cues and their estimated distances were recorded using a laser range finder, and bird activity (e.g., fly over, perched, foraging, fly through) was noted, although flyovers were not used in analysis. Each 100m fixed-radius point count was 6 minutes, which included a 1-minute rest period before the 5-minute count period started. These timings are based on previous studies by Lynch (1995), who suggests more than 10 minutes at each point will reduce detectability of new species, as most species are detected within the first 5-10 minutes. The detection rate to find new species in the 0–5-minute interval was three times as high as the 10–15-minute interval. Keeping count time minimised at each point allows more point counts to be completed within the three-hour window each day. Hamel et al. (1996) state that point counts should only be 10 minutes long if time taken to walk from point to point is longer than 15 minutes. Weather conditions are highlighted as an important variable to keep constant during bird surveys: strong winds and moderate to heavy rainfall were avoided during surveys (Catterall et al., 2012; Giraudo et al., 2008; Marini, 2001).

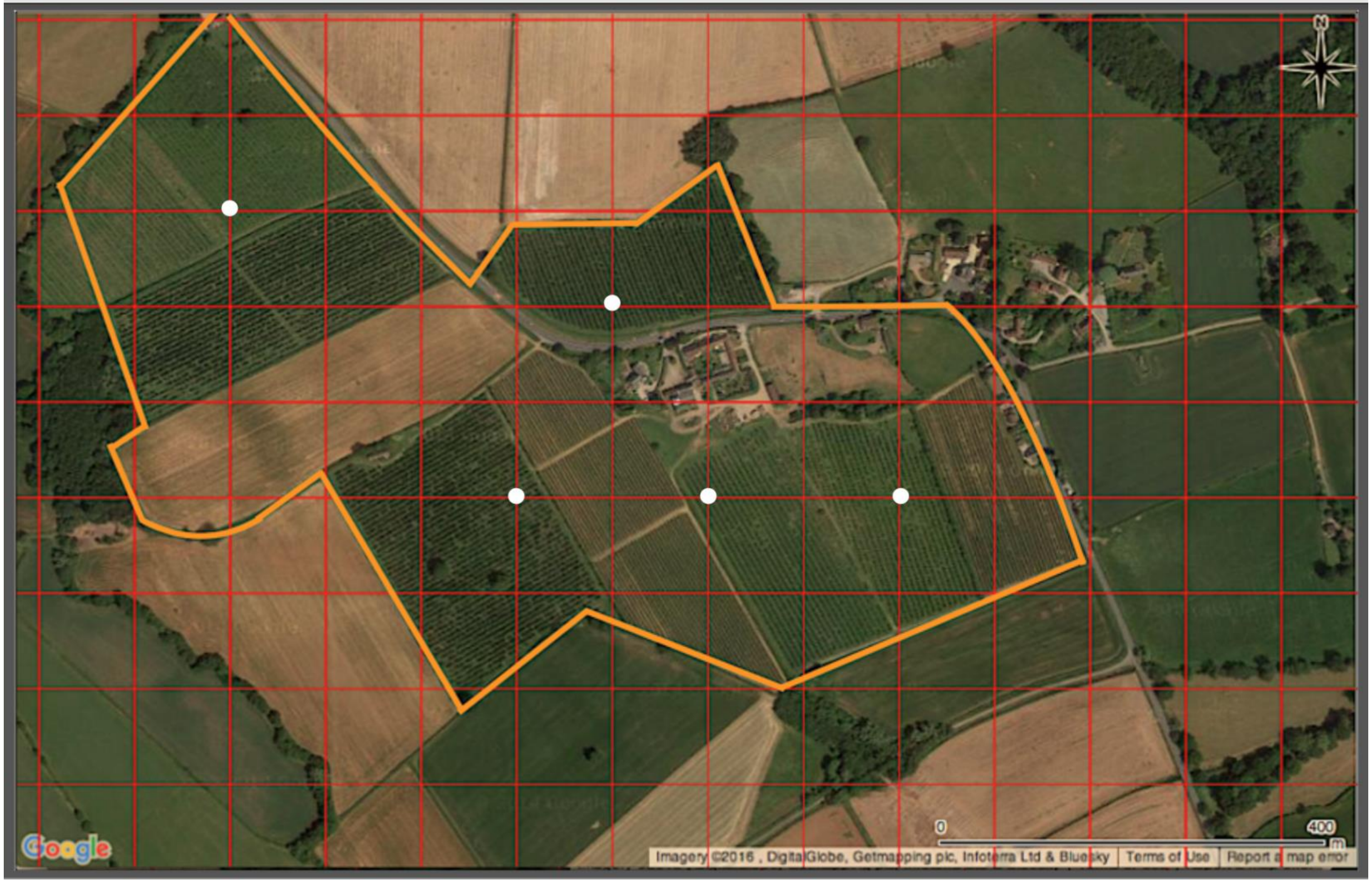


Figure 2.3 Site map of an example farm with a 100 m^2 grid layer with point count locations indicated (white dots) over one farm's orchard area (orange outline).

2.3.4 Estimating butterfly biodiversity

Butterfly surveys were carried out between May and October 2016. Every farm site was surveyed in favourable weather conditions (Pollard, 1977) every month in the butterfly survey timeframe, to avoid sampling the same individuals twice. The maximum visits per transect was five, with most transects being visited four times. Two sites were only visited once due to unfavourable weather, logistical issues, or ongoing farm management.

2.3.4.1 Butterfly sampling strategy

At each site a 250m transect was set out per orchard, and divided into two sections (i) orchard boundary and (ii) the orchard centre, in order to capture different orchard characteristics such as hedgerows, orchard edge, orchard centre, and any areas of dense shrub or natural habitat types found on each field, as described in Van Swaay et al. (2012; 2015).

The number of transects per farm was calculated relative to farm size; the smallest farms held one 250m transect and the largest held 20, 250m transects.

2.3.4.2 Butterfly field methods

Each transect was walked at a slow pace over 10 – 12minutes or 1km/hour, as recommended by van Swaay et al. (2015). Surveyors used a timer to ensue 12 minutes was not exceeded and to pause the timer at any stops where identification of butterflies may take longer.

Surveys were completed in only favourable conditions (Pollard, 1977):

- When temperature was between 13 – 35 °C
- No rain or strong winds (above Beaufort Scale 5, 19mph)
- During lower temperatures of 13 – 18 °C, 60% minimum sunshine is required
- During the hours between 09.30 – 16.30 (van Swaay et al., 2012)

The Pollard walk methodology was used (Pollard, 1977), where surveyor monitored for butterflies 2.5m either side of the transect line, 5m above and 5m in front. Any additional butterflies seen by the surveyor either outside the transect line or time frame were recorded separately, so not to encourage surveyors to search for rare species (Van Swaay et al., 2015). These additional butterflies were not included in analysis.

Species that were unidentifiable during flight were netted for observation and then released. Species that were unable to be distinguished in-field, such as *Pieris brassicae* and *Pieris rapae*, were classed as an aggregate species (Carter, 1982). Species that were still unidentifiable were marked as "unknown". Observers used the Field Study Council's guide for butterfly identification (Lewington & Bebbington, 1998). The use of this field identification booklet was essential to identify all butterflies to ensure the correct species was recorded.

2.3.4.3 Butterfly field observers

Butterfly observers were chosen based on previous experience and skills of identification. The only observer other than the author was Observer K. Both the author and Observer K had the same experience, both practicing with each other before the start of surveys and undertaking three surveys with the Butterfly Conservation UK in a local nature reserve. A local expert trained both observers on three surveys on orchards and made recommendations on which butterflies will likely use the orchard habitat most (Dean Fenton, *Pers Comms,* 2016).

2.3.5 Surrounding woodland and hedgerow cover

The effect of local surrounding semi-natural habitat such as coppices and hedgerows and woodland cover were grouped together and classed as "woody cover". Woody cover was developed by Tebbs & Rowland (2014), combining airborne radar data from NEXTMap® and satellite optical imagery. The woody cover product includes 'tall' features, above 2m in height, in the landscape by using canopy height information at a 5m x 5m spatial resolution scale. NDVI imagery was used to separate these tall features with other non-woody features such as buildings. The final woody cover product incorporates the Land Cover Map 2007, to identify larger areas of woodland, and the National Forest Inventory dataset (2015) leading to an accuracy of 85.6%, whereas LCM2007 is known to overestimate woody cover areas (Tebbs & Rowland, 2014).

This product has been tested using spatial imagery across Wales. The next test section was then created across Herefordshire to include all farms within this study.

2.3.6 Predation estimates

A popular method used for estimating predation is exclusion experiments, which involve erecting netted zones to allow insects access to a plant or crop, but not birds. Exclusion experiments in northern Spanish cider apple orchards haven shown how the presence of insectivorous birds actively reduce pest pressure in orchards (García et al., 2018). Further studies using exclusion zones also show how pests can be controlled by birds and bats, as well as arthropods in tropical and temperate agro-ecosystems (Perfecto et al., 2004; Maas et al., 2013; Garfinkel & Johnson, 2015; Kross et al., 2016; Mangan et al., 2017). However, such exclusion experiments do not allow for intra-guild predation complexities, as explored by Martin et al. (2015). In other words, to exclude birds from an experimental area will not consider the natural predation of arthropod natural enemies, as well as pests (Mooney et al., 2010), overestimating pest predation from arthropods rather than birds in bird exclusion zones.

Predation rates from birds in this study were inferred per farm by the use of sentinel prey as a live prey proxy, also used in various studies and ecosystems (Barbaro et al., 2016; Bereczki et al., 2014; 2015 and González-Gómez et al., 2006). The use of sentinel prey without the need for construction of exclusion zones, was used over live prey due to a study by Sam et al. (2015), who found no significant deference between the predation rates of live prey (meal worms) and dummy prey in a tropical ecosystem study.

Peisley et al. (2016) suggests that real prey are more useful as a measure of pest predation levels from birds, however other studies suggest plasticine prey are more useful as they allow attack marks to be identified to taxa level and exclosures are not necessary (Howe et al., 2009). Underestimation of predation must be considered because live prey are likely to be predated upon more; live herbivorous caterpillars cause plants to release volatiles as leaf damage is caused, which chemically attracts predators (Mantyla et al., 2008; Sam et al., 2015).

Sentinel prey mimicked two of the key pest caterpillars to orchards: Codling moth (*Cydia pomonella L*) and fruit tree tortix (*Archips podana (Scolopi))*. Both plasticine and dough were used as the model material. For more information on the reasoning behind choice of sentinel prey, see *Supplementary Materials 1*. True predation rates are not necessary in this study, instead inferred predation rates from attack rates and impressions are used, like Sam et al. (2015).

2.3.6.1 Sentinel prey field methods

Plasticine sentinel prey experiments took place between April and August 2016. Each 'prey presentation station' was the point count location used previously for bird surveys. The prey presentation stations had eight green and eight cream plasticine caterpillars: 16 prey in total, with green caterpillars on leaves and cream caterpillars on the bark of one apple tree. Bird behavioural experiments were undertaken to understand if birds avoided or were more drawn to a certain colour, see *Supplementary Material 2.* Each prey presentation station was set up with a camera trap on an adjacent tree in a position that would catch predation attempts from birds. Camera traps were used for secondary predator identification evidence and serve as a calibration of predation marks left such as Howe et al. (2009).

Plasticine sentinel prey were constructed using a non-toxic plaster called "Newplast". 10mm and 2mm lengths were rolled into a small cylinder with a curve in the centre to resemble a moving caterpillar (*figure 2.4*).

Figure 2.4 Sentinel prey 10mm in length, 2mm wide plasticine model caterpillar. Bird attack marks are shown on the right-hand side of the sentinel prey.

Dough sentinel prey experiments took place from December 2016 to March 2017. At each prey presentation station, ten dough sentinel prey were planted at each point count on the trunk of one apple tree (*figure 2.5*). A 3:1 flour to lard mixture was rolled out to 2mm thickness. 10mm long strips were cut and manipulated to produce a model wider in the middle and tapered towards the ends, to resemble codling moth pupae, which mature under bark during winter months (SIR, 2018).

Figure 2.5 Dough model codling moth pupae, the marks are bird attack marks, leaving the model dis-shaped.

Sentinel prey were set in place using "superglue" (Sam et al., 2015) and presentation stations were left for 24 hours of exposure and then visually identified for predation attempts. The time limit of 24 hours was chosen to give a daily predation rate per farm, yet not allowing the resident breeding birds to become accustomed to the artificial prey. Predation marks were identified to course taxonomic levels – between birds, arthropods, or mammals (*figure 2.6).* Conclusions from Low et al. (2014) recommend identifying predators to coarse taxonomic levels to Class rather than more precise levels. The identification of these attack marks was based on the findings and identification key of Low et al. (2014). The findings were derived from presenting model prey caterpillars, made from light green non-toxic modelling clay to a range of captive predators (birds, mammals, birds, and arthropods), wild predators (arthropods and rodents), and to museum specimen impressions. The sentinel prey identification key was used for each predation attack mark to use as a reference in-field, another recommendation from Low et al. (2014). Finally, farmer knowledge of local predators in orchards was drawn upon to draw conclusions from attack mark identification, the most likely predators in orchards were arthropods, especially earwigs as farmers sometimes release earwigs into the orchards to act as a natural pest control (Malcolm Davies, *Pers Comms,* 2016).

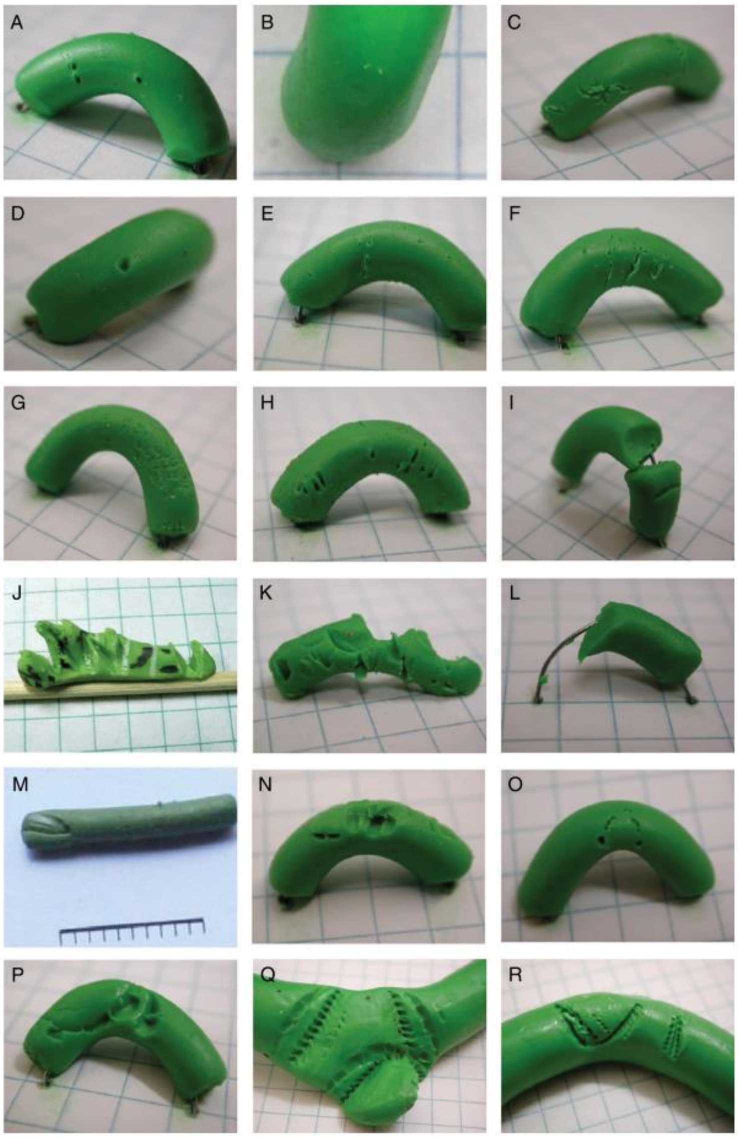


Figure 2.6 - Sentinel prey identification key taken from Low et al. (2014; 124). These are examples of attack marks from arthropods (A – H), birds (I – M), mammals (N – P), and reptiles (Q – R). Low et al. (2014) placed these example prey on 5mm grid to show scale, other than M which is shown with a scale of 1mm intervals.

This assessment had to be in-situ and with an assistant to additionally identify the marks left, as sentinel prey would change shape easily and any predation attempt may have been lost if the shape was lost in transit. Therefore, the guidelines of Low et al. (2014) couldn't be followed here, where independent assessment of the attack mark is recommended to increase confidence in the predator assessment.

Co-ordinates were noted down of each point count location to ensure the pest presentation station was in the same location as bird surveys. Flagging tape was used to identify each tree, but placed on the opposite row, away from the dummy caterpillar so birds are less likely to associate the flag with unpalatable prey, as recommended by Sam et al. (2015).

Each prey presentation station was set up with Ltl Acorn camera traps, *model 5310* on an adjacent tree in a position that would catch predation attempts from individual birds to identify if certain bird species were more dominant in the predation of sentinel prey than others, but also to serve as a calibration of predation marks left (Howe et al., 2009).

During collection, inspection of the dummy was taken using a microscope; notes and pictures were taken to describe the damage and any nearby predators, such as ants, as suggested by Low et al. (2014) and Sam et al. (2015).

A single predation event was counted as either one bite mark or multiple bite marks on the same dummy caterpillar, of the same predator type (Ferrante et al., 2014). The two bites marks could have been from different individual predators if the same taxa. However, the reason for counting two bites as a single predation, is if the caterpillar was bitten then it would have been predated upon and not remained in situ. If two bites were counted as separate predation attempts, there runs a risk of overestimating predation rates.

2.3.7 Pest estimates

UK apple orchards have six pest moths associated with them, affecting different parts of an apple tree. Codling moth (*Cydia pomonella L.)* and fruit tree tortrix *(Archips podana (Scolopi))* are most significant in terms of economical damage caused to apples (Solomon et al., 2000), and apple ermine (*Yponomeuta malinellus)* is a more localised pest to Herefordshire and monophagous on apple trees (Menken, Herrebout, & Wiebes, 1992), which is becoming prevalent locally and causing economic damage due to reduced yield (John Worle, *Pers Comms,* 2018). Light brown apple moth is a prevalent pest but is not specific to apple trees (UK Moths, 2018) and winter moth is also prevalent but due to seasonality of the species and timings of the surveys in spring this species was unable to be surveyed. For more details on UK moth pests to apple orchards see *Appendix H.*

Farms often measure the abundance of pest moths in the orchards using males as a population proxy. Pheromone traps are frequently used for a multitude of moth pests within an integrated pest management system when monitoring pest levels. Small amounts of pheromones are used for communication in insects, for example for mating to attract males towards female moths (Witzgall et al., 2010). Sex pheromone traps can be used as a population estimate for specific moth species to determine a threshold population for optimum time to use chemical insecticides (Carden, 1987; Reddy & Guerrero, 2001) and to infur moth populatons (Riedl & Croft, 1974). As sex pheromone traps are species-specific, they have been recommended as a control method for specific pest populations as a crop protection method in itself, instead of using chemical insecticides (Witzgall et al., 2010). This specific monitoring of pest species is one of the main techniques used within an IPM approach (EC, 2017a).

The pheromone is held within a silicone holder and placed in the centre of a sticky pad within a plastic tripod structure (*figure 2.6*), attached to an apple tree canopy at head height by string or wire. The pheromone of the female moth inside will attract male moths in a 2-hectare area for regular bush orchards. This method is widely used on apple orchards that control moth pests and monitoring takes place throughout the year, depending on the pest species. Through monitoring pest species using these pheromones, chemical insecticide usage against the moth may be reduced depending on the pest thresholds. ADHB recommend threshold for each pest species (*table 2.2*), where under the threshold there is no need to spray, and if you reach over the threshold then it is recommended to spray against the moth pest before population increases (ADHB, 2018).

Table 2.2 Pest moth thresholds. This table shows the economic thresholds of the 3 moth pests monitored in this study. The thresholds are a measure of when to start using chemicals to control the pest according to the “Apples best practice guide” by the Agriculture and Horticulture Development Board (AHDB, 2018). As apple ermine is a new pest to the area, they will not measure the threshold for this pest, rather measure for codling moth and fruit tree tortrix and then control apple ermine secondarily.

Pest Moth species	Economic Threshold (per week, per trap)
Codling moth (*Cydia pomonella L.)*	5
Apple ermine (*Yponomeuta malinellus)*	*Not available*
Fruit tree tortrix *(Archips podana (Scolopi))*	30

Geographical Information Systems were used to identify the centre of each orchard (QGIS Development Team, 2015). This was to ensure the moth population was being recorded within the orchard, rather than from the outside.

Figure 2.7 Example of pheromone trap situation in apple canopy (A). Example of the sticky pads and silicone pheromone holder (B).

2.3.8 Farm financials

Finally, the last piece of primary data collected was farmers' financial information, following my outlines ethical code of conduct in *section 2.3.1*. Questionnaires were sent to each farmer at the end of apple harvest around November, December in 2015 and 2016. The questionnaire aimed to find out the total saleable harvest sent off for that year, the total volume of insecticide used and the cost of that insecticide volume for that year. The answers to these questions remained anonymous and coded for analysis. This ensured the requests of farmer confidentiality were met and I abided by an ethical code of conduct when dealing with the personal and sensitive information of my study farmers (Newing et al., 2011). Questionnaires were sent via email and farmers were politely reminded throughout December, January, and February to complete the questionnaire. All farmers completed the 2015 and 2016 questionnaire. Full questionnaires can be found in *Appendix A*.

2.4 REFERENCES

AHDB. (2018). *Apples Best Practice Guide. Pests and Disease Control*. Retrieved from http://apples.ahdb.org.uk/ipdm.asp#

Barbaro, L., Rusch, A., Muiruri, E. W., Gravellier, B., Thiery, D., & Castagneyrol, B. (2016). Avian pest control in vineyards is driven by interactions between bird functional diversity and landscape heterogeneity. *Journal of Applied Ecology*. http://doi.org/10.1111/1365-2664.12740

Barzman, M., Bàrberi, P., Birch, A. N. E., Boonekamp, P., Dachbrodt-Saaydeh, S., Graf, B., … Sattin, M. (2015). Eight principles of integrated pest management. *Agronomy for Sustainable Development*, *35*(4), 1199–1215. http://doi.org/10.1007/s13593-015-0327-9

Bengtsson, J., Ahnström, J., & Weibull, A. C. (2005). The effects of organic agriculture on biodiversity and abundance: A meta-analysis. *Journal of Applied Ecology*, *42*, 261–269. http://doi.org/10.1111/j.1365-2664.2005.01005.x

Bereczki, K., Hajdu, K., & Báldi, A. (2015). Effects of forest edege on pest control service provided by birds in fragmented temperate forests. *Acta Zoologica Academiae Scientiarum Hungaricae*, *61*(3), 289–304. http://doi.org/10.17109/AZH.61.3.7.2015

Bereczki, K., Ódor, P., Csóka, G., Mag, Z., & Báldi, A. (2014). Effects of forest heterogeneity on the efficiency of caterpillar control service provided by birds in temperate oak forests. *Forest Ecology and Management*, (327), 96–105.

http://doi.org/10.1016/j.foreco.2014.05.001

Bregman, T. P., Lees, A. C., MacGregor, H. E. A., Darski, B., de Moura, N. G., Aleixo, A., … Tobias, J. A. (2016). Using avian functional traits to assess the impact of land-cover change on ecosystem processes linked to resilience in tropical forests. *Proceedings of the Royal Society B: Biological Sciences*, *283*(1844). http://doi.org/10.1098/rspb.2016.1289

BTO, JNCC, & RSPB. (2014). *Breeding Bird Survey Instructions*. Norfolk. Retrieved from http://www.bto.org/volunteer-surveys/bbs/taking-part/download-forms-instructions

Büchs, W., Harenberg, A., Zimmermann, J., & Weiß, B. (2003). Biodiversity, the ultimate agri-environmental indicator? *Agriculture, Ecosystems & Environment*, *98*(1–3), 99–123. http://doi.org/10.1016/S0167-8809(03)00073-2

Carden, P. W. (1987). Supervised control of apple pests in southern England. *Crop Protection*, *6*(4), 234–243.

Carter, D. (1982). *Butterflies & Moths in Britain and Europe.* Pan Books (in association with the British Museum Natural History).

Catterall, C. P., Freeman, A. N. D., Kanowski, J., & Freebody, K. (2012). Can active restoration of tropical rainforest rescue biodiversity? A case with bird community indicators. *Biological Conservation*, *146*(1), 53–61. http://doi.org/10.1016/j.biocon.2011.10.033

E. Birch, A. N., Begg, G. S., & Squire, G. R. (2011). How agro-ecological research helps to address food security issues under new IPM and pesticide reduction policies for global crop production systems. *Journal of Experimental Botany*, *62*(10), 3251–3261. http://doi.org/10.1093/jxb/err064

EC. (2017a). Integrated Pest Management.

EC. (2017b). *Report from the Commission to the European Parliament and the Council on Member State National Action Plans and on progress in the implementation of Directive 2009/128/EC on sustainable use of pesticides*. Brussels.

Ehrlich, P. R., & Raven, P. H. (1964). Butterflies and Plants: A Study in Coevolution. *Evolution*, *18*(4).

European Commission. (2018). *CO-FREE Strategies to reduce copper pesticides* (No. Project ID: 289497). Germany.

Evelyn, J. (1664). *Pomona: Sylva, or a discourse of forest-trees and the propagation of timber*. London, UK: Royal Society.

Ferrante, M., Cacciato, L. O., & Lövei, G. L. (2014). Quantifying predation pressure along an urbanisation gradient in denmark using artificial caterpillars. *European Journal of Entomology*, *111*(5), 1–6. http://doi.org/10.14411/eje.2014.082

Fuller, R. J. (2000). *Relationships between recent changes in lowland British agriculture and farmland bird populations: an overview*. Thetford, Norfolk.

Garfinkel, M., & Johnson, M. (2015). Pest-removal services provided by birds on small organic farms in northern California. *Agriculture, Ecosystems & Environment*, *211*(2015), 24–31. http://doi.org/10.1016/j.agee.2015.04.023

Giraudo, A. R., Matteucci, S. D., Alonso, J., Herrera, J., & Abramson, R. R. (2008). Comparing bird assemblages in large and small fragments of the Atlantic Forest hotspots. *Biodiversity and Conservation*, *17*(5), 1251–1265. http://doi.org/10.1007/s10531-007-9309-9

González-Gómez, P. ., Estades, C. ., & Simonetti, J. . (2006). Strengthened insectivory in a temperate fragmented forest. *Oecologia*, (148), 137–143.

Goodman, L. A. (1961). Snowball sampling. *The Annals of Mathematical Statistics*, *32*, 148–170.

Hamel, P. B., Smith, W. P., Twedt, D. J., Woehr, J. R., Morris, E., Hamilton, R. B., & Cooper, R. J. (1996). *A Land Manager's Guide to Point Counts of Birds in the Southeast*. Asheville.

Hardman, C. J., Harrison, D. P. G., Shaw, P. J., Nevard, T. D., Potts, S. G., & Norris, K. (2016). Supporting local diversity of habitats and species on farmland : a comparison of three wildlife-friendly schemes. *Journal of Applied Ecology*, *53*, 171–180. http://doi.org/10.1111/1365-2664.12557

Hardman, C. J., Norris, K., Nevard, T. D., Hughes, B., & Potts, S. G. (2016). Delivery of floral resources and pollination services on farmland under three different wildlife-friendly schemes. *Agriculture, Ecosystems & Environment*, *220*(2016), 142–151. http://doi.org/10.1016/j.agee.2016.01.015

Henckel, L., Börger, L., Meiss, H., Gaba, S., & Bretagnolle, V. (2015). organic fields sustain weed metacommunity dynamics in farmland landscapes. *Proc R Soc B, 282*, 1–9.

Hillocks, R. J. (2012). Farming with fewer pesticides: EU pesticide review and resulting challenges for UK agriculture. *Crop Protection, 31*. http://doi.org/10.1016/j.cropro.2011.08.008

JNCC. (2014). *Uk Biodiversity Indicators 2014. Measuring progrees towards halting biodiversity loss*.

Katayama, N. (2016). Bird diversity and abundance in organic and conventional apple orchards in northern Japan. *Nature*, *6*(September), 1–7. http://doi.org/10.1038/srep34210

Kross, S. M., Kelsey, T. R., Mccoll, C. J., & Townsend, J. M. (2016). Field-scale habitat complexity enhances avian conservation and avian-mediated pest-control services in an intensive agricultural crop. *Agriculture, Ecosystems and Environment*, *225*(2016), 140–149. http://doi.org/10.1016/j.agee.2016.03.043

LEAF. (2014). *LEAF Marque Global Standards*. Warwickshire.

LEAF. (2016). *LEAF in 2016: Delivering more sustainable food and farming*. Warwickshire.

Lewington, R., & Bebbington, J. (1998). Guide to the butterflies of Britain. Field Studies Council.

Low, P. A., Sam, K., McArthur, C., Posa, M. A. C., Hochuli, D. F. (2014). Determining predator identify from attach marks left in model caterpillars: guidelines for best practice. *Entomologia Experimentalis et Applicata 152: 120–126*. http:// doi: 10.1111/eea.12207

Lynch, J. F. (1995). *Effects of Point Count Duration, Time-of-Day , and Aural Stimuli on Detectability of Migratory and Resident Bird Species in Quintana Roo, Mexico*.

Maas, B., Clough, Y., & Tscharntke, T. (2013). Bats and birds increase crop yield in tropical agroforestry landscapes. *Ecology Letters*, *16*(12), 1480–1487. http://doi.org/10.1111/ele.12194

Mangan, A. M., Pejchar, L., & Werner, S. J. (2017). Bird use of organic apple orchards: Frugivory, pest control and implications for production. *PloS One*, *12*(9), 1–15. http://doi.org/10.1371/journal.pone.0183405

Marini, M. A. (2001). Effects of forest fragmentation on birds of the cerrado region, Brazil. *Bird Conservation International*, *11*, 13–25.

Martin, E. A., Reineking, B., Seo, B., & Steffan-Dewenter, I. (2015). Pest control of aphids depends on landscape complexity and natural enemy interactions. *PeerJ*, *3*. http://doi.org/10.7717/peerj.1095

McGill, B. J., Enquist, B. J., Weiher, E., & Westoby, M. (2006). Rebuilding community ecology from functional traits. *Trends in Ecology and Evolution*, *21*(4), 178–185. http://doi.org/10.1016/j.tree.2006.02.002

Menken, S. B. J., Herrebout, W. M., & Wiebes, J. T. (1992). Small Ermine Moths (Yponomeuta): Their Host Relations And Evolution. *Annual Review of Entomology*, *37*(1), 41–66. http://doi.org/10.1146/annurev.ento.37.1.41

Natural England. (2010). *Traditional orchards: orchards and wildlife*. *Technical Information Note* (Vol. TIN020).

Natural England. (2013). *National Character Area profile: 100. Herefordshire Lowlands*.

Natural England. (2014a). *National Character Area profile: 101. Herefordshire Plateau*.

Natural England. (2014b). *National Character Area profile: 104. South Herefordshire and Over Severn*.

Newing, H., Eagle, C. M., Puri, R. K., & Watson, C. W. (2011). *Conducting Research in Conservation. A Social Science Perspective*. Routledge Taylor & Francis Group, 2 Park Square, Milton Park, Abingdon, Oxon OX14 4RN.

Pearce, S., & Zalucki, M. P. (2006). Do predators aggregate in response to pest density in agroecosystems? Assessing within-field spatial patterns. *Journal of Applied Ecology*, *43*(1), 128–140. http://doi.org/10.1111/j.1365-2664.2005.01118.x

Pepsico. (2010). *Passionate about growing: PepsiCo UK sustainable farming report 2010*. UK & Ireland.

Perfecto, I., Vandermeer, J. H., Bautista, G. L., Nuñez, G. I., Greenberg, R., Bichier, P., &

Langridge, S. (2004). Greater predation in shaded coffee farms: The role of resident neotropical birds. *Ecology*, *85*(10), 2677–2681. http://doi.org/10.1890/03-3145

Pollard, E. (1977). A method for assessing changes in the abundance of butterflies. *Biological Conservation*, *12*.

Kogan, M. (1998). Integrated Pest Management: Historical Perspectives and Contemporary Developments. *Annual Review of Entomology*, *43*(1), 243–270. http://doi.org/10.1146/annurev.ento.43.1.243

QGIS Development Team. (2015). QGIS Geographic Information System. Open Source Geospatial Foundation Project. Retrieved from http://qgis.osgeo.org

Quinn, J. E., Brandle, J. R., & Johnson, R. J. (2012). The effects of land sparing and wildlife-friendly practices on grassland bird abundance within organic farmlands. *Agriculture, Ecosystems & Environment*, *161*, 10–16. http://doi.org/10.1016/j.agee.2012.07.021

Ralph, C. J., Droege, S., & Sauer, J. R. (1995). *Managing and Monitoring Birds Using Point Counts*.

Reddy, G. V. P., & Guerrero, A. (2001). Optimum timing of insecticide applications against diamondback moth Plutella xylostella in cole crops using threshold catches in sex pheromone traps. *Pest Management Science*, *57*(1), 90–94. http://doi.org/10.1002/1526-4998(200101)57:1<90::AID-PS258>3.0.CO;2-N

Riedl, H., & Croft, B. A. (1974). A study of pheromone trap catches in relation to codling moth (Lepidoptera: Olethreutidae) damage. *The Canadian Entomologist*, (106), 525–537.

Schneider, M. K., Lüscher, G., Jeanneret, P., Arndorfer, M., Ammari, Y., Bailey, D., … Herzog, F. (2014). Gains to species diversity in organically farmed fields are not propagated at the farm level. *Nature Communications*, *5*(May), 4151. http://doi.org/10.1038/ncomms5151

SIR. (2018). The Pests: Codling Moth Life Cycle. Retrieved May 31, 2018, from https://www.oksir.org/the-pests/codling-moth/lifecycle/

Soil Associaton (2019). Soil Association Standards: Farming and growing. Version 18.1: 1-180. Retrieved from https://www.soilassociation.org/our-standards/read-our-organic-standards/farming-growing-standards/

Solomon, M. G., Cross, J. V., Fitzgerald, J. D., Campbell, C. a. M., Jolly, R. L., Olszak, R. W., … Vogt, H. (2000). Biocontrol of Pests of Apples and Pears in Northern and Central Europe - 3. Predators. *Biocontrol Science and Technology*, *10*(2), 91–128. http://doi.org/10.1080/09583150029260

Tebbs, E., & Rowland, C. (2014). *A high spatial resolution woody cover map for Great Britain: preliminary results.* Lancaster.

The Cider Route. (2014). The Cider Route. Retrieved from http://www.ciderroute.co.uk/cider-producers/

Trewavas, A. (2001) Urban myths of organic farming. *Nature commentary, 410,* 409-41

Tscharntke, T., Sekercioglu, C. H., Dietsch, T. V., Sodhi, N. S., Hoehn, P., & Tylianakis, J. M. (2008). Landscape Contraints on Functional Diversity of BIrds and Insects in Tropical Agroecosystems. *Ecology*, *89*(4), 944–952.

Tuck, S. L., Winqvist, C., Mota, F., Ahnström, J., Turnbull, L. a., & Bengtsson, J. (2014). Land-use intensity and the effects of organic farming on biodiversity: a hierarchical meta-analysis. *Journal of Applied Ecology*, *51*(3), 746–755. http://doi.org/10.1111/1365-2664.12219

UCL Research Ethics Committee (2016) *Accepted Ethical Standards.*
https://ethics.grad.ucl.ac.uk/accepted-ethical-standards.php. [Accessed on 23rd September 2019].

van Swaay, C Brereton, T Kirkland, P., & Warren, M. (2012). *Manual for Monitoring Butterflies.* Wageningen.

Van Swaay, C., Regan, E., Ling, M., Bozhinovska, E., Fernandez, M., Marini-Filho, O. J., … Underhill, L. (2015). *Guidelines for Standardised Global Butterfly Monitoring* (Vol. GEO BON Te). Leipzig, Germany.

Wilcove, D. S. (1985). Nest Predation in Forest Tracts and the Decline of Migratory Songbirds. *Ecology*, *66*(4).

Witzgall, P., Kirsch, P., & Cork, A. (2010). Sex pheromones and their impact on pest management. *Journal of Chemical Ecology*, *36*(1), 80–100. http://doi.org/10.1007/s10886-009-9737-y

Appendix A

Farm yield financial questionnaire

This questionnaire was the same structure and layout for 2016.

Yield Questionnaire for 2015

Please fill in and return in a reply to this email

Question 1:
What was the total apple area that was harvested in **2015**?
.... ha/acres

Question 2:
What was the total saleable apple yield in tonnes for **2015**? And for which end product?

Answer:
…………………… Tonnes for dessert
…………………… Tonnes for cider
…………………… Tonnes for juice

Question 3:
A) What insecticides were used this year on the apple orchards? B) What quantity was used and C) how much did this cost you this year in total (2016)?

Answer A:
Chemical 1 name …………………..
Chemical 2 name …………………..
Chemical 3 name …………………..
Chemical 4 name …………………..
Chemical 5 name …………………..
Chemical 6 name …………………..

Answer B:
Chemical 1 quantity (kg/L) used on apples in 2015 …………….
Chemical 2 quantity (kg/L) used on apples in 2015 ……………..
Chemical 3 quantity (kg/L) used on apples in 2015……………….
Chemical 4 quantity (kg/L) used on apples in 2015 …………….
Chemical 5 quantity (kg/L) used on apples in 2015 …………….
Chemical 5 quantity (kg/L) used on apples in 2015 …………….

Answer C:
Chemical 1 total cost for apple fields applications in 2015 £………………….
Chemical 2 total cost for apple fields applications in 2015 £………………….
Chemical 3 total cost for apple fields applications in 2015 £………………….
Chemical 4 total cost for apple fields applications in 2015 £………………….
Chemical 5 total cost for apple fields applications in 2015 £………………….
Chemical 6 total cost for apple fields applications in 2015 £………………….

End of Questionnaire

Appendix A1

Initial farmer questionnaire

This questionnaire was conducted during January and February of 2015 during first conversations with study farmers.

Questions	*Please provide your answers on the space below:*
1. What county is your farm located?	
2. How many hectares is your whole farm?	
3. Approximately what percentage of your farm's land area is used for growing apples?	…………………… %
4. What other crops are on your farm?	
5. Approximately what percentage of your farm's land area is left as natural habitat? (This may include hedgerows and any wild areas)	……………………..%
6. Is your farm LEAF certified?	
7. Are you in the process of being LEAF certified?	
8. Are you certified for any other environmental scheme? (e.g Organic)	
9. Does your farm carry out any environmentally integrated management practices such as *integrated pest management* (IPM)?	To some extent / To a great extent / Not at all If so please provide some detail:
10. Is your farm managed in a way that allows you to receive benefits from Government's environmental stewardship schemes?	Yes / No If so, please indicate the level (delete as appropriate): Higher Level (HLS) / Organic Entry Level (OELS) / Entry Level (ELS)
11. Do you grow apples mainly for dessert apples, for juicing or mixed?	

CHAPTER 3

Organic farming, not LEAF or IPM, support farmland biodiversity in UK apple orchards

3.1 ABSTRACT

The effects of organic and government-run agricultural environmental schemes on biodiversity have been vastly reported within academic literature throughout Europe, however new farm certification schemes continue to arise with little research undertaken on them. We have compared four types of farm management practiced on apple orchards in Herefordshire, Gloucestershire, and Worcestershire to compare the impact they have on farmland biodiversity. These include Linking Environment and Farming (LEAF), integrated pest management (IPM), organic, and conventionally managed orchards. We investigated the levels of biodiversity across the management types by using (i) butterfly abundance, species richness and diversity and (ii) bird abundance, density, species richness and diversity. These biodiversity indicators were sampled using a representative selection of each management type across 30 farms in total. Organic apple orchards supported a higher diversity, abundance, species richness and density of birds than LEAF, IPM and conventional orchards. IPM orchards had significantly less butterfly abundance and butterfly abundance were lower on larger organic orchards in comparison to larger conventional orchards. Butterfly species richness and diversity did not differ significantly between apple management types. However, butterfly abundance was low nationwide in 2016, which are discussed as a possible influence on results. These findings support the plethora of studies highlighting organic farming's positive impact on farmland biodiversity and that organic farming is significantly more efficient at protecting biodiversity than other farm environmental schemes.

3.2 INTRODUCTION

3.2.1 Global agricultural intensification and biodiversity loss

Agriculture intensification dramatically increased in Europe after World War II: farms decreased in number and increased in size whilst farm labour decreased due to mechanisation and yield increased fourfold which all led to a decline in biodiversity

(Robinson & Sutherland, 2002). By 2050 global food demand is expected to double, yet approximately half of useable land – land which is defined as non-tundra, dessert, rock or boreal - is already used by pastoral or intensive agriculture (Tilman et al., 2001; Tilman et al., 2002). Agricultural intensity happens on two scales: by simplifying wider landscapes that creates fragments of natural habitat, and secondly on a local scale through increasing the use of chemical fertilisers and pesticides as well as modern-day farm practices. For more examples of local and landscape intensification see Tscharntke et al. (2005).

Wider environmental impacts caused by agricultural expansion and intensification have contributed to global warming through nitrogen fertilisers, that contribute to the greenhouse gas Nitrous Oxide, increased rice paddies, and livestock farming - the highest anthropogenic contributor of the greenhouse gas Methane (Tilman et al., 2002), where ruminants in the developed world cause 75% of global GHG emissions (Herrero et al., 2013). With current trends of agricultural practices intensifying in developed countries and extensively farming through land clearing in developing countries, total land cleared is estimated to reach approximately 1 billon hectares by 2050, significantly increasing greenhouse gas emissions and nitrogen use (Tilman et al., 2011; Burney et al., 2010). These increases have subsequent negative effects on species extinctions (WWF, 2014), through water pollution, land alteration and climate change (Vitousek et al., 2010; Tilman et al., 2011; Thomas et al., 2004).

Increased intensification and expansion of agricultural practices, due to resource and food demands from people (Munang et al., 2010; Tilman, 2011), has fragmented natural habitats that support biodiversity – the variety of living plants and animals (Bianchi, 2006; Dirzo & Raven, 2003). Latest reports on the state of our world's species populations has highlighted that in the last half a century species populations have halved: 52% of vertebrate species populations have declined since 1970 (WWF, 2014). A major cause of this loss is through habitat destruction due to agricultural expansion (Kissinger, 2012; Tilman et al., 2001), especially the conversion of forests into agricultural systems (Pereira et al., 2010). Intensification of farming has an overall negative impact on biodiversity and proves to be the biggest threat to bird species globally in both the developing and developed countries, accounting for 57% and 33% for near threatened species, respectively; and 40% and 24% of threats in threatened species, respectively (Green et al., 2005). However, agriculture may also benefit current biodiversity through promotion of low-input farming on the 40% of land it inhabits (Garfinkel & Johnson, 2015; Mclaughlin, 2011; Tscharntke et al., 2005).

The well-known debate of land-sparing or land-sharing (Phalan et al., 2011) has demonstrated that, theoretically, sparing land for preservation will increase most species populations. Yet, currently the mechanisms are not yet in place to enable these theoretical farming methods to be a success in reality. Balmford et al. (2012) caution that vigilance must be taken in land sparing practices to ensure that the spared land is not used for other intensive purposes for non-wildlife habitat uses. This is a likely scenario as when agriculture becomes more intensive it starts to produce higher yields over a small area, which could be seen as an opportunity to produce more food in the same way on the 'spared' land (Guitierrez-Velez et al., 2011). Other human activities that may find economic use for any spared land are housing developments, which are more cost efficient and profitable than converting land back into agriculture (Godfray et al., 2010; Nellemann et al., 2009), or natural reserves. With other human activities deemed more essential this makes land sparing, which had the intention to revert to natural reserves, a less likely scenario without strong governmental enforcement. Therefore, intensification in certain areas must be accompanied by increased conservation and development efforts (Burney et al., 2010). Land sharing, or wildlife-friendly farming through organic farming or farmland environmental schemes, is the alternative theory to land sparing that also has advantages and disadvantages, often including yield penalties (Phalan et al., 2011; Green et al., 2005; Donald, 2004).

3.2.2 Farmland environmental schemes

Farmland environmental schemes that support the protection of semi-natural habitat patches on farms, such UK government's agri-environment schemes and High Nature Value farms, have been known to benefit species and abundances in some cases, although this is dependent on the landscapes in which they were conducted (Aue, 2014; Batáry et al. 2011). However Kleijn et al. (2006) found only marginal benefits to biodiversity of some government-led schemes, especially in uncommon species. Various other studies, including those on organic systems, have shown that having higher proportions of semi-natural landscape elements to increase landscape heterogeneity with low management intensity is beneficial for species abundance and richness (Bengtsson et al. 2005; Benton et al., 2003; Billeter et al., 2008).

Although individual studies have looked at how agri-environmental schemes can still be beneficial to biodiversity through the provision of habitat, instead of a reduction in chemical inputs (Hardman et al., 2016). These have had mixed results elsewhere in terms of the effect on biodiversity (Kleijn et al., 2001; 2003; 2011), and still somewhat unexplored in terms of scientific research on their effectiveness to biodiversity and ecosystem services (Samnegård et al., 2019).

Governmental agri-environmental schemes and their benefits to biodiversity are contested (Kleijn et al., 2001; 2006; Butler et al. 2007) and although correlations have previously been demonstrated, they have been rejected due to weak experimental design with bias towards favourable sites (Kleijn & Sutherland, 2003). Leventon et al. (2017) suggest that the deployment and management of agri-environmental schemes under the Common Agricultural Policy need a fundamental rethink to become effective in reducing actor fragmentation, and to promote collaboration. Furthermore, not all environmental farming schemes, especially industry-led schemes such as LEAF certification, have been assessed scientifically in terms of benefits to biodiversity, semi-natural habitat cover, or ecosystem service provisioning. To date the only study on benefits to biodiversity from LEAF farming is a study published in partnership with LEAF by Reed et al. (2017), who interviewed LEAF demonstration farmers. These farmers reported several increases to biodiversity seen on their farms, including 66% of participants reporting increases to biodiversity on their farms due to LEAF Marque certification, with one farmer indicating a tripling number of bird species since certification. As this study relies on farmer disclosure to be completely accurate and scientifically sound, this document cannot be used as a source of reference for the benefits to biodiversity LEAF farming brings.

Similarly to LEAF, despite the vast expanse of literature on the comparison of conventional and organic farming practices, relative little is known about the effectiveness of IPM farming (Stenberg, 2017). Within the literature few studies compare organic to IPM, which include Todd et al. (2011) who found that invertebrate diversity was much greater in organic in comparison to IPM kiwifruit orchards in New Zealand. However, in contrast Genghini et al. (2006) found that both IPM and organic had significantly increased levels of biodiversity, in terms of bird diversity on Italian orchards compared to conventional farming. IPM farming's impact on biodiversity is still not as clear as organic, in the current literature.

Low-intensity farming, such as organic farming, greatly promoted habitat diversity during the centuries leading up to the intensification after the Second World War (Tscharnkte et al., 2005). Organic farming is beneficial to species richness and abundance to multiple taxa (Bengtsson et al., 2005), and has been found to benefit biodiversity on a range of farming systems (Feber et al., 1997), from perennial systems in Japan (Katayama, 2016) to annual wheat growing farms in the UK (Hardman et al., 2016), across European and African regions (Schneider et al., 2014), and even provide spill-over effects to sustain diversity in neighbouring conventionally management fields (Henckel et al., 2015). Furthermore, a meta-analysis from Tuck et al. (2014) garnered data from 94 studies over the last 30 years and found a 30% increase in species richness on organic farms. The plethora of studies that support the finding of increased species richness on organic farms indicates that these findings are likely to be true for most organic farms compared to conventional.

Despite academic support, agricultural land under organic management in the UK has decreased from 656,000 hectares in 2011 to 508,000 hectares in 2016 (DEFRA, 2012; 2017). Most farming in the UK is practiced with the use of chemical pesticides.

Although organic farming has not been adopted nationwide, restricted chemical regulations (Hillocks, 2012) and the threat of growing resistance to pests due to the overuse of chemical active ingredients (Oerke, 2006), is a major concern for farmers. Farmers still opt for differing types of management including government-led and industry-led wildlife-friendly or low-input farming practices, over organic. These novel farming practices work with farmers' yield securities on top of trying to support biodiversity through farming practices. The question is whether these alternative management schemes work as well as or can work alongside organic farms to reduce overall chemical input. Can non-organic farm management provide habitats that support the UK's farmland biodiversity, whilst ensuring farmers continue yielding the quantities needed for their business and to meet our growing population's demand? There has been a lack of studies comparing differing management practices' effects on biodiversity under perennial systems (Katayama, 2016); thus, this study uses apple orchards in Herefordshire and surrounding countries in the UK as the study system, to try to tackle these questions.

3.2.3 Objectives of study

This study had the opportunity to look at four types of management practices on 30 apple orchards within a similar landscape in Herefordshire, UK to understand the potential aspects of each management strategy that best supports biodiversity. The four types of farm management are (i) LEAF, (ii) organic (iii) IPM and (iv) conventional farming. By comparing organic and non-organic farms we can compare whether non-organic, but environmentally aware certifications like LEAF, are important influencers on farm biodiversity. This is the first comparison between LEAF, organic and IPM in apple orchards of their ability to support biodiversity in terms of butterfly and bird species richness, abundance, and diversity.

3.3 METHODS

3.3.1 Study system

This study took place between April and October 2015 and 2016 in the west of England. 30 apple orchards were identified as study sites in the counties of Worcestershire, Gloucestershire, and Herefordshire. A full description of the study system can be found in Methods *section 2.1 - Study System.*

Four types of farming systems were identified during farmer interviews: organic, integrated pest management (Hillocks, 2012; Lamichhane, 2017), conventional and LEAF Marque farms (LEAF, 2017). Descriptions of these farming management systems are found in Methods *section 2.2 - Farming Systems.*

3.3.2 Landscape variables

Semi-natural habitat within and surrounding orchards were included during analysis of this chapter to understand if dependant variables and external factors had more of an impact on results than independent variables. These landscape variables are described in Methods *section 2.3.5* as "woody cover".

Whilst the woody cover dataset is the most detailed mapping tool available for identifying areas of woody areas that are not woodland, some hedgerows were missed due to their size. This most likely occurred on farms where hedgerows have been extensively managed, causing the height to be less than 2m. To overcome this, each farm's aerial imagery would need to be manually digitised. Due to the large area of interest, this was not something that could be completed due to time constraints. No hypotheses directly related to hedgerows, so this decision was justified.

3.3.3 Estimating biodiversity

Birds and butterflies were chosen as indicators for biodiversity during this study. Both taxa are commonly used to monitor environmental changes in previous studies due to their susceptibility to land-use changes (Blair, 1999). Both birds and butterflies are UK biodiversity indicators, monitored in order to measure the status of national biodiversity over time (JNCC, 2014). More details of the reasoning behind these biodiversity indicators can be found in Methods *section 2.3.2.*

3.3.4 Bird survey methods

Orchard bird surveys were carried out by three experienced observers, who conducted point count surveys in each farm during the survey periods of 2015 and 2016. Chapter 2 sets out a detailed field survey methodology used for point counts during bird surveys and observer descriptions and training. The field methods used for bird surveys was distance sampling and fully explained in Chapter 2, where *figure 2.3* shows the layout of an example orchard.

The use of a laser range finder was used and explained in Chapter 2. Buckland et al. (2008) highlight the importance of the use of laser rangefinders in distance sampling to ensure the reliability of distance measurements received by the observers. It is still possible to get more accurate distance measurements with aural detections and the location of a tree or hedge can be found rather than estimated by hearing. The use of a laser rangefinder reduces one potential source of error in field methods.

3.3.5 Butterfly survey methods

The survey method used for butterfly counts were 'Pollard walk transects', described fully in Chapter 2 *section 2.3.4.* These were carried out by the author and one additional observer between May and October 2016; with each transect being visited as many times as favourable weather conditions permitted (*section 2.3.4.2)*. The maximum visits per transect was five, with most transects being visited four times. Two sites were only visited once.

3.4 DATA ANALYSIS

All analysis was completed using the statistical programme 'R' version 3.4.3 (R Core Team, 2015). A variety of R packages were used during analysis, including lme4 (Bates et al., 2015), MASS (Venables & Ripley, 2002), VEGAN (Oksanen et al., 2017) and MuMIn (Bartoń, 2017).

Generalised Linear Mixed Effect Models (GLMMs) were used with each dataset to account for both random and fixed effects as predictor variables (Harrison et al., 2017). With each dataset and analysis, a distribution, link function, and structure of random effect(s) were specified to control for pseudoreplication and is described as the "best tool for analysing non-normal datasets" (Bolker et al., 2009: 127; Crawley, 2002).

3.4.1 Abundances

Abundance of bird species here is simply, the total number of birds counted per point count. Each farm type used this data as an average (mean) of bird abundance.

3.4.2 Species richness

Bird species richness was calculated as the number of species per point count. This was averaged for each farm type. Species richness analysis was completed using the VEGAN pack in R. For butterfly analysis, species accumulation curves were created to visualise the data and understand if all the species were sampled, or whether further sampling was needed to capture the communities on each farm management type. *Appendix E* shows the plateau of all four management types in the butterfly species accumulation curve thus, sampling was assumed to be sufficient.

3.4.3 Species diversity

The diversity index used for this analysis is the Shannon-Wiener Index. Simpson's Index of Diversity scores were calculated first in Microsoft Excel per point count however gave higher weight to abundant species (Buckland et al., 2005) such as large numbers of crows, which made those farms look more diverse than others. Shannon-Wiener Index was therefore used to account for rare species to give a more reliable diversity index for the farms in this study.

The calculation for Shannon-Wiener Diversity used was:

$$D = -SUM[(pi) \times \ln(pi)]$$

Diversity scores were firstly calculated per point count and then averaged over farm type.

3.4.4 Model simulation using GLMMs

Total bird abundance, diversity, and species richness were modelled with the four farm categories, considering other variables that may be influencing bird abundance, species richness and diversity by using generalised linear mixed effect models (GLMMs) in R. A Poisson error family was most suited for models of abundance and species richness due to count data as the response variable, and Guassian error family was used for diversity as the response variable is continuous. All models' restrictions to maximum likelihood were increased. Each model had the response variable as the bird and butterfly community metric (abundance, species richness or diversity). Farm identification was modelled with year as nested random effects to account for repeated data at farms across years. Other fixed effect variables deemed ecologically important *a priori,* were included: observer (initials of each field assistant), day number (Julian days), woody cover (to test landscape variables), orchard size (ha), temperature (°C), farm management types (organic, conventional, IPM, LEAF), bird box presence, and apple type (dessert, cider, juice). For butterfly response variables (abundance, species richness and diversity), error structures followed the same as bird community models, with fixed effect variables: woody cover, orchard size (ha), temperature (°C), day number (Julian days), sunshine (%), and also insecticide use (tonnes/ha), as this would likely cause significant impact on butterfly community information. The random effect structure of butterfly models was 'round' nested within 'farm ID'. All continuous fixed effect variables were standardised in R using the 'scale' function.

Correlation was tested between each continuous variable using a correlation coefficient, to test whether variables were too alike to be kept in the model. Woody cover and orchard size were tested and highly correlated. For the purpose of this chapter, woody cover was used over orchard size with bird community models, as this was more likely to be affected through farm management decisions, whereas orchard size cannot be changed; and either woody cover or orchard size was chosen in initial model selection using AIC for butterfly community models.

Likely interactions were tested using Aikike Information Criterion (AIC). Where a model with the interaction was tested against a non-interaction model using anova. The model with the lowest AIC was chosen when the p value showed significant difference from the non-interaction model. When the tested models showed a difference, but the difference was not significant, the model with the lowest AIC and lowest degrees of freedom (DF) were chosen.

Overdispersion was tested for using the dispersion parameter where the residual degrees of freedom were found to be equal, and within the required boundaries (<2), to the residual deviance of the model, thus there was no need to zero-inflate errors to account for this (Crawley, 2007; Harrison, 2014).

The global model was finally put through an automated model selection process, in the package MuMIn, based on information theoretic criteria (detailed below), to fit all subsets of the global model and ranked this suite of models based on Aikike Information Criterion (AIC) (Kamil, 2017).

Collinearity tests were performed on the final fitted model using variance inflation factor (James et al., 2014). This tested the correlation between continuous variables that were chosen to be kept in the model during the correlation coefficient tests above, to make sure all variables were independent from each other and that the information each variable in the model provides about the response variable is not redundant (Bruce & Bruce, 2017). If VIF value of a variable is higher than 5 it is deemed problematic and should be removed from the model to simplify the model and improve model accuracy (Bruce & Bruce, 2017).

3.4.4.1 Model averaging process

An information theoretic (IT) approach was used over the use of a null hypothesis significance testing. This was based on findings from Richards et al. (2011) and Burnham & Anderson (2002), where evidence suggests that using IT approaches when using observation datasets, such as this study, is the most appropriate technique compared to stepwise techniques when external factors are outside experimental control and when testing multiple biological hypotheses are required (Richards et al., 2011).

To obtain models included in model averaging, the global model with all independent variables ecologically likely to affect the dependant variable was constructed. This global model was put through an automated model selection process, in the package MuMIn, based on AIC to create a subset of models that included the independent variables in order of the highest AIC (Kamil, 2017). 'Top models' were chosen out of these subset models using AIC Delta < 6 AIC values, as advised in Harrison et al., (2017).

After these top models were selected, a model average can be acquired with model average parameters including standard errors and confidence intervals taken using the top models. Two model averages are reported: full and conditional (Kamil, 2017; Anderson & Burnham, 2002). To report each model average, the candidate top models and their confidence intervals are displayed using R library "AICcmodavg" (Mazerolle, 2017). The 'conditional' average is chosen over the 'full' average during the model average confidence intervals. The conditional average excludes zeros in the averaging process, whereas the full average includes them. Although both averages were similar in their results, the conditional average is used when constructing model average confidence intervals in preparation to present the overall model average results for each GLMM. The full averages have been reported in appendices.

3.4.5 Density estimates

Density estimates were used to account for species that were not detected during the bird point counts. Using different detection function curves per management type allows us to estimate how many species were detected and undetected in total.

Orchard bird densities were calculated using the free software, Distance *version 7.0 Alpha* (Thomas et al., 2010). Within Distance, the analysis engine used was "Conventional Distance Sampling" (CDS), which assumes that the probability of detecting the species of interest at zero distance from the point count or track line is 1, g(0)=1.

The model definition properties used for each detection function were Half-normal key function with Simple Polynomial series expansion. In data filter properties six intervals of 12.5 were manually selected to amend for rounding during distance estimation in the field. The data collected was truncated in Distance at 75m instead of 100m, which was the in-field truncation value. The value of 75m was chosen as distances above this had higher degrees of rounding and therefore less likely to be as accurate as nearer distances. Truncation was an effort to reduce distance estimation errors.

Model selection was based on minimising the AIC and model parameters.

This survey design involved stratification per farm, where point counts were allocated proportionally, depending on size of farm. Due to time restrictions on the BTO breeding bird survey methodology no more than 10 point counts could be surveyed per morning. To keep the bird surveys to the breeding period and survey each point count twice, field surveys were restricted to three months. These restrictions meant some of the smallest farms could only have one point count on. In order to estimate population densities and abundances using Distance, a minimum number of 15 point counts is necessary (Buckland et al., 2006). Grouping farms into their farm type (i.e. organic, LEAF, conventional or IPM) allowed this minimum number per stratum to be reached, thus the following results have been calculated per stratum, not per farm or orchard. A separate detection function could be fitted to each stratum, as observations reached over 60 per stratum, recommended by Williams & Thomas (2006).

In order to compare total bird densities on different farm types the difference in densities using variance has been estimated, as described in Buckland et al. (2001; section 3.6.5). Degrees of Freedom (df) are already approximated by Distance, using Satterthwaite's approximation, which can be used to obtain a z-statistic (Buckland et al. 2001). The equation used to obtain a z-value:

$$Z = \frac{D_1 - D_2}{\sqrt{\mathrm{Var}\,(D_1 - D_2)}}$$

These Z values were converted to P-values in R statistical programme (R Core Team, 2015), and then double checked using Z tables, to test whether those differences were significant.

3.5 RESULTS

3.5.1 Bird abundance, diversity and species richness

Over the two years of bird surveys, between 2015 and 2016, 7,178 individual birds were recorded and 66 bird species. A full record of bird species can be found in *Appendix C*. Organic orchards were found to support significantly higher bird abundance (*figure 3.1*) diversity and species richness than conventional farms. Organic farm management showed the most significant influence on bird abundance (*figure 3.2*), species richness (*figure 3.3*) and diversity (*figure 3.4*). Tukey post-hoc analysis between other farm categories and their p-values are noted on *figure 3.5.1* to show differences between farm management types to show that the only significance is between organic and other managements, with no significant variation between LEAF, IPM and conventional.

Although organic orchards hosted a higher abundance, diversity and richness of birds, it was important to understand what other factors might also be influencing biodiversity indices and whether it was just farm category that was affecting them. After considering a host of non-controllable factors, the variable that had the biggest impact on the increase in bird biodiversity indices was the farm management category. In other words, the type of orchard management was the biggest factor affecting bird abundance, diversity, and species richness. Below are more detailed reports of each biodiversity indices with their respective model average coefficient summaries. The patterns displayed with bird abundance, species richness and diversity are due to the type of farm management, not other external influences on or surrounding the farm.

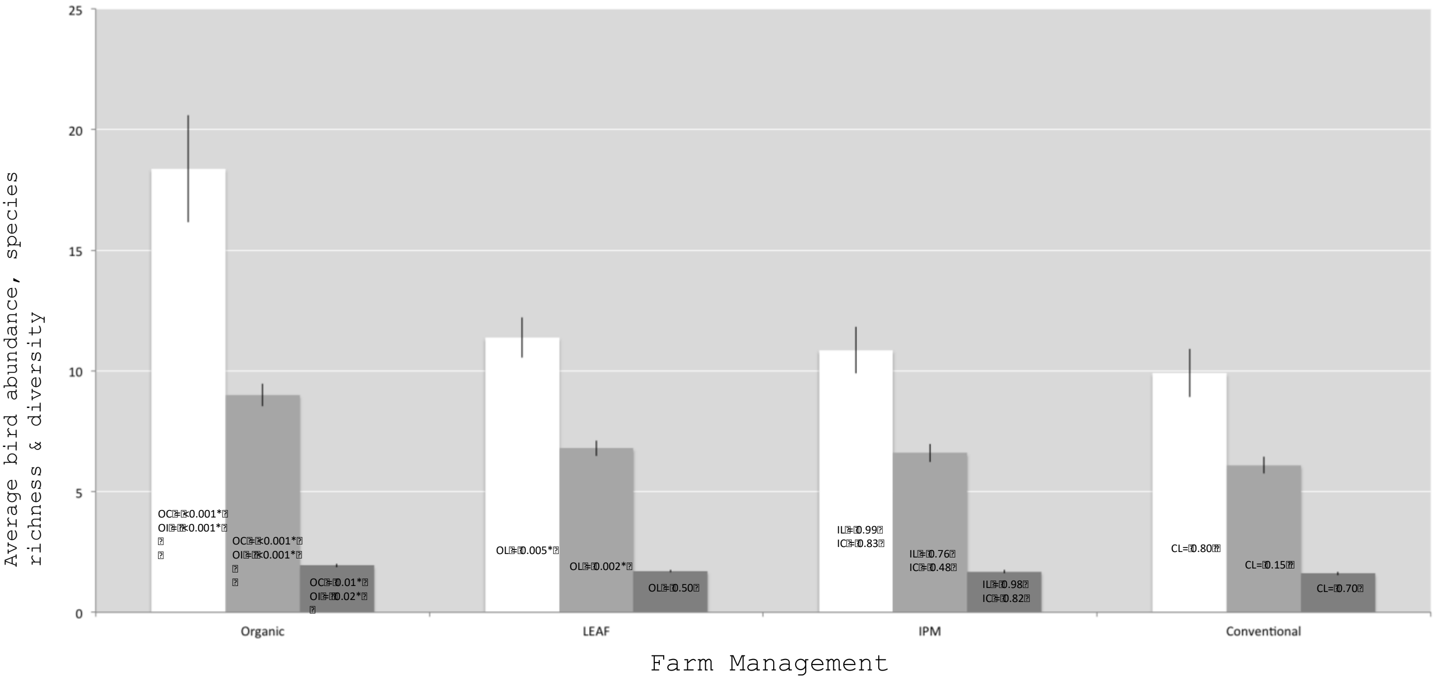


Figure 3.1 Mean bird abundance (white), mean Shannon diversity (dark grey) and mean bird species richness (light grey) per farm category with standard errors. Four management categories in experiment: Org = organic, Conv = conventional, IPM = Integrated Pest Management, LEAF = Linking Environment and Farming. Numbers inside the columns are the p-values of Tukey Post Hoc analysis results comparing farm management categories to LEAF and organic (OL = organic and LEAF; CL = conventional and LEAF; IL = IPM and LEAF; OC= organic and Conventional; OI = organic and IPM; IC = IPM and conventional).

3.5.1.1 Bird abundance

Organic orchards are shown here to hold a significantly higher abundance of birds (*Figure 3.2*). Increased Woody Cover had no significant effect on bird abundance (*Figure 3.2*). There is an observer effect that shows both observers here recorded significantly fewer birds than the intercept (Observer CP), but this effect is considered and continues to show organic orchards support high bird abundances. Day number was also significant in increasing bird abundance, thus as day number increased throughout the survey season, so did the abundance of birds.

No other variable had a significant effect on bird abundance, as shown by the coefficients used in the model average summary (full coefficients table in *Appendix D*). Neither juice, dessert or cider orchards have significant impact on bird abundance, in other words the type of apple grown does not impact bird abundance (*Figure 3.2).*

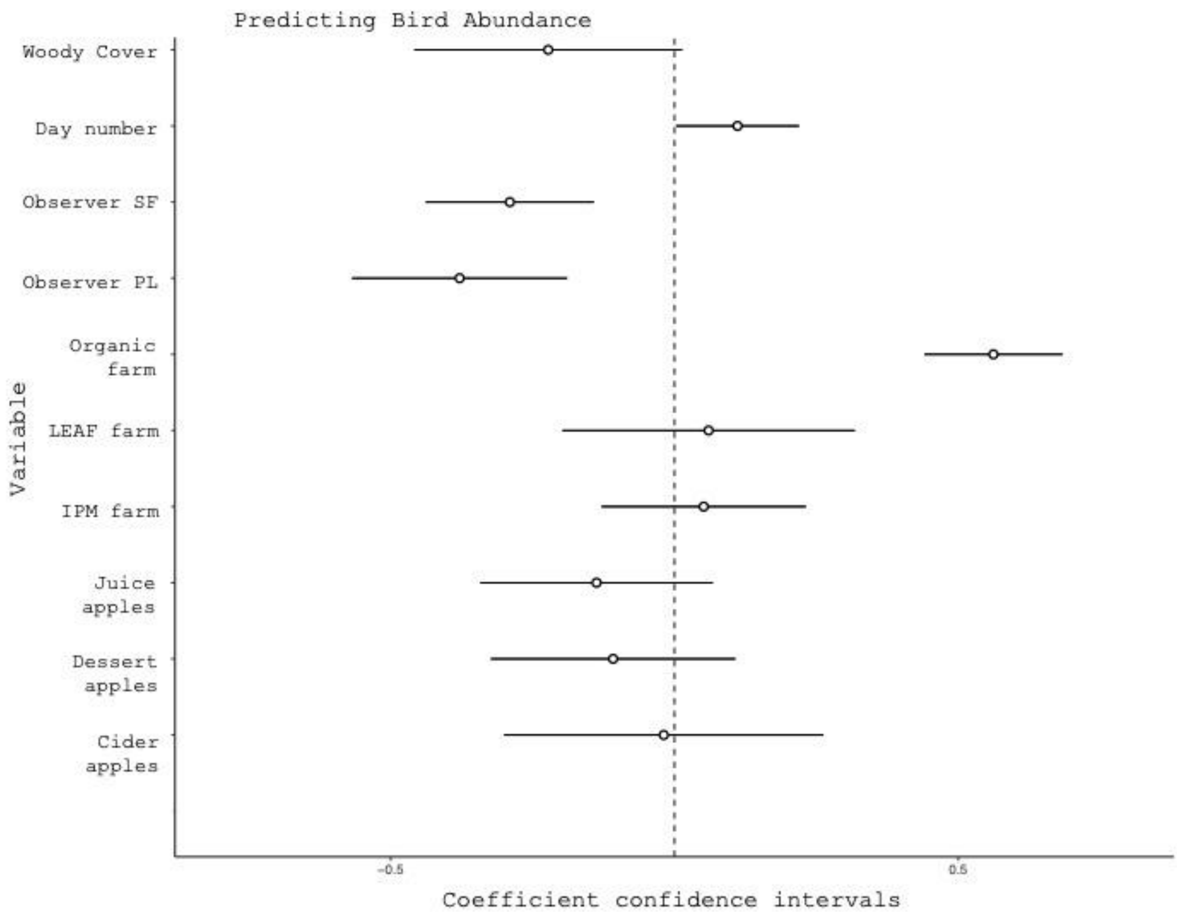


Figure 3.2 Bird abundance model average with Delta < 6 coefficients confidence intervals, taken from top models used for conditional model average (full model average in *Appendix D*). These coefficients are the difference from the intercept, (farm category 'conventional' with observer 'CP').

From these coefficients we can see farm category 'organic' was the most influencing factor to bird abundance in apple orchards.

3.5.1.2 Bird species richness

Bird species richness gave a similar pattern across farm management categories with organic orchards supporting significantly higher species richness than conventionally managed orchards (*figure 3.3*).

Although bird box presence improved the model fit during initial model analyses, there was no significant impact of bird box on bird species richness. Woody cover had no significant effect on the model output for species richness. An observer effect is seen with two observers: Observer SF and Observer PL, who both captured significantly less bird species richness during their surveys. This observer effect is taken into account in the model, and although significant, farm management still had a significant effect on bird species richness.

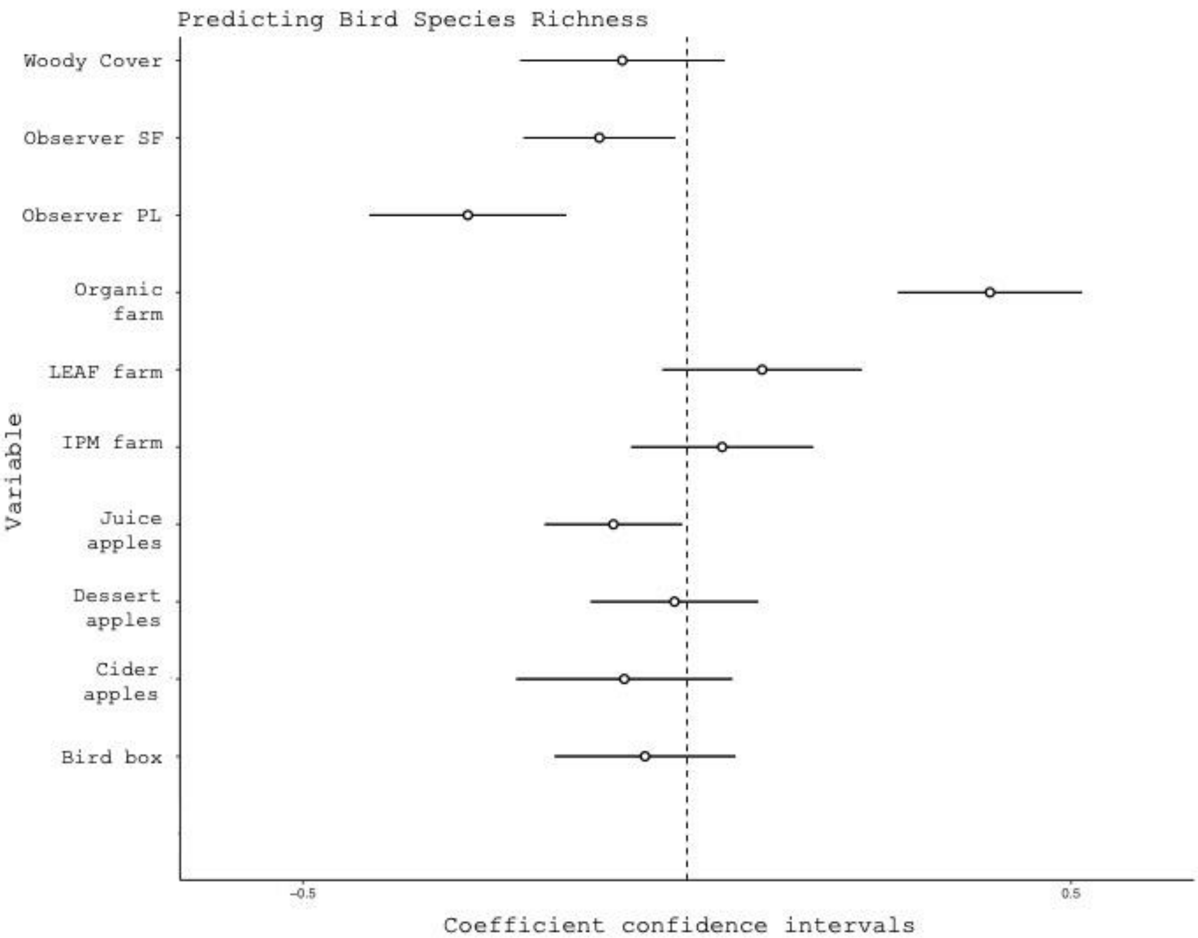


Figure 3.3 Bird species richness model average delta < 6 coefficients confidence intervals, taken from conditional model average results (full model average in *Appendix D*). These coefficients are the

difference from the intercept, (farm category 'conventional' with observer 'CP'). From these coefficients we can see farm category 'organic' was the most influencing positive factor to bird species richness in apple orchards.

3.5.1.3 Bird diversity

Similar to abundance, Shannon diversity of bird species was significantly higher on organic orchards (*figure 3.4*). No other farm management group had a significant effect on bird diversity. Both woody cover and orchard size were not significant during model selection so were not included in the global model for model averaging. Observer PL recorded significantly less bird diversity than the intercept (Observer CP), whereas there was no difference with Observer SF.

Temperature was the only other variable that significantly affected bird diversity other than farm management type, however this was a negative relationship. Woody cover was not significant in explaining the model during model selection, so this variable was removed in the global model before model averaging.

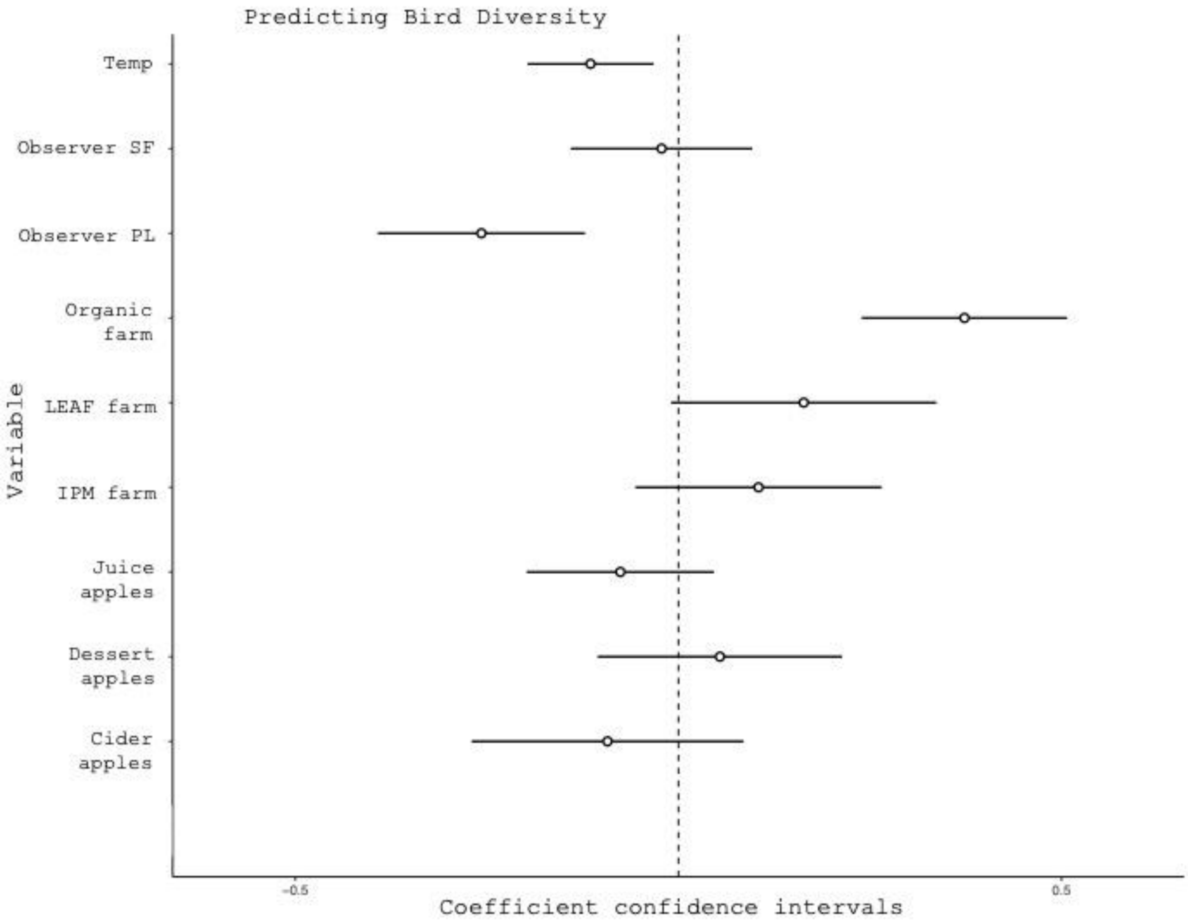


Figure 3.4 Bird diversity model average delta < 6 coefficients confidence intervals, taken from conditional model average results (full model average see *Appendix D*). These coefficients are the difference from the intercept, (farm category 'conventional' with observer 'CP'). From these coefficients we can see farm category 'organic' was the most influencing positive factor to bird diversity in apple orchards.

3.5.2 Bird density

The density of birds per hectare is shown in *figure 3.5*, with organic farms displaying the highest density of birds across all four farm management categories, with statistical significance. LEAF and conventional farms were not significantly different from each other, but IPM had slightly significant results during Z tests. Although Z tests shown on graph compare only to conventional farming, IPM and LEAF were both very similar to each other. Organic was significantly greater than both IPM (*Z=6.02; P<0.01*) and LEAF (*Z=6.13; P<0.01*). Density detection functions are given in *Appendix G* and support the previous findings using a different monitoring technique.

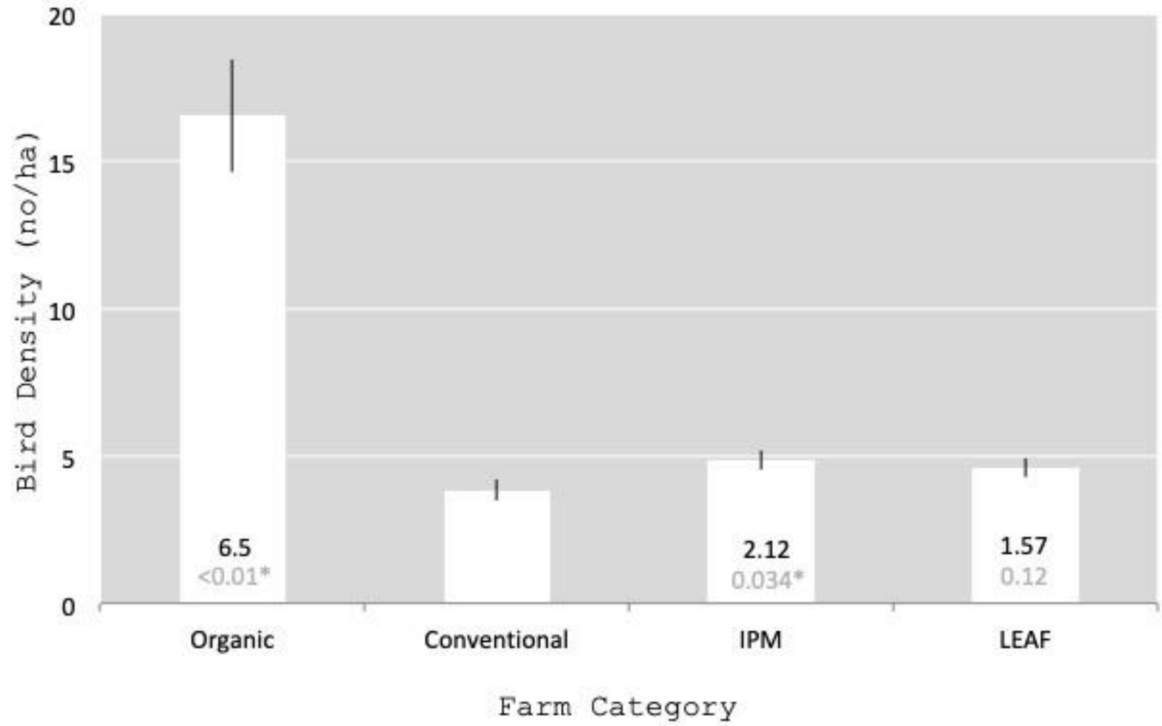


Figure 3.5 Total bird density shown as number of birds per hectare, results from Distance analysis with standard error bars. The numbers within the bars indicate the Z (black) and P values (grey) during two-way comparisons, between conventional farming to other three categories (LEAF, IPM and organic).

3.5.3 Butterfly abundance, species richness and diversity

During the butterfly survey year in 2016, 1,372 butterflies were recorded, and 20 species were identified. A full record of butterfly species can be found in *Appendix B*. Organic orchards supported higher bird abundance, species richness and diversity than other farm categories, but these were only found to be significant between organic orchard abundance and IPM, and conventional and IPM. Like birds, butterfly abundance showed the biggest difference between the farm management categories. Tukey post-hoc analysis p-values highlight that difference between farm categories were found to be insignificant, with only organic - IPM and then IPM - conventional showing significance difference, where IPM held less butterfly abundance than conventional (*figure 3.6*).

Further analysis was undertaken to include non-controllable factors including sunshine, as this is an important variable to include for butterflies. The variables with the biggest impact on the increase in butterfly biodiversity indices were sunshine, day number, temperature and orchard size. Here, the type of orchard management was *not* the biggest factor affecting butterfly abundance, diversity and species richness and instead other external factors were more important. Below is a more detailed report of each biodiversity indices with their respective co-efficients summaries.

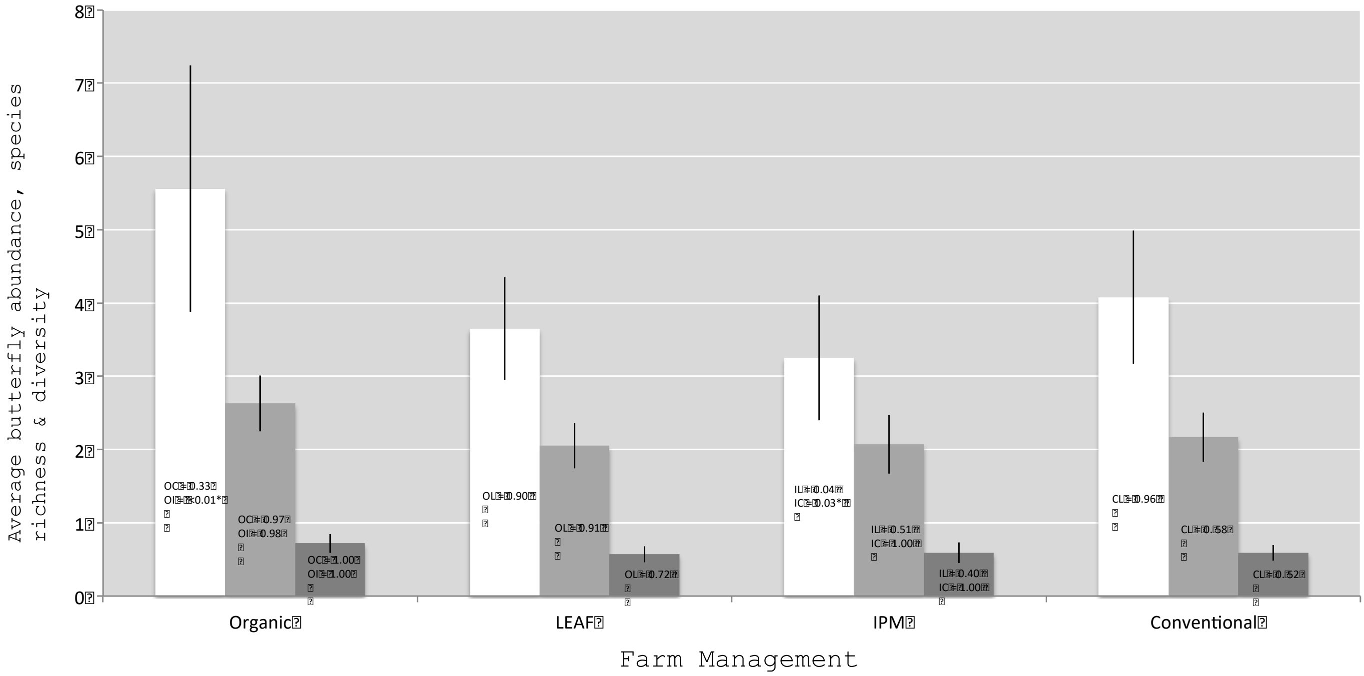


Figure 3.6 Mean butterfly abundance (white), mean butterfly species richness (light grey), mean Shannon diversity (dark grey) per farm category with standard errors. Numbers inside the columns are the p-values of Tukey Post Hoc analysis results comparing farm management categories to each other (OL = organic and LEAF; CL = conventional and LEAF; IL = IPM and LEAF; OC= organic and conventional; OI = organic and IPM; IC = IPM and conventional

3.5.3.1 Butterfly abundance

No farm management was significant in positively influencing butterfly abundance (*figure 3.7)* and IPM orchards had significantly less butterfly abundance than conventional orchards and organic orchards (*figure 3.7*). However, an interaction between woody cover and farm management highlights that IPM orchards with higher woody cover have significantly less butterflies than conventional orchards with higher woody cover. Furthermore, an interaction with orchard size and farm management highlight that large organic orchards have less butterfly abundance than large conventional orchards. External factors, day number and sunshine had significant positive effects on butterfly abundance. No observer effect is evident during butterfly data collection.

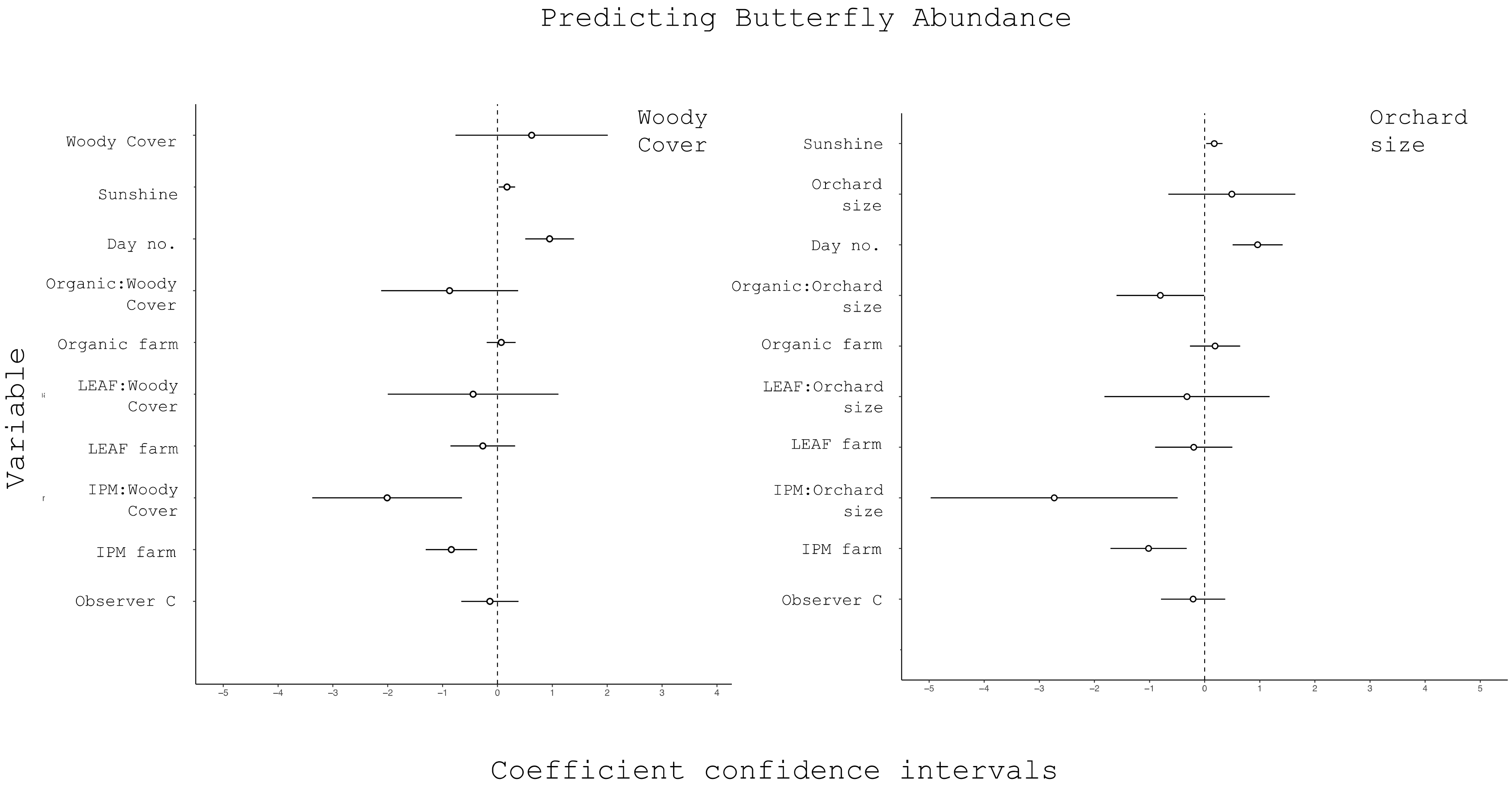


Figure 3.7 Butterfly abundance model average delta < 6 coefficients confidence intervals, taken from conditional model average results. These coefficients are the difference from the intercept, (farm category 'conventional' with observer 'K'). Butterfly abundance had a management and woody cover interaction (left), which was significant in the butterfly abundance model comparisons, so this was the final model that was used for the model average. The management and orchard size interaction model (right) was found to be significant in model selection, although orchard size alone was not. Random effect structure is round nested within farm ID. Full model average summaries are given in *Appendix F.*

3.5.3.2 Butterfly species richness

Butterfly species richness was significantly influenced by sunshine, temperature, and day number in this model average. No other variables kept in the global model was significant to species richness (*figure 3.8*). Although, 'total insecticides' (broad and specific combined) were kept in the model during the model selection process, this was not a significant influential parameter in the model average. Woody cover was not included in the global model due to lack of significance in the model fit.

Neither Woody cover nor orchard size interactions were significant in the model selection process so interactions were not included with butterfly species richness.

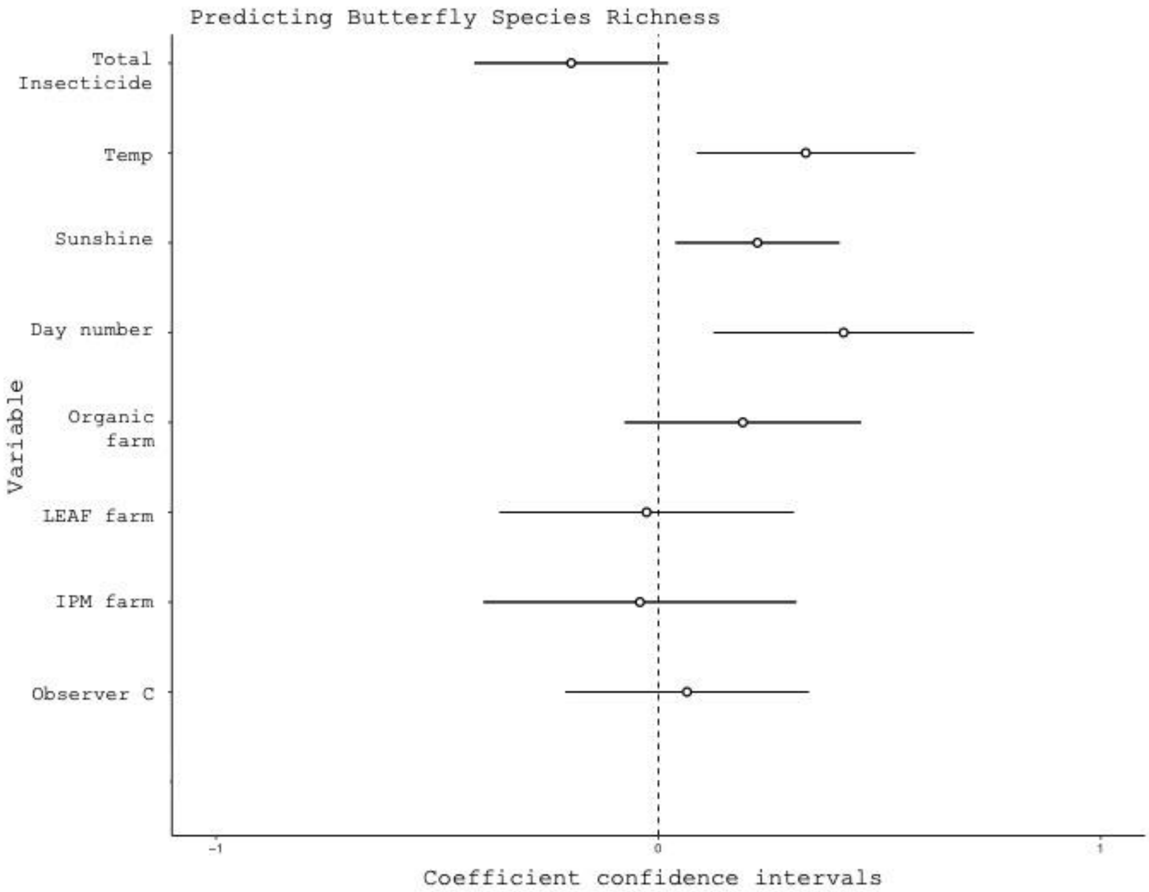


Figure 3.8 Butterfly species richness model average delta < 6 coefficients confidence intervals, taken from conditional model average results. These coefficients are the difference from the intercept, (conventional management and Observer 'K'). Random effect structure = (farmer / round). Full model average summaries are given in *Appendix F.*

3.5.3.3 Butterfly Diversity

Shannon diversity of butterflies saw no significant influences from farm management types. Only day number and sunshine had a significant positive impact on butterfly diversity. Insecticides also shown here (*figure 3.9)* are the 'total' of broad spectrum and specific insecticides that although a negative relationship is shown between insecticide use and butterfly diversity, this is not significant. Interactions between management and orchards size, and management and woody cover were tested and were not significant to the model fit, so were not included in the global model.

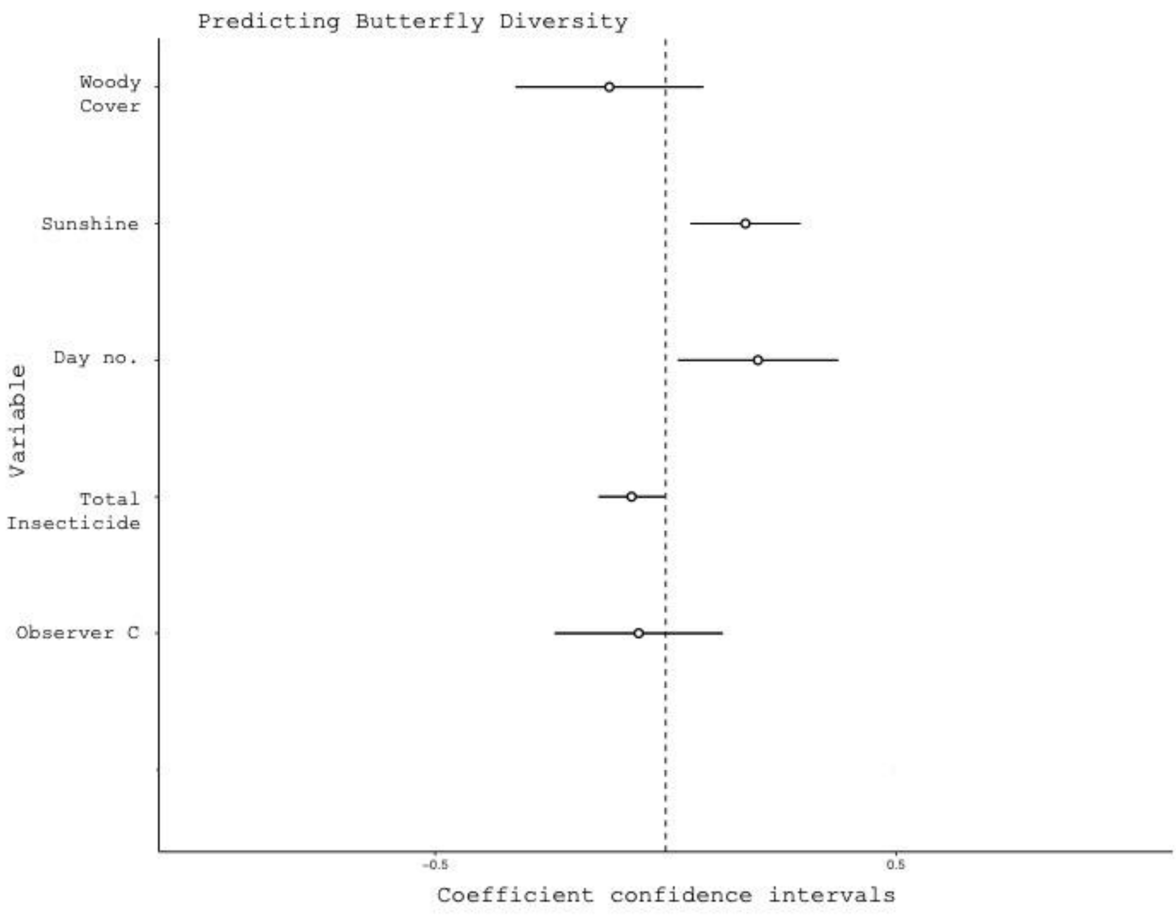


Figure 3.9 Butterfly diversity model average, delta < 6 coefficients confidence intervals, taken from conditional model average results. These coefficients are the difference from the intercept, (Observer 'K' and conventional management). Random effects structure = (farmer / round). Here, management type was not included by the 'top model' chosen through AIC, delta <6. Full model average summaries are given in *Appendix F.*

3.6 DISCUSSION

3.6.1 On-farm biodiversity on four farm management options

Organic farms supported a higher abundance, density, species richness and diversity of birds over all other farm management types. This is consistent with a multitude of studies (Hole et al., 2005; Hardman et al., 2016; Katayama, 2016). Farms which have higher inputs of chemicals applied during the breeding season have been shown to have negative direct impacts on bird species (Boatman et al., 2004), which is a possible reason organic farms, although not altering their habitat, still supported higher bird abundance, density, species richness and diversity than LEAF, IPM and conventional farms.

The type of apple (dessert, cider, or juice) was included in all analysis and was shown to have no importance in bird abundance, species richness and diversity. These results are contrary to the literature, where more mature trees that are older and taller, found on cider orchards (Robertson et al., 2012), host more insects under the matured bark, giving more shelter and food sources for birds. Apple orchards, especially traditional orchards, which are not managed on a commercial scale, can provide habitat for birds, as they have in Colorado, USA (Mangan et al., 2017) and are recognised as a priority habitat for birds by Natural England under the UK Biodiversity Action Plan (Natural England, 2010). However, we have not shown that particular apple types hold more or less bird abundance, species richness or diversity but it is important to recognised that majority of organic orchards in this study were in fact cider apple trees in traditional orchard settings.

An observer effect is shown in bird abundance, species richness and diversity. Where two observers find less bird abundance and species richness than the intercept (Observer CP), and PL finds less that SF and CP diversity. Although these observer effects have been taken into account by the model itself to be able to show the effect of farm management, it is worth noting there is still an observer effect present. The observer CP was the older of the three bird observers and seems to consistently find higher bird abundance even though they all have similar experiences. Without knowing exactly what happened during CP's surveys other than the initial training I conducted, this observer effect is unable to be explained. A way to avoid these discrepancies in future work would be to use two observers on each survey so to avoid an observer being "swamped with bird records" in the short survey window available (Buckland et al., 2008: 95). Timekeeping may also be ensured with two observers, and insurance that birds outside of the 100m distance range were not being included in the records. Even though an observer effect is present, the model average still shows organic farm management held significantly higher bird abundance, density, species richness and diversity than other farm management.

Contrary to our expectations, butterfly abundance, species richness and diversity was not affected by farm management, this has also been supported by others in the literature (Kleijn et al., 2006). Significantly less abundance of butterflies were found on IPM farms compared to conventional, but otherwise our mixed models highlight only external variables, such as temperature, sunshine and seasonality, significantly influenced butterfly communities on apple orchards. Although many studies support the notion that organic orchards hold higher butterfly abundance than conventional farms using pesticides (Rundlöf & Smith, 2006; Feber et al., 1997), our study shows that conventional orchard butterflies are not significantly different from organic, which does not support the majority of the literature.

One explanation for the lack of significant increase in butterfly abundance, species richness or diversity on organic orchards could be the permitted use of copper sulphite on organic farms. We have shown here that orchard size and farm management interact to effect butterfly abundance: large organic orchards hold significantly less butterfly abundance than large conventional orchards. Two organic orchards in our study were more commercial and productive orchards (the largest orchards), using copper as a substitute for insecticide and fungicide. This is permitted under Soil Association organic standards, however there has been controversy over its use as to whether it causes soil contamination, especially as the frequency of copper sprays are not restricted (Stolze et al., 2000). Seven of the nine organic farms, although following the Soil Association guidelines, were not certified organic as they did not benefit from the certification financially. These were the smallest orchards, who chose a zero-input management, without the use of copper sprays too. This result could be explained also due to the small field sizes of organic farms, where smaller fields have been linked to higher habitat and species diversity due to the farmed area - perimeter ratio (Norton et al., 2009; Hardman et al., 2016).

Although species accumulation curves (*Appendix E*) suggest no further sampling of butterflies was needed, the year butterflies were sampled (2016) was known to be the worst year on record since the citizen science “Big Butterfly Count” started, in 2010 (Butterfly Conservation, 2016). As butterflies were sampled only within this year, we were not able to compare across years and can only assume that our results may be due to extremely low butterfly abundance, nation-wide.

In summary, organic farming increases species richness, diversity, and abundance of birds, but the signal is not the same for butterflies. These findings are common with other studies, in that effects of farm management differ across taxa (Birkhofer et al., 2014; Bengtsson et al., 2005). Organic apple orchards have shown here to provide habitat to support diverse and abundant bird communities (Mangan et al., 2017). Furthermore, LEAF and IPM, although promoted as environmentally focussed farming, do not show any significant difference, in terms of how their management impacts on-farm biodiversity, to conventional farming; implying that guidelines may not be sufficiently enforced or followed.

3.6.2 Impacts of woody areas on farm biodiversity

Within LEAF management hedgerows, coppices, woodlands, and shrub areas area advised to be kept with minimal management and not removed from the landscape (LEAF, 2017). One of the recommended practices for farmers to follow states there should be “a minimum of 5% farm area available as habitats, not used for cropping and food production” (LEAF, 2017: 48). During this study we tested whether on-farm woody cover, and the immediate surrounding area (100m), influences bird and butterfly abundance and species richness.

The woody cover variable was able to identify any shrubs above 2m in height, therefore identifying hedges and shrub areas (Tebbs & Rowland, 2014). Small-scale landscape features such as these have only recently been developed by Kings College, where previously only larger expanses of woodland would be included as the wider landscape variable from Landsat. While some studies have shown that when there is more heterogeneous landscape there is higher biodiversity (Henderson et al., 2012), we find no correlation between increased on-farm and immediate surrounding woody cover with increased bird diversity, species richness, or diversity.

Immediate landscape variables, including hedgerows and woodland areas within the farm and adjacent (up to 100m), are not important at conserving biodiversity on orchards. Woody cover was not a significant farm feature in determining the abundance, species richness, and diversity of birds. Woody cover was also included as a landscape variable for butterfly community analysis. This on-site woody cover was not significant in any model as a single parameter for butterflies. Interactions were tested to understand if a particular farm type with higher woody cover changed butterfly composition. Only IPM orchards showed significance here, where IPM orchards with more woody cover supported less butterfly abundance. This was a significant finding that contradicted our expectations: increased availability of suitable habitat increases the abundance of butterflies (Steffan-dewenter & Tscharntke, 2000).

Findings here differ from other studies (Benton et al., 2003; Pärt & Söderström, 1999), as orchards, especially those unmanaged (Natural England, 2010), should naturally produce food sources and shelter for birds so one would expect higher bird community assemblages. Results found here mirror results found from an Italian orchard study measuring bird diversity in respect to hedgerows and woodlots (Genghini et al., 2006). The type of management that a farm has decided to practise has been shown here to be more important than landscape features in terms of supporting birds, where organic and zero-input orchards support the highest biodiversity. However, this study only took immediate adjacent landscape and on-farm landscape into consideration.

Furthermore, the woody cover tool struggled to pick up some very well-managed hedgerows (Emma Tebbs, *Pers Comms*, 3rd March 2017), so these habitat areas were absent during the analysis. Although this study didn't take into account wider landscape effects, it has been shown that landscape variety plays a more significant role than farm management type for butterfly species richness, diversity and abundance, also in conjunction with other studies including Weibull et al. (2000; 2003) and Benton et al. (2003). Future studies should include larger areas to understand this variable in greater detail, especially considering that the study area of Herefordshire and surrounding counties have a diverse landscape in comparison to arable landscapes in the UK.

3.6.3 Implications for future farm management

There has been a growing concern in the farming community over the uncertainty of future pesticide availability. Popular insecticides have been discontinued in European farming. Most recently, at the time of writing, the commonly used Chlorpyrifos was discontinued due to risk to human health during application (HSE, 2016). Although not the main reason for the ban, this chemical has negative effects on beneficial predatory insects on farms, such as the European Earwig *Forficula auricularia L.* (Fountain & Harris, 2015). The destruction of natural enemies by using insecticides is not a new insight and is known to cause pest outbreaks within crops, such as the examples in apple and cotton crops given by Pimentel and co authors (1992).

Insecticides are used to reduce crop pests and obtain a high yield - which is generally found to be lower in low-input farming such an organic or similar agri-environment schemes (Ekroos et al., 2014; Gabriel et al., 2013). However, a growing number of studies have disproved this (Zhang, 2016; Pergola et al., 2013; Mäder et al., 2002; Harpinder et al., 2007). With yields at similar levels as non-organic farming, less chemical reliance, and lower-input farming may be a viable, attractive and a realistic management option to use. Although pesticide costs were not analysed for this study, we collected data between 2015 and 2016 to give an average cost incurred by farmers for those who chose to use insecticides. The difference between the first year of spraying, with an average of £1658.52, and second year, averaging £2,778.45, was due to the EU ban on Chlorpyrifos, which was a cheap, broad spectrum chemical used by many farmers (HSE, 2016). The alternative chemicals available are more expensive, specific insecticide, thus multiple chemicals needed to be used to replace Chlorpyrifos. This uncertainty and increasing expense for farmers may force farmers towards a low chemical management alternative which, although may initially increase pest numbers in the short term before natural enemy numbers increase, after also being affected by the insecticides (Fountain & Harris, 2015), will alleviate the increasing costs and risks of spraying insecticides.

Although IPM was developed to tackle this problem and farms are required to spray a reduced amount of insecticides (Barzman et al., 2015), they are not reducing the impact of chemical management as much as organic in this study. These results are in line with Todd et al. (2011), which find organic invertebrates have higher numbers than IPM invertebrates on New Zealand kiwifruit farms. However, our results are in conflict with Genghini et al. (2006), where reduced spray and no spray saw significant increases in bird diversity on Italian orchards than conventional. The lack of scientific rigor (Stenberg, 2017) and IPM farm checks may be the reason no different between conventional and IPM is shown here, in terms of biodiversity on farms. Other than agronomist advice, there is no guideline to spray under a certain threshold of insecticides, which are likely to leave IPM and conventional spraying at similar amounts of chemicals per hectare. Furthermore, only farms using level 1 IPM were considered but it has been demonstrated there is no difference between level 1 IPM and conventional (Kogan, 1998). To further test IPM in its entirety, all three levels of IPM farms would need to be selected in future studies. Farmer self-diagnosis of farming system may be another reason for this similarity. As IPM does not have clear guidelines or a certification system, some conventional farms could already be practicing IPM at level 1 without realising. To overcome this in future studies, a detailed outline of every management strategy should be discussed with the farmer prior to deciding on farm categorisation.

LEAF is promoted as an alternative and viable option for farmers who may not wish to be fully organic but want to start thinking about sustainability across their whole farm area. Leading supermarkets, such as Waitrose, pay premium prices for LEAF Marque produce, indicating its perceived value to supporting wildlife. The practices promoted through LEAF, such as providing nesting habitat, managing margins “sympathetically” and providing winter and summer food for birds (LEAF, 2017: 46), have not been sufficient in increasing bird or butterfly abundance, species richness, or diversity to levels significantly higher than conventionally managed orchards. These types of practices have been proven by over 35 studies, based in UK agriculture, and then further scrutinised to conclude that the practises LEAF encourage are highly likely to benefit bird conservation (Sutherland et al., 2018). This suggests a disconnect between LEAF advice on practices to implement for nature conservation, and the management practices tangibly put in place by the farmer.

Although there has been a magnitude of studies comparing different farming methods, new practices are constantly arising which seek to work with farmers as well as trying to support biodiversity. The growing food crisis has not yet been solved and pressure mounts upon farmers to produce more food with less land at the cheapest prices. Moreover, the prospect of the UK exiting the European single market leads to uncertainty regarding the future of CAP payments to UK farmers (Swinbank, 2016). Moving away from government funded schemes to increase biodiversity, and towards market driven schemes like LEAF, organic and Conservation Grade, allows products that incorporate environmental practices to be marketed by supermarkets at slightly a higher price (Firbank et al., 2013), as consumers are willing to pay for the perceived health and environmental benefits that are associated with the label (Batte et al., 2007). This incentive may be more risk-averse than the future of CAP or insecticides. It is evident within the literature that higher diverse ecosystems support important ecosystem services (Tscharntke et al., 2005), such as pest control and pollination. Managing a farm with a biodiversity focus, may lead to more opportunities in the market, through organic and LEAF, however these market opportunities must outweigh potential loses in yield for biodiversity-friendly farming to remain a viable farming option (Crowder & Reganold, 2015; Reganold & Wachter, 2016).

Changing just one farming practice will not enhance all taxa on farmland, and although these butterfly results were inconclusive it is worth noting that trying to optimise conservation actions or management practices for species richness for one taxa, may not lead to increased species richness for all taxa (Weibull et al., 2003). A holistic approach to management across the whole farm, as LEAF promotes, along with reduced chemical use, as organic promotes, may be an optimum scenario, which needs to be explored further.

Furthermore, in terms of supplying the global population with food, we may require not just a change to the way we farm our food but, cultural shifts to our diets, eating habits, food waste and food distribution – to equally distribute food to the world's growing human population (Godfrey et al., 2010). This is something that farmers do not have the power to do, but multinational corporations and governments, who decide upon market-led or government agri-environment schemes, should strive to address alongside the way we farm our food.

3.6.4 Conclusion

Four distinctive farm management approaches were examined to identify their ability in supporting biodiversity within UK apple orchards. Organic orchards supported the highest level of biodiversity where bird abundance, species richness, and diversity were significantly higher than conventional orchards. Butterfly biodiversity metrics were very weak and did not show any difference between farm management types, only external factors such as temperature and sunshine were of significance. Orchards using a LEAF and IPM farm management approach showed that bird abundance, species richness, and diversity did not differ significantly to conventional farming. The use of pesticides in farming has shown reductions to the biodiversity across Europe (Geiger et al., 2010), and this study supports that.

This work supports the plethora of studies suggesting that organic farming supports farmland bird biodiversity. Furthermore, it is the only known study to compare LEAF farming practices with organic, IPM and conventional orchards, and found no significant evidence to suggest that LEAF can promote both optimum yield as well as farmland biodiversity in a greater capacity than what conventional farming can provide.

3.7 REFERENCES

Aue, B., Diekötter, T., Gottschalk, T. K., Wolters, V., & Hotes, S. (2014). How High Nature Value (HNV) farmland is related to bird diversity in agro-ecosystems – Towards a versatile tool for biodiversity monitoring and conservation planning. *Agriculture, Ecosystems & Environment*. http://doi.org/10.1016/j.agee.2014.04.012

Balmford, A., Green, R., & Phalan, B. (2012). What conservationists need to know about farming. *Proceedings of the Royal Society B: Biological Sciences*, *279*(1739), 2714–2724. http://doi.org/10.1098/rspb.2012.0515

Bartoń, K. (2017). *MuMIn: Multi-Model Inference. R package version 1.40.0.*

Barzman, M., Bàrberi, P., Birch, A. N. E., Boonekamp, P., Dachbrodt-Saaydeh, S., Graf, B., … Sattin, M. (2015). Eight principles of integrated pest management. *Agronomy for Sustainable Development*, *35*(4), 1199–1215. http://doi.org/10.1007/s13593-015-0327-9

Batáry, P., Báldi, A., Kleijn, D., & Tscharntke, T. (2011). Landscape-moderated biodiversity effects of agri-environmental management: a meta-analysis. *Proceedings. Biological Sciences / The Royal Society*, *278*(November), 1894–1902. http://doi.org/10.1098/rspb.2010.1923

Bates, D., Maechler, M., Bolker, B., & Walker, S. (2015). Fitting Linear Mixed-Effects Models Using lme4. *Journal of Statistical Software*, *67*(1), 1–48. http://doi.org/10.18637/jss.v067.i01

Batte, M. T., Hooker, N. H., Haab, T. C., & Beaverson, J. (2007). Putting their money where their mouths are: Consumer willingness to pay for multi-ingredient, processed organic food products. *Food Policy*, *32*(2), 145–159. http://doi.org/10.1016/j.foodpol.2006.05.003

Bengtsson, J., Ahnström, J., & Weibull, A. C. (2005). The effects of organic agriculture on biodiversity and abundance: A meta-analysis. *Journal of Applied Ecology*, *42*, 261–269. http://doi.org/10.1111/j.1365-2664.2005.01005.x

Benton, T. G., Vickery, J. a., & Wilson, J. D. (2003). Farmland biodiversity: Is habitat heterogeneity the key? *Trends in Ecology and Evolution*, *18*(4), 182–188. http://doi.org/10.1016/S0169-5347(03)00011-9

Benton, T. I. M. G., Bryant, D. M., Cole, L., & Crick, H. Q. P. (2002). Linking agricultural practice to insect and bird populations: a historical study over three decades. *Journal of Applied Ecology*, *39*, 673–687.

Bianchi, F. J. J. a, Booij, C. J. H., & Tscharntke, T. (2006). Sustainable pest regulation in agricultural landscapes: a review on landscape composition, biodiversity and natural pest control. *Proceedings. Biological Sciences / The Royal Society*, *273*(1595), 1715–27. http://doi.org/10.1098/rspb.2006.3530

Billeter, R., Liira, J., Bailey, D., Bugter, R., Arens, P., Augenstein, I., … Edwards, P. J. (2008). Indicators for biodiversity in agricultural landscapes: A pan-European study. *Journal of Applied Ecology*, *45*(1), 141–150. http://doi.org/10.1111/j.1365-2664.2007.01393.x

Birkhofer, K., Ekroos, J., Corlett, E. B., & Smith, H. G. (2014). Winners and losers of organic cereal farming in animal communities across Central and Northern Europe. *Biological Conservation*, *175*, 25–33. http://doi.org/10.1016/j.biocon.2014.04.014

Blair, R. B. (1999). Birds and butterflies along an urban gradient: Surrogate taxa for assessing biodiversity? *Ecological Applications*, *9*(1), 164–170. http://doi.org/10.1890/1051-0761(1999)009[0164:BABAAU]2.0.CO;2

Boatman, N. D., Brickle, N. W., Hart, J. D., Milsom, T. I. M. P., Morris, A. J., Murray, A. W. A., … Robertson, P. A. (2004). Evidence for the indirect effects of pesticides on farmland birds. *Ibis*, *146*(2), 131–143.

Bolker, B. M., Brooks, M. E., Clark, C. J., Geange, S. W., Poulsen, J. R., Stevens, M. H. H., & White, J. S. S. (2009). Generalized linear mixed models: a practical guide for ecology and evolution. *Trends in Ecology and Evolution*, *24*(3), 127–135. http://doi.org/10.1016/j.tree.2008.10.008

Bruce, P., Bruce, A. (2017). Practical Statistics for Data Scientists. O'Reilly Media.

BTO, JNCC, & RSPB. (2014). *Breeding Bird Survey Instructions*. Norfolk. Retrieved from http://www.bto.org/volunteer-surveys/bbs/taking-part/download-forms-instructions

Buckland, S. T., Anderson, D. R., Burnham, K. P., Laake, J. L., Borchers, D. L., & Thomas, L. (2001). *Introduction to distance sampling*. Oxford: Oxford University Press.

Buckland, S. T., Magurran, a E., Green, R. E., & Fewster, R. M. (2005). Monitoring change in biodiversity through composite indices. *Philosophical Transactions of the Royal Society of London. Series B, Biological Sciences*, *360*(1454), 243–254.

http://doi.org/10.1098/rstb.2004.1589

Buckland, S. T., Summers, R. W., Borchers, D. L., & Thomas, L. (2006). Point transect sampling with traps or lures. *Journal of Applied Ecology*, *43*(2), 377–384. http://doi.org/10.1111/j.1365-2664.2006.01135.x

Buckland, S. T., Marsden, S. J., Green, R. E. (2008). Estimating bird abundance: making methods work. *Bird Conservation International, 18* (S1), 91-108. http://www.journals.cambridge.org/abstract_S0959270908000294

Burney, J. A., Davis, S. J., & Lobell, D. B. (2010). Greenhouse gas mitigation by agricultural intensification. *PNAS*, *107*(26), 1–6. http://doi.org/10.1073/pnas.0914216107/-/DCSupplemental.www.pnas.org/cgi/doi/10.1073/pnas.0914216107

Burnham, K. P., & Anderson, D. R. (2002). *Model selection and multimodel inference: a practical information-theoretic approach* (2nd editio). New York: Springer.

Butler, S. J., Vickery, J. A., & Norris, K. (2007). Farmland Biodiversity and the Footprint of Agriculture. *Science*, *315*(January), 381–384.

Butterfly Conservation. (2016). Big Butterfly Count. Retrieved November 16, 2017, from http://www.bigbutterflycount.org/2016mainresults

Catterall, C. P., Freeman, A. N. D., Kanowski, J., & Freebody, K. (2012). Can active restoration of tropical rainforest rescue biodiversity? A case with bird community indicators. *Biological Conservation*, *146*(1), 53–61. http://doi.org/10.1016/j.biocon.2011.10.033

CBD. (2018). Aichi Biodiversity Targets. Retrieved January 18, 2018, from https://www.cbd.int/sp/targets/

Crawley, M. J. (2002). *Statistical Computing: An Introduction to Data Analysis Using S-PLUS*. Chichester: John Wiley & Sons Ltd.

Crowder, D. W., & Reganold, J. P. (2015). Financial competitiveness of organic agriculture on a global scale. *Proceedings of the National Academy of Sciences*, *112*(24), 7611–7616. http://doi.org/10.1073/pnas.1423674112

Cumming, G. S., Buerkert, A., Hoffmann, E. M., Schlecht, E., von Cramon-Taubadel, S., & Tscharntke, T. (2014). Implications of agricultural transitions and urbanization for ecosystem services. *Nature*, *515*(7525), 50–57. http://doi.org/10.1038/nature13945

DEFRA. (2012). organic *Statistics 2011, United Kingdom*.

DEFRA. (2014). *Agriculture in the United Kingdom*. Retrieved from https://www.gov.uk/government/uploads/system/uploads/attachment_data/file/3

15103/auk-2013-29may14.pdf

DEFRA. (2017). organic *farming statistics 2016*.

Dirzo, R., & Raven, P. H. (2003). Global State of Biodiversity loss. *Annu. Rev. Environ. Resour.*, *28*, 137–167. http://doi.org/10.1146/annurev.energy.28.050302.105532

Donald, P. F. (2004). Biodiversity Impacts of Some Agricultural. *Conservation Biology*, *18*(1).

Donald, P. F., Green, R. E., & Heath, M. F. (2001). Agricultural intensification and the collapse of Europe's farmland bird populations. *Proceedings. Biological Sciences / The Royal Society*, *268*(1462), 25–29. http://doi.org/10.1098/rspb.2000.1325

EC. (2017). *Report from the Commission to the European Parliament and the Council on Member State National Action Plans and on progress in the implementation of Directive 2009/128/EC on sustainable use of pesticides*. Brussels.

Ekroos, J., Olsson, O., Rundlöf, M., Wätzold, F., & Smith, H. G. (2014). Optimizing agri-environment schemes for biodiversity, ecosystem services or both? *Biological Conservation*, *172*, 65–71. http://doi.org/10.1016/j.biocon.2014.02.013

FAO. (2006). *World agriculture: towards 2030 / 2050 - Interim Report*. Rome, Italy.

Firbank, L., Bradbury, R. B., McCracken, D. I., & Stoate, C. (2013). Delivering multiple ecosystem services from Enclosed Farmland in the UK. *Agriculture, Ecosystems & Environment*, *166*, 65–75. http://doi.org/10.1016/j.agee.2011.11.014

Fountain, M. T., & Harris, A. L. (2015). Non-target consequences of insecticides used in apple and pear orchards on Forficula auricularia L . (Dermaptera: Forficulidae). *Biological Control*, *91*(2015), 27–33. http://doi.org/10.1016/j.biocontrol.2015.07.007

Fuller, R. J. (2000). *Relationships between recent changes in lowland British agriculture and farmland bird populations: an overview*. Thetford, Norfolk.

Gabriel, D., Sait, S. M., Kunin, W. E., & Benton, T. G. (2013). Food production vs. biodiversity: Comparing organic and conventional agriculture. *Journal of Applied Ecology*, *50*(2), 355–364. http://doi.org/10.1111/1365-2664.12035

Garfinkel, M., & Johnson, M. (2015). Pest-removal services provided by birds on small organic farms in northern California. *Agriculture, Ecosystems & Environment*, *211*(2015), 24–31. http://doi.org/10.1016/j.agee.2015.04.023

Genghini, M., Gellini, S., & Gustin, M. (2006). organic and integrated agriculture: The effects on bird communities in orchard farms in northern Italy. *Biodiversity and*

Conservation, *15*(9), 3077–3094. http://doi.org/10.1007/s10531-005-5400-2

Giraudo, A. R., Matteucci, S. D., Alonso, J., Herrera, J., & Abramson, R. R. (2008). Comparing bird assemblages in large and small fragments of the Atlantic Forest hotspots. *Biodiversity and Conservation*, *17*(5), 1251–1265. http://doi.org/10.1007/s10531-007-9309-9

Godfray, H. C. J., Beddington, J. R., Crute, I. R., Haddad, L., Lawrence, D., Muir, J. F., … Toulmin, C. (2010). Food Security: The challenge of Feeding 9 Billion people. *Science*, *327*(812). http://doi.org/10.1126/science.1185383

Green, R. E., Cornell, S. J., Scharlemann, J. P. W., & Balmford, A. (2005). Farming and the Fate of Wild Nature. *Science*, *550*(307). http://doi.org/10.1126/science.1106049

Guitierrez-Velez, V. H., DeFries, R., Pinedo-Vasquez, M., Uriarte, M., Padoch, C., Baethgen, W., … Lim, Y. (2011). High-yield oil palm expansion spares land at the expense of forests in the Peruvian Amazon. *Environmental Resource Letters*, *6*. http://doi.org/10.1088/1748-9326/6/4/044029

Hamel, P. B., Smith, W. P., Twedt, D. J., Woehr, J. R., Morris, E., Hamilton, R. B., & Cooper, R. J. (1996). *A Land Manager's Guide to Point Counts of Birds in the Southeast*. Asheville.

Hardman, C. J., Harrison, D. P. G., Shaw, P. J., Nevard, T. D., Potts, S. G., & Norris, K. (2016). Supporting local diversity of habitats and species on farmland : a comparison of three wildlife-friendly schemes. *Journal of Applied Ecology*, *53*, 171–180. http://doi.org/10.1111/1365-2664.12557

Harrison, X. A., Donaldson, L., Correa-cano, M. E., Evans, J., Goodwin, C., Robinson, B., … Inger, R. (2017). Best practice in mixed effects modelling and multi-model inference in ecology. *PeerJ*.

Henckel, L., Börger, L., Meiss, H., Gaba, S., & Bretagnolle, V. (2015). organic fields sustain weed metacommunity dynamics in farmland landscapes. *Proc R Soc B*, *282*, 1–9.

Henderson, I. G., Holland, J. M., Storkey, J., Lutman, P., Orson, J., & Simper, J. (2012). Effects of the proportion and spatial arrangement of un-cropped land on breeding bird abundance in arable rotations. *Journal of Applied Ecology*, *49*(4), 883–891. http://doi.org/10.1111/j.1365-2664.2012.02166.x

Herrero, M., Havlík, P., Valin, H., Notenbaert, A., Rufino, M. C., Thornton, P. K., … Obersteiner, M. (2013). Biomass use, production, feed efficiencies, and greenhouse gas emissions from global livestock systems. *PNAS*, *110*(52), 1–6. http://doi.org/10.1073/pnas.1308149110

Hillocks, R. J. (2012). Farming with fewer pesticides: EU pesticide review and resulting challenges for UK agriculture. *Crop Protection*, *31*. http://doi.org/10.1016/j.cropro.2011.08.008

Hole, D. G., Perkins, a. J., Wilson, J. D., Alexander, I. H., Grice, P. V., & Evans, a. D. (2005). Does organic farming benefit biodiversity? *Biological Conservation*, *122*(1), 113–130. http://doi.org/10.1016/j.biocon.2004.07.018

HSE. (2016). Changes to authorisations for products containing chlorpyrifos. Retrieved November 16, 2017, from http://www.hse.gov.uk/pesticides/news/information-update-0316.htm

James, G., Witten, D., Hastle, T., Tibshirani, R. (2014). *An Introduction to Statistical Learning: With Application in R.* Springer Publishing Company, Incorporated.

JNCC. (2014). *Uk Biodiversity Indicators 2014. Measuring progrees towards halting biodiversity loss*.

Kamil, B. (2017). MuMIn: Multi-Model Inference. R Package version 1.40.0

Katayama, N. (2016). Bird diversity and abundance in organic and conventional apple orchards in northern Japan. *Nature*, *6*(September), 1–7. http://doi.org/10.1038/srep34210

Kissinger, G. (2012). *Corporate social responsibility and supply agreements in the agricultural sector decreasing land and climate pressures* (No. 14). Copenhagen, Denmark.

Kleijn, D., Baquero, R. a., Clough, Y., Díaz, M., De Esteban, J., Fernández, F., … Yela, J. L. (2006). Mixed biodiversity benefits of agri-environment schemes in five European countries. *Ecology Letters*, *9*(March), 243–254. http://doi.org/10.1111/j.1461-0248.2005.00869.x

Kleijn, D., Berendse, F., Smit, R., & Gilissen, N. (2001). Agri-environment schemes do not effectively protect biodiversity in Dutch agricultural landscapes. *Nature*, *635*, 723–725.

Kleijn, D., & Sutherland, W. J. (2003). How effective are European agri-environment schemes in conserving and promoting biodiversity? *Journal of Applied Ecology*, *40*(6), 947–969. http://doi.org/10.1111/j.1365-2664.2003.00868.x

Kogan, M. (1998). Integrated Pest Management: Historical Perspectives and Contemporary Developments. *Annual Review of Entomology*, *43*(1), 243–270. http://doi.org/10.1146/annurev.ento.43.1.243

Krebs, J. R., Wilson, J. D., Bradbury, R. B., & Siriwardena, G. M. (1999). The second Silent Spring? *Nature*, *400*, 611–612. http://doi.org/10.1038/23127

Kremen, C., Williams, N. M., & Thorp, R. W. (2002). Crop pollination from native bees at risk from agricultural intensification. *Proceedings of the National Academy of Sciences of the United States of America*, *99*(26), 16812–16816. http://doi.org/10.1073/pnas.262413599

LEAF. (2014). *LEAF Marque Global Standards*. Warwickshire.

LEAF. (2017). *LEAF Marque Standard v.14.1*. Warwickshire.

Leventon, J., Schaal, T., Velten, S., Dänhardt, J., Fischer, J., Abson, D. J., & Newig, J. (2017). Land Use Policy Collaboration or fragmentation? Biodiversity management through the common agricultural policy. *Land Use Policy*, *64*, 1–12. http://doi.org/10.1016/j.landusepol.2017.02.009

Lewington, R., & Bebbington, J. (1998). Guide to the butterflies of Britain. Field Studies Council.

Lynch, J. F. (1995). *Effects of Point Count Duration, Time-of-Day , and Aural Stimuli on Detectability of Migratory and Resident Bird Species in Quintana Roo, Mexico*.

Mäder, P., Fliessbach, A., Dubois, D., Gunst, L., Fried, P., & Niggli, U. (2002). Soil fertility and biodiversity in organic farming. *Science*, *296*(5573), 1694–7. http://doi.org/10.1126/science.1071148

Mangan, A. M., Pejchar, L., & Werner, S. J. (2017). Bird use of organic apple orchards: Frugivory, pest control and implications for production. *PloS One*, *12*(9), 1–15. http://doi.org/10.1371/journal.pone.0183405

Marini, M. A. (2001). Effects of forest fragmentation on birds of the cerrado region, Brazil. *Bird Conservation International*, *11*, 13–25.

Mazerolle, M. J. (2017). AICcmodavg: Model selection and multimodel inference based on (Q)AIC(c). Retrieved from https://cran.r-project.org/package=AICcmodavg.

Mclaughlin, D. W. (2011). Land, Food, and Biodiversity. *Conservation Biology*, *25*(6), 1117–1120. http://doi.org/10.1111/j.1523-1739.2011.01768.x

Munang, R., Thiaw, I., Rivington, M., & Goldman, R. (2010). *UNEP Policy Series. Ecosystem Management: The Role of Ecosystems in Developing a Sustainable 'Green Economy'* (Policy Brief 2).

Natural England. (2010). *Traditional orchards: orchards and wildlife*. *Technical Information Note* (Vol. TIN020).

Nellemann, C., & et al. (2009). *The Environmental Food Crisis*. Nairobi, Kenya.

Norton, L., Johnson, P., Joys, A., Stuart, R., Chamberlain, D., Feber, R., … Fuller, R. J. (2009). Consequences of organic and non-organic farming practices for field, farm and landscape complexity. *Agriculture, Ecosystems & Environment*, *129*(1–3). http://doi.org/10.1016/j.agee.2008.09.002

O'Hara, R. B., & Kotze, D. J. (2010). Do not log-transform count data. *Methods in Ecology and Evolution*, *1*(2), 118–122. http://doi.org/10.1111/j.2041-210X.2010.00021.x

Oerke, E.-C. (2006). Crop losses to pests. *The Journal of Agricultural Science*, *144*, 31–43. http://doi.org/10.1017/S0021859605005708

Oksanen, J., Blanchet, F. G., Friendly, M., Kindt, R., Legendre, P., McGlinn, D., … Wagner, H. (2017). *vegan: Community Ecology Pack. R package version 2.4-5*. Retrieved from https://cran.r-project.org/package=vegan

Pärt, T., & Söderström, B. O. (1999). Conservation Value of Semi-Natural Pastures in Sweden: Contrasting Botanical and Avian Measures. *Conservation Biology*, *13*(4), 755–765.

Peisley, R. K., Saunders, M. E., & Luck, G. W. (2016). Cost-benefit trade-offs of bird activity in apple orchards. *PeerJ*, 1–20. http://doi.org/10.7717/peerj.2179

Pereira, H. M., Leadley, P. W., Proença, V., Alkemade, R., Scharlemann, J. P. W., Araújo, M. B., … Walpole, M. (2010). Scenarios for Global Biodiversity in the 21st Century. *Science*, *330*(6010), 1496–1501.

Pergola, M., D'Amico, M., Celano, G., Palese, a M., Scuderi, a, Di Vita, G., … Inglese, P. (2013). Sustainability evaluation of Sicily's lemon and orange production: an energy, economic and environmental analysis. *Journal of Environmental Management*, *128*, 674–82. http://doi.org/10.1016/j.jenvman.2013.06.007

Petit, L. J., & Petit, D. R. (2003). Evaluating the Importance of Human-Modified Lands for Neotropical Bird Conservation. *Conservation Biology*, *17*(3), 687–694.

Phalan, B., Onial, M., Balmford, A., & Green, R. E. (2011). Reconciling food production and biodiversity conservation: land sharing and land sparing compared. *Science*, *333*, 1289–1291. http://doi.org/10.1126/science.1208742

Pimentel, D., Acquay, H., Biltonen, M., Rice, P., Silva, M., Nelson, J., … D'Amore, M. (1992). Environmental and Economic Costs of Pesticide Use. *BioScience*, *42*(10), 750–760.

Pollard, E. (1977). A method for assessing changes in the abundance of butterflies. *Biological Conservation*, *12*.

Puech, C., Baudry, J., Joannon, A., Poggi, S., & Aviron, S. (2014). organic vs. conventional farming dichotomy: Does it make sense for natural enemies? *Agriculture, Ecosystems & Environment*, *194*, 48–57. http://doi.org/10.1016/j.agee.2014.05.002

QGIS Development Team. (2015). QGIS Geographic Information System. Open Source Geospatial Foundation Project. Retrieved from http://qgis.osgeo.org

Quinn, J. E., Brandle, J. R., & Johnson, R. J. (2012). The effects of land sparing and wildlife-friendly practices on grassland bird abundance within organic farmlands. *Agriculture, Ecosystems & Environment*, *161*, 10–16. http://doi.org/10.1016/j.agee.2012.07.021

R Core Team. (2015). R: A Language and Environment for Statistical Computing. Vienna, Austria: R Foundation for Statistical Computing. Retrieved from https://www.r-project.org/

Ralph, C. J., Droege, S., & Sauer, J. R. (1995). *Managing and Monitoring Birds Using Point Counts*.

Reed, M., Lewis, N., and Dwyer, J. (2017) "The effect and impact of LEAF Marque in the delivery of more sustainable farming: a study to understand the added value to farmers." The CCRI, Gloucester, England.

Reganold, J. P., & Wachter, J. M. (2016). organic agriculture in the twenty-first century. *Nature Plants*, *2*(February), 15221. http://doi.org/10.1038/nplants.2015.221

Richards, S. A., Whittingham, M. J., & Stephens, P. A. (2011). Model selection and model averaging in behavioural ecology: the utility of the IT-AIC framework. *Behavioural Ecology and Sociobiology*, *65*, 77–89. http://doi.org/10.1007/s00265-010-1035-8

Robertson, H., Marshall, D., Slingsby, E., & Newman, G. (2012). *Economic, biodiversity, resource protection and social values of orchards: a study of six orchards by the Herefordshire Orchards Community Evaluation Project*.

Robinson, R. A., & Sutherland, W. J. (2002). Post-war changes in arable farming and biodiversity in Great Britain. *Journal of Applied Ecology*, *39*, 157–176.

Rundlöf, M., & Smith, H. G. (2006). The effect of organic farming on butterfly diversity depends on landscape context. *Journal of Applied Ecology*, *43*(6), 1121–1127. http://doi.org/10.1111/j.1365-2664.2006.01233.x

Samnegård, U., Alins, G., Boreux, V., Bosch, J., García, D., Happe, A. K., … Hambäck, P. A. (2019). Management trade-offs on ecosystem services in apple orchards across Europe: Direct and indirect effects of organic production. *Journal of Applied Ecology, 56*(4), 802–811. https://doi.org/10.1111/1365-2664.13292

Schneider, M. K., Lüscher, G., Jeanneret, P., Arndorfer, M., Ammari, Y., Bailey, D., … Herzog, F. (2014). Gains to species diversity in organically farmed fields are not propagated at the farm level. *Nature Communications*, *5*(May), 4151. http://doi.org/10.1038/ncomms5151

Sotherton, N. W., Self, M. J., Game, T., & Trust, C. (2000). *Changes in plant and arthropod biodiversity on lowland farmland : an overview*. *The Ecology and Conservation of Lowland Farmland Birds*.

Steffan-dewenter, I., & Tscharntke, T. (2000). Butterfly community structure in fragmented habitats. *Ecology Letters*, *3*, 449–456.

Stenberg, J. A. (2017). A conceptual Framework for Integrated Pest Managment. *Trends in Plant Science*, *22*(9), 759–769.

Stolze, M., Piorr, A., Häring, A., & Dabbert, S. (2000). *The Environmental Impacts of organic Farming in Europe*. (Anna Häring, Ed.) (Organic Fa). Stuttgart: Prof Dr. Stephan Dabbert.

Sutherland, W. J., Dicks, L. V., Ockendon, N., Petrovan, S. O., & Smith, R. K. (2018). *What Works in Conservation*. Cambridge, UK: Open Book Publishers.

Sutherland, W. J., Newton, I., & Green, R. E. (2004). *Bird Ecology and Conservation. A Handbook of Techniques*. Oxford: Oxford University Press.

Swinbank, A. (2016). Brexit or Bremain? Future Options for UK Agriculture Policy and the CAP. *EuroChoices*, *15*(2), 5–10. http://doi.org/10.1111/1746-692X.12126

Tebbs, E., & Rowland, C. (2014). *A high spatial resolution woody cover map for Great Britain: preliminary results.* Lancaster.

TEEB. (2008). *The economics of ecosystems & biodiveristy: An interim report*. Cambridge, UK.

Thomas, C. D., Cameron, A., Green, R. E., Bakkenes, M., Beaumont, L. J., Collingham, Y. C., … Williams, S. E. (2004). Extinction risk from climate change. *Nature*, *427*.

Thomas, L., Buckland, S. T., Rexstad, E. a., Laake, J. L., Strindberg, S., Hedley, S. L., … Burnham, K. P. (2010). Distance software: Design and analysis of distance sampling surveys for estimating population size. *Journal of Applied Ecology*, *47*(1), 5–14. http://doi.org/10.1111/j.1365-2664.2009.01737.x

Tilman, D., Balzer, C., Hill, J., & Befort, B. L. (2011). Global food demand and the sustainable intensification of agriculture. *PNAS*, *108*(50). http://doi.org/10.1073/pnas.1116437108

Tilman, D., Cassman, K. G., Matson, P. A., Naylor, R., & Polasky, S. (2002). Agricultural sustainability and intensive production practices. *Nature*, *418*(August).

Tilman, D., Fargione, J., Wolff, B., D'Antonio, C., Dobson, a, Howarth, R., … Swackhamer, D. (2001). Forecasting agriculturally driven global environmental change. *Science (New York, N.Y.)*, *292*(5515), 281–4. http://doi.org/10.1126/science.1057544

Todd, J. H., Malone, L. A., McArdle, B. H., Benge, J., Poulton, J., Thorpe, S., & Beggs, J. R. (2011). Invertebrate community richness in New Zealand kiwifruit orchards under organic or integrated pest management. *Agriculture, Ecosystems and Environment*, *141*(1–2), 32–38. http://doi.org/10.1016/j.agee.2011.02.007

Tscharntke, T., Clough, Y., Wanger, T. C., Jackson, L., Motzke, I., Perfecto, I., … Whitbread, A. (2012). Global food security, biodiversity conservation and the future of agricultural intensification. *Biological Conservation*, *151*(1), 53–59. http://doi.org/10.1016/j.biocon.2012.01.068

Tscharntke, T., Klein, A. M., Kruess, A., Steffan-Dewenter, I., & Thies, C. (2005). Landscape perspectives on agricultural intensification and biodiversity - ecosystem service management. *Ecology Letters*, *8*(8), 857–874. http://doi.org/10.1111/j.1461-0248.2005.00782.x

Tuck, S. L., Winqvist, C., Mota, F., Ahnström, J., Turnbull, L. a., & Bengtsson, J. (2014). Land-use intensity and the effects of organic farming on biodiversity: a hierarchical meta-analysis. *Journal of Applied Ecology*, *51*(3), 746–755. http://doi.org/10.1111/1365-2664.12219

UN. (2010). *World Population Prospects: The 2010 Revision. Vol. I: Comprehensive Tables*. New York.

Van Dyck, H., Van Strien, A. J., Maes, D., & Van Swaay, C. A. M. (2009). Declines in Common, Widespread Butterflies in a Landscape under Intense Human Use. *Conservation Biology*, *23*(4), 957–965. http://doi.org/10.1111/j.1523-1739.2009.01175.x

van Swaay, C Brereton, T Kirkland, P., & Warren, M. (2012). *Manual for Monitoring Butterflies.* Wageningen.

Van Swaay, C., Regan, E., Ling, M., Bozhinovska, E., Fernandez, M., Marini-Filho, O. J., … Underhill, L. (2015). *Guidelines for Standardised Global Butterfly Monitoring* (Vol. GEO BON Te). Leipzig, Germany.

Venables, W. N., & Ripley, B. D. (2002). *Modern Applied Statistics with S.* (Fourth). New York: Springer. Retrieved from http://www.stats.ox.ac.uk/pub/MASS4

Vitousek, P. M., Aber, J. D., Howarth, R. W., Likens, G. E., Matson, P. A., Schindler, D. W., … 2Department. (2010). Human alteration of the global nutrogen cycle: sources and consequences. *Ecological Applications*, *7*(3), 737–750.

Weibull, A.-C., Bengtsson, J., & Nohlgren, E. (2000). Diversity of butterflies in the agricultural landscape: the role of farming system and landscape heterogeneity. *Ecography*, *23*, 743–750. http://doi.org/DOI 10.1034/j.1600-0587.2000.230611.x

Weibull, A., Ostman, O., & Granqvist, A. (2003). Species richness in agroecosystems: the effect of landscape, habitat and farm management. *Biodiversity and Conservation*, *12*.

WWF. (2014). *Living Planet Report 2014: Species and spaces, people and places*. Gland, Switzerland.

Zhang, Y., Sha, Z., Guan, F., Wang, C., & Li, Y. (2016). Impacts of geese on weed communities in corn production systems and associated economic benefits. *Biological Control*, *99*, 47–52. http://doi.org/10.1016/j.biocontrol.2016.04.011

Appendix B
Butterfly Species List

A list of common butterfly species and their Latin names that were surveyed on apple orchards in 2016, spring to autumn during conditions suitable for butterfly surveys.

Common name	Latin name
Brimstone	*Gonepteryx rhamni*
Comma	*Polygonia c-album*
Common Blue	*Polyommatus icarus*
Gatekeeper	*Pyronia tithonus*
Green-Veined White	*Pieris napi*
Holly Blue	*Celastrina argiolus*
Large skipper	*Ochlodes sylvanus*
Large White	*Pieris brassicae*
Marbled White	*Melanargia galathea*
Meadow Brown	*Maniola jurtina*
Orange tip	*Anthocharis cardamines*
Painted Lady	*Vanessa cardui*
Peacock	*Aglais io*
Red Admiral	*Vanessa atalanta*
Silver-washed fritillary	*Argynnis paphia*
Small Copper	*Lycaena phlaeas*
Small Skipper	*Thymelicus sylvestris*
Small Tortoiseshell	*Aglais urticae*
Small White / Cabbage White	*Pieris rapae*
Speckled Wood	*Pararge aegeria*

Appendix C

Bird Species List

Bird species that were identified during field surveys of 2015 and 2016. The total bird species data was used in Chapter 3. Birds identified with '*' are insectivorous birds used for analysis in Chapter 4 and Chapter 5.

Common name	Latin name
Blackbird*	*Turdus merula*
Blackcap*	*Sylvia atricapilla*
BlueTit*	*Cyanistes caeruleus*
Bullfinch*	*Pyrrhula pyrrhula*
Buzzard	*Buteo buteo*
CanadaGoose	*Branta canadensis*
CarrionCrow	*Corvus corone*
Chaffinch*	*Fringilla coelebs*
Chiffchaff*	*Phylloscopus collybita*
CoalTit*	*Periparus ater*
Coot	*Fulica*
CollaredDove	*Streptopelia decaocto*
Cuckoo*	*Cuculidae*
Dunnock*	*Prunella modularis*
GSWoodpecker*	*Dendrocopos major*
Goldcrest*	*Regulus regulus*
Goldfinch*	*Carduelis carduelis*
GreatTit*	*Parus major*
GreenWoodpecker	*Picus viridis*
Greenfinch	*Chloris chloris*
GreyHeron	*Ardea cinerea*
HerringGull	*Larus argentatus*
HouseMartin*	*Delichon urbicum*
HouseSparrow	*Passer domesticus*
Jackdaw*	*Corvus monedula*
LesserSWoodpecker*	*Dryobates minor*
LongTailedTit*	*Aegithalos caudatus*
Magpie	*Pica pica*
Mallard	*Anas platyrhynchos*
MistleThrush*	*Turdus viscivorus*
Moorhen	*Gallinula*
MuteSwan	*Cygnus olor*
Pheasant	*Phasianus colchicus*
PiedWagtail*	*Motacilla alba*
Robin*	*Erithacus rubecula*

Skylark*	*Alauda arvensis*
SongThrush	*Turdus philomelos*
Starling*	*Sturnus vulgaris*
StockDove	*Columba oenas*
Barn Swallow*	*Hirundo rustica*
Swift*	*Apodidae*
WhiteThroat*	*Sylvia communis*
WillowWarbler*	*Phylloscopus trochilus*
WoodPigeon	*Columba palumbus*
Wren*	*Troglodytidae*
Yellowhammer*	*Emberiza citrinella*
Linnet*	*Linaria cannabina*
Treecreeper*	*Certhiidae*
Siskin*	*Spinus spinus*
Red Legged Partridge	*Alectoris rufa*
Raven	*Corvus corax*
Rook	*Corvus frugilegus*
MeadowPip*	*Anthus pratensis*
FeralPidgeon	*Columba livia*
RedKite	*Milvus milvus*
SpottedFlyCatcher*	*Muscicapa striata*
Sparrowhawk	*Accipiter nisus*
RedStart*	*Phoenicurus phoenicurus*
LRedPoll	*Acanthis cabaret*
LesserBBGull	*Larus fuscus*
Jay*	*Garrulus glandarius*
PiedFlyCatcher*	*Ficedula hypoleuca*
Kestrel	*Falco tinnunculus*
LWhiteThroat*	*Sylvia curruca*
GreyWagtail*	*Motacilla cinerea*
Nuthatch*	*Sitta*

Appendix D

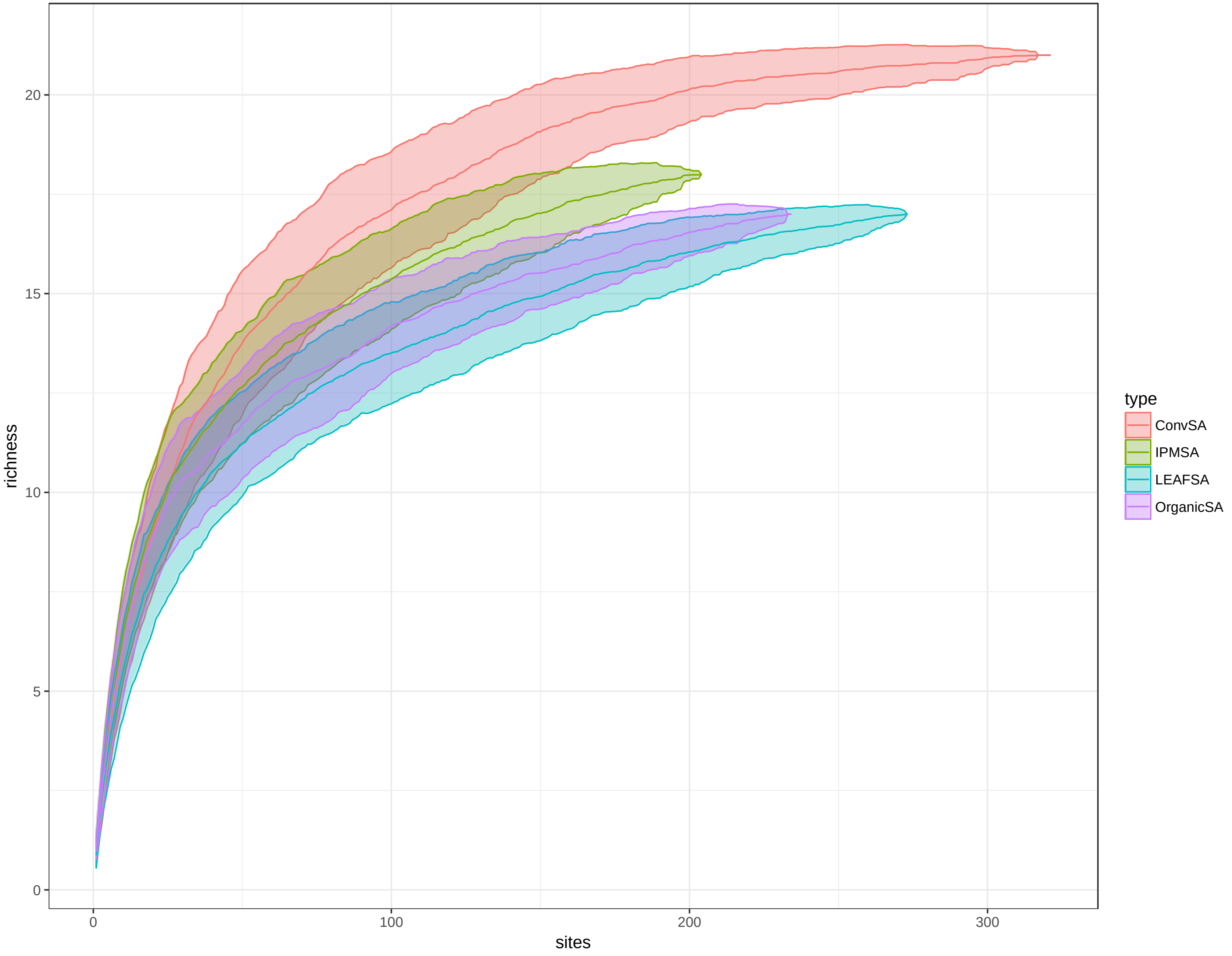
Bird GLMM 'Full' Model Average summary

Generalised linear mixed effects full model average summary examining bird community metrics as the response variable (bird abundance, species richness and Shannon diversity) across two years 2015 and 2016 in apple orchards of the study area. Investigation of the response variables in relation to farm management, observer, apple type, Woody Cover, day number and bird box presence. VIF is the variance inflation factor value derived from squared GVIF^(1/(2*Df)), where any variable above 5 was considered correlated and removed from the global model before dredging again. The intercept is conventional farm management with CP as the observer. Values in bold are significant (P=<0.05).

		Bird Abundance				**Bird Species Richness**				**Bird Diversity**			
Fixed Effects		Value	Standard Error	*P*-value	VIF	Value	Standard Error	*P*-value	VIF	Value	Standard Error	*P*-value	VIF
Farm Management	Conventional (Intercept)	2.554	0.114	< **0.001**	-	1.940	0.069	< **0.001**	-	1.666	0.078	< **0.001**	**-**
	IPM	0.051	0.092	0.575	2.314	0.046	0.060	0.449	2.604	0.105	0.081	0.201	2.214
	LEAF	0.060	0.131	0.647	2.314	0.098	0.066	0.140	2.604	0.164	0.088	0.063	2.214
	Organic	0.562	0.062	< **0.001**	2.314	0.394	0.061	< **0.001**	2.604	0.373	0.068	< **0.001**	2.214
Observer	PL	-0.379	0.097	< **0.001**	2.092	-0.285	0.065	< **0.001**	2.194	0.251	0.079	**0.002**	2.12
	SF	-0.290	0.075	< **0.001**	2.092	-0.114	0.050	**0.023**	2.194	0.021	0.06	0.72	2.12
Apple types	Dessert	-0.04	0.084	0.639	2.652	-0.004	0.03	0.884	2.984	0.003	0.025	0.887	2.696
	Juice	-0.064	0.099	0.515	2.59	-0.072	0.057	0.213	2.64	0.006	0.027	0.822	2.152
	Cider	-0.005	0.076	0.945	2.696	-0.034	0.061	0.578	2.588	-0.01	0.04	0.818	2.256
Environmental/ Landscape	Day number	0.082	0.068	0.227	2.064	-	-	-	-	-	-	-	-

Woody Cover	-0.145	0.144	0.311	2.818	-0.039	0.063	0.534	4.218	-	-	-	-
Bird box presence	-	-	-	-	-0.021	0.046	0.648	3.468	-	-	-	-
Temperature	-	-	-	-	-	-	-	-	0.066	0.065	0.311	2.18

Appendix E Butterfly species accumulation



Appendix F **Butterfly GLMM 'Full' model average summary**

Generalised linear mixed effects full model average summary examining butterfly community metrics as the response variable (bird abundance, species richness and Shannon diversity) across two years 2015 and 2016 in apple orchards of the study area. Investigation of the response variables in relation to farm management, observer, apple type, orchard size, day number, sunshine. Interaction between management type and orchards size proved significant in initial model analysis and included in global model. The intercept is conventional farm management with KI as the observer, and interaction conventional:orchard_size. Values in bold are significant (P=<0.05). VIF is the variance inflation factor value derived from squared GVIF^(1/(2*Df)), where any variable above 5 was considered correlated and removed from the global model before dredging.

This intercept is KI observer, as farm management was not significant in the top model selection for diversity.

		Butterfly Abundance				**Butterfly Species Richness**				**Butterfly Diversity**			
Fixed Effects		Value	Standard Error	*P*-value	VIF	Value	Standard Error	*P*-value	VIF	Value	Standard Error	*P*-value	VIF
Farm Management	Conventional (Intercept)	1.148	0.256	<**0.001**	-	0.672	0.073	<**0.001**	-	0.590*	0.047	<**0.001**	**-**
	IPM	-1.017	0.351	**0.004**	3.152	-0.004	0.061	0.941	2.438	-	-	-	2.604
	LEAF	-0.199	0.356	0.577	3.152	-0.003	0.056	0.959	2.438	-	-	-	2.604
	Organic	0.189	0.232	0.415	3.152	0.021	0.074	0.781	2.438	-	-	**-**	2.604
Observer	CS	-0.067	0.194	0.73	2.638	0.017	0.077	0.826	2.404	-0.003	0.025	**0.903**	2.43
Interactions	IPM: orch_size	-1.470	1.6	0.358	3.582	-	-	-	-	-	-	-	-

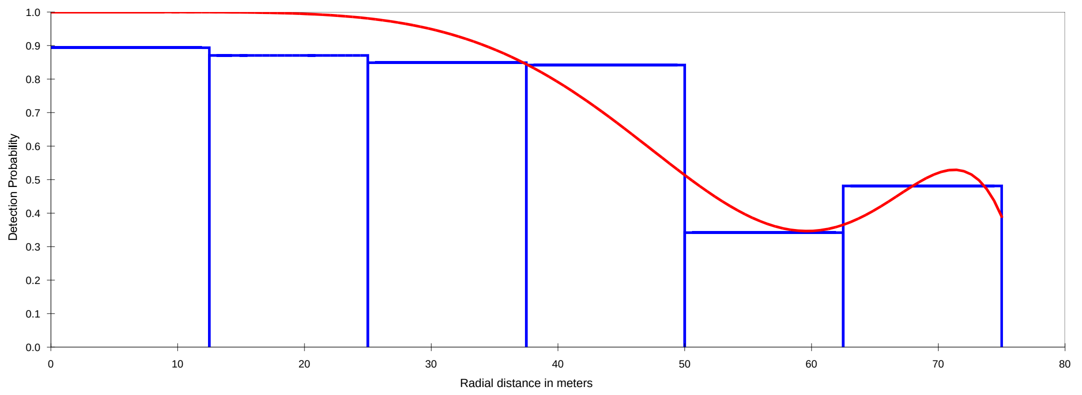
	LEAF: orch_size	-0.173	0.581	0.767	3.582	-	-	-	-	-	-	-	-
	Organic: orch_size	-0.433	0.5	0.385	3.582	-	-	-	-	-	-	-	-
	IPM: Woody_Cover	-1.504	1.059	0.156	3.046	-	-	-	-	-	-	-	-
	LEAF: Woody_Cover	-0.333	0.710	0.640	3.046	-	-	-	-	-	-	-	-
	Organic: Woody_Cover	-0.652	0.666	0.329	3.046	-	-	-	-	-	-	-	-
Environmental/ Landscape	Day number	0.960	0.23	**<0.001**	2.022	0.419	0.149	**0.005**	2.146	0.097	0.117	0.412	2.026
	Orchard size	0.322	0.529	0.543	2.508	-	-	-	-	-	-	-	-
	Woody Cover	-	-	-	-	-	-	-	-	-0.016	0.055	0.776	2.52
	Sunshine	0.149	0.093	0.111	2.032	0.200	0.112	0.076	2.076	0.136	0.089	0.125	2.028
	Total insecticide amount	-	-	-	-	0.126	0.130	0.331	3.446	-0.014	0.033	0.673	4.364
	Temperature	-	-	-	-	0.319	0.139	0.022	2.224	-	-	-	-

Appendix G

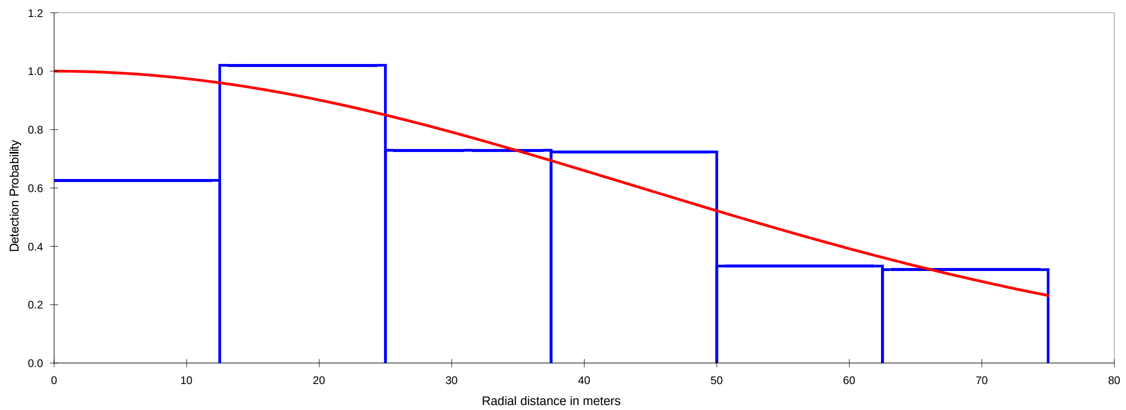
All-bird density detection functions

All bird detection probability plots using the model with lowest AIC (20347.73, no of parameters 1). The model used half normal and simple polynomial distribution and constructed in Distance *version 7.0*.

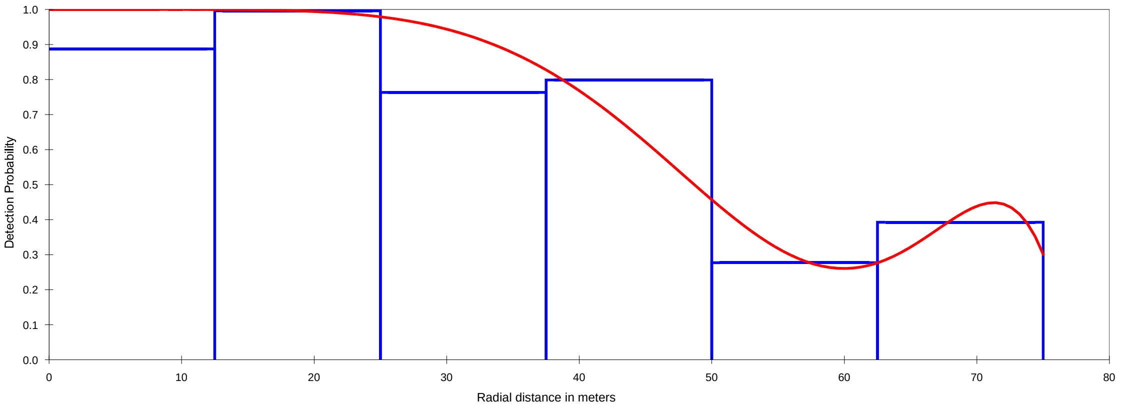
Conventional detection probability plot



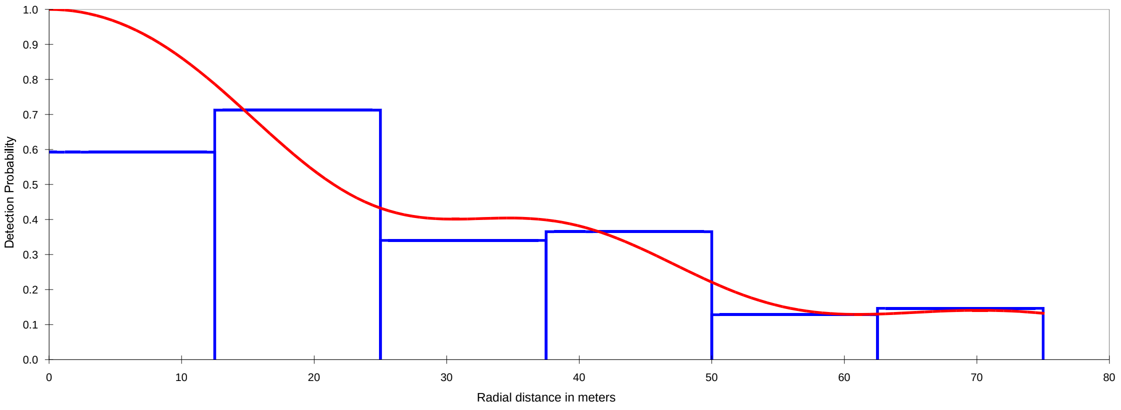
IPM detection probability plot



LEAF detection probability plot



Organic detection probability plot



CHAPTER 4

Pest control services from birds is higher on organic rather than non-organic apple orchards

4.1 ABSTRACT

Globally, biodiversity in agro-ecosystems is found to be higher on organic farms but the functional consequence of this is still poorly understood. Pest regulation ecosystem services have been well studied in the tropics, especially by arthropods. However, the impact of avian pest regulation on reducing pests in temperate, especially perennial, agro-ecosystems is still elementary. Some studies suggest increases in pest regulators positively impact the predation of pest insects, but an extensive comparison between organic and non-organic farm management regarding the provision of a viable pest control ecosystem service by birds has not been established. The understanding of this is important for future development of nature-friendly farming, to protect biodiversity and increase ecosystem service provision.

Here, I measure the abundance, diversity, density, and species richness of the functional group of insectivore birds, to understand how these potential pest regulators change over different farm managements including organic and non-organic orchards that have different levels of biodiversity. I use sentinel artificial prey to infer a daily predation rate by these pest regulators and then measure pest moth levels on farm for pest level information across each farm type.

As insectivorous birds increase on organic orchards in abundance, species richness and diversity, the predation rate of dough caterpillars mirror this pattern and significantly increase. Orchard pest moth levels here are shown to be controlled by two different agro-ecosystems; organic orchards use a natural pest control from birds to reduce moth pests, whereas chemical control of pests is used on non-organic orchards.

Results here show that organic orchards support a higher abundance, species richness, density, and diversity of the insectivore bird functional group, which can significantly

increase predation on artificial caterpillar pests, providing a functioning natural pest control ecosystem service that is currently available only on organic orchards.

4.2 INTRODUCTION

4.2.1 Biodiversity and pest regulating ecosystem services

The Millennium Ecosystem Assessment (MA) identifies that biodiversity itself supports ecosystem services that society receives, including: the origins and yields of pharmaceuticals from ecosystems rich in species diversity (e.g. the Tropics) (MA, 2005a); regulating ecosystem services such as pollination, seed dispersal and carbon storage (MA, 2005b); wild crop diversity for continued improvements in crop modifications (Jenkins et al., 2004) and resilience against agricultural pests (Mace et al., 2012; Cardinale et al., 2003). Although we know that biodiversity supports ecosystem services, there is still relatively little known on the impact that increased biodiversity has on the provisioning and extent of ecosystem services, especially to biological control of pests (Porcel et al., 2018; Jonsson, Kaartinen, & Straub, 2017).

Natural pest control, or biological control, is the control of crop pests and is an important regulating ecosystem service in agricultural landscapes. This global phenomenon gives agricultural benefits such as from arthropod abundance and ability to control pest species (Samnegård et al., 2019; Lemessa et al., 2015; Solomon et al., 2000); the importance of bats as an agricultural pest predator in tropical lowland forests (Kalka et al., 2008); ants and birds providing pest control to Kenyan coffee farms (Milligan et al., 2016); and finally kestrels controlling fruit eating birds in US orchards (Shave et al., 2018). It has also been suggested that ecosystem services not only increases yield, such as Mols & Visser (2002) that suggested great tits (*Parus major*) reduced caterpillar damage and increased apple yield per tree, and has also been found to increase crop quality in coffee (Classen et al., 2014) and apples (Garratt et al., 2014).

4.2.2 Pest control services from birds

The subject of ecosystem services provided by birds had a plethora of studies conducted on it in the 1880s to 1920s (Kronenberg, 2014). Although the concept of ecosystem services was not reached at the time, "economic ornithology" arose and brought change to policy in terms of conservation status of birds and furthered research in the area. Kronenburg (2014) gives a detailed review of the era of economic ornithology and highlights the links between its areas of interest in using birds as a commodity with the current ecosystem service concept of provisioning, regulating, and supporting services.

Economic ornithology focussed on services and dis-services of birds to farmers, which is both practical and end-user based research. However, as soon as birds could be replaced by a more efficient service mechanism - organochlorine chemicals during the industrial revolution (Kirk et al., 1996; Krebs et al., 1999) - the study of economic ornithology was no longer needed and rapidly declined in published papers (Kronenberg, 2014).

Since the negative effects of the industrial revolution on agricultural areas was realised, on humans and nature - firstly the direct effects on nature (Carson, 1962), and more recently the indirect effects of intensification (Krebs et al., 1999) - the role of birds as ecosystem services as a research area has increased with emphasis, not only on the commodity and value of birds, but on how biodiversity is linked with the increase in service provision rather than focussing solely on bird diets (Kronenburg, 2014).

Birds do, however, predate on all insects, including beneficial predaceous insects (Martin et al., 2005). Mooney et al. (2010) meta-analysis revealed vertebrate insectivores (including birds, bats, and lizards) still reduce herbaceous insects by 39%, reducing plant damage by 40%. Birds are shown to be capable, through their high functional diversity and feeding habits, of playing a top-predator role in top-down biological pest control in global agro-ecological landscapes (García et al., 2018; Mooney et al., 2010) in both tropical (Maas et al., 2013; Milligan et al., 2016) and temperate environments (Barbaro et al., 2016; Garfinkel & Johnson, 2015). But to what extent biodiversity plays in providing this pest control service by birds is less studied (Porcel et al., 2018; Jonsson, Kaartinen & Straub, 2017).

4.2.3 Current research

Insectivorous birds are the functional group responsible for a pest control ecosystem service and have been studied more recently in relation to landscape heterogeneity, species richness, abundance of insectivores and different farming practices. This type of research helps our understanding on the links between current farming, wider landscapes, and how these relate to the provisioning of pest control services. For example, the presence of birds in three climatic areas: boreal, temperate and tropical, increased performance of plants due to an increase in bird abundance and avian insectivore ability to provide top-down trophic cascades to protect plants from insect herbivory (Mäntylä et al., 2011). Yet the link between biodiversity and pest control, rather than just the abundance of natural enemies and pest control, is less researched.

Previous studies have assumed that an increase in natural enemy abundance can be equated to a large prevalence of pest control service provisioning (Howe, Lövei & Nachman, 2009). However, a wide range of ecological interactions can complicate this link (Schmitz, 2007) so a more direct measure of predation is advised to be experimented within field conditions (Howe et al. 2009). Sentinel live prey (Garfinkel & Johnson, 2015; Boyle, 2012) and dummy caterpillars made with either dough or plasticine (Ferrante, Cacciato, & Lövei 2014; Loiselle & Farji-Brener, 2002; Mantyla et al., 2008; Rowland et al., 2008; Sam et al., 2015) are often used to estimate and infer predation from predators including birds, insects and mammals (Loiselle & Farji-Brener, 2002; Martin, 1987).

The research currently available, using sentinel prey experiments, have found that higher biodiversity measured through bird functional evenness and diversity (Barbaro et al., 2016; 2014), species richness (Milligan et al., 2016) and abundance of insectivores (Bereczki et al., 2014; 2015) support higher avian predation within forests and agro-ecosystems. However, the comparison of these ecosystem service studies is between the presence of an ecosystem service and the total absence of one, rather than comparison to a man-made one, i.e. comparison between types of farming systems that aim to increase biodiversity and ecosystem services, like LEAF and IPM as well as organic has not been well-studied. García et al. (2018) highlight that while such studies have been undertaken in tropical agro-ecosystems, there is a lack of studies demonstrating the role of birds in temperate systems, and more so within northern temperate agro-ecosystems like the UK, with a less diverse pool of insectivores than southern Europe (Dirzo & Raven, 2003). Farming systems have been shown to remain productive whilst reducing harm to biodiversity (Cunningham et al., 2013; Clough et al., 2011). Whether biodiversity on organic farms can provide pest predation to the same degree than a man-made chemical pest control can, is not well known.

Only three studies, known at the time of writing, use a comparison of farming practices of organic and non-organic to understand avian pest services available under these different farming techniques. Mols & Visser (2007) compare IPM farming with organic farming, or insecticide-sprayed versus unsprayed, and two others compare unsprayed farms with sprayed farms to show an increase in pest control services provided by birds on unsprayed orchards (Howe et al., 2009; Peisley et al., 2016). However, only six sites were used in Peisley et al. (2016) study as comparison between farm management intensity. This is the only known study that looks at farm, rather than plot level. Although Mols and Visser (2007) also explore farming management effects on avian ability to provide pest control, rather than measure predation rates, they infer predation through caterpillar density monitoring in areas with and without breeding great tits. A more recent study by García et al. (2021) investigated predation pressure in apple orchards from birds using nest boxes to find that the inclusion of nest boxes increased predation on apple pests and decreased pest pressure on apples. However, the impact of farm management strategies was not included in this study.

There is not yet a well-established relationship between biodiversity, pest regulators (natural enemies) and pest reduction potential (Daniels et al., 2017; Chaplin-Kramer et al., 2013; Letourneau et al., 2015; Samnegård et al., 2019), especially in perennial agro-ecosystems (Porcel et al., 2018). Although studies on predation rates and pest reduction have been increasing in recent years, especially with arthropods, our understanding of how *avian predation* rates transfer directly to pest abundance on farmland in temperate regions, is elementary. In comparison to similar studies, this study is an extensive comparison of organic compared to different types of non-organic managed orchards, including IPM, LEAF and conventional spray regimes, where the availability and abundance of insectivorous birds as the pest regulator over farms with differing levels of biodiversity, has been explored. From this it is possible to uncover whether a higher biodiversity on orchards generates a natural pest control ecosystem service or not. To the best of my knowledge, at the time of writing, this is the first extensive farm-level study comparing organic and non-organic levels of pest predation services available from birds, supported by biodiversity.

4.2.4 Objectives of study

Here, the insectivorous bird functional group has been measured in terms of abundance, density, species richness, and diversity on four different farm management types ranging in levels of biodiversity, where organic farms have the highest (Chapter 3). This will help in our understanding of whether supporting an insectivorous bird functional group can provide a pest control service to reduce pest abundances on apple orchards. This study uses 30 farms across a similar landscape across Herefordshire and its associated borders, with varying levels of management intensity on apple orchards to understand how changes in farmland biodiversity affect the insectivorous bird functional group, to ultimately impact *Lepidoptera* pest levels.

This chapter sets out to address three key questions:

1. The first objective of this study is to ask whether the insectivorous bird functional group varies in abundance, density, species richness, and diversity between farm types: organic, conventional, LEAF, and IPM?

2. The second objective is to understand how the level of insectivorous birds affects predation rates, assessed using artificial sentinel prey?

3. Thirdly, what are the implications of 1 and 2 for pest control, which is assessed through patterns in the abundance of key moth pests?

4.3 METHODS AND MATERIALS

All 30 orchards used were the same as Chapter 3, located within a $60km^2$ area in Herefordshire, Worcestershire and Gloucestershire, UK (*figure 2.2*). The orchard fields ranged between 0.6 Ha to 115 Ha in size. The management of each study orchard differed and were separated into four categories: conventional (7 farms), IPM (7 farms), LEAF (8 farms) and organic (9 farms). Full details of farm managements and site selection can be found in Chapter 2, *section 2.2*.

4.3.1 Insectivore bird community metrics

Data from Chapter 3 bird survey results were used for insectivore metrics. Birds were split into guilds and insectivores were analysed separately. Insectivores were chosen based on being insectivorous during the breeding season. These were not just restricted to gleaners, ground probers, canopy foragers and hawkers, but all functional groups of insectivores were included so to capture the predation of pest moths at all life stages. A list of insectivores used for this chapter is highlighted in *Appendix C*.

4.3.2 Pest predation rates

The decision was made to use artificial sentinel prey, rather than live prey, to measure predation rates because of three main factors: experiment execution is simple; identification of predation marks is efficient; and it's an inexpensive method for monitoring in field conditions without the need for sophisticated equipment (Howe et al., 2009; Lövei

& Ferrante, 2016). However, the use of live prey in exclusion experiments is thought to provide more realistic. Therefore, it is important to take into account that artificial sentinel prey data may underestimate true predation rates of birds (Lövei & Ferrante, 2016). More details about this decision are outlined in Chapter 2 *section 2.3.6.*

Both plasticine (green and white) and dough (cream) were used as the sentinel prey material. Two key pest caterpillars of orchards were being mimicked: *Lepidoptera* species (i) *Cydia pomonella L.* codling moth larvae (cream dough and white plasticine) and *Archips podana (Scolopi)* fruit tree tortrix larvae (green plasticine).

Pest presentation stations were set up at the same point count locations as bird surveys (Chapter 3) and took place over two rounds: (i) plasticine prey between April and August 2016 and (ii) dough prey between December 2016 and March 2017. Each sentinel prey was used to resemble the life stages of the two *Lepidoptera* species. Codling moth overwinters as non-feeding larvae under loose bark of apples trees and emerges in summer again to feed on apples, along with fruit tortrix caterpillars in the spring and summer.

Plasticine

Pest presentation stations were set up with eight green and eight white plasticine caterpillars: 16 prey in total, with green caterpillars on leaves and cream caterpillars on the bark of one apple tree (*figure 4.1*).

Dough

At each pest presentation station, ten dough sentinel prey were planted at each point count, on the trunk of one apple tree (*figure 4.1*)

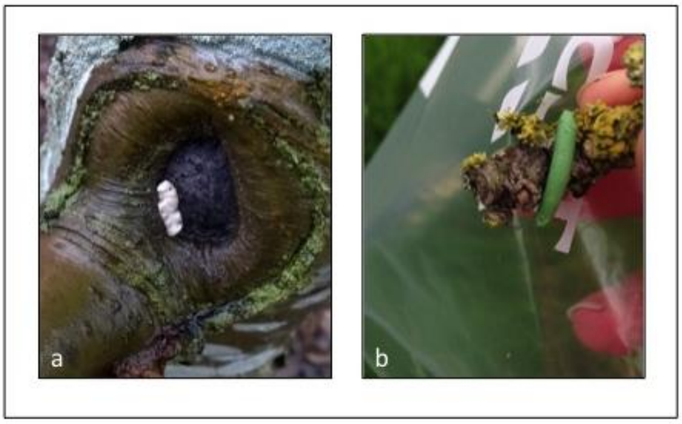


Figure 4.1 – Sentinel prey types; a) dough caterpillar placed on the truck of apple trees over winter, to resemble over-wintering coding moth larvae; b) green plasticine prey placed on leaves or on branch ends to resemble tortix moth caterpillars.

The identification of predation marks had to be in-situ as sentinel prey would change shape easily and any predation attempt may have been lost if the shape was lost in transit. For full details of sentinel prey materials methodology, please see Chapter 2 *section 2.3.6*.

4.3.3 Measuring pest moth levels

Sex pheromone traps were used as population estimates for the specific moth species of interest, usually used to determine a threshold population for optimum time to use chemical insecticides (Carden, 1987; Reddy & Guerrero, 2001), but also to infer moth populations (Riedl & Croft, 1974). Pheromone traps were purchased through Agralan, an agricultural management company, for three main apple pest moth species: codling moth (*Cydia pomonella L.),* fruit tree tortrix *(Archips podana (Scolopi)),* and apple ermine *(Yponometua malinellus)*.

One trap can attract each specific moth species from two hectares in a standard orchard. All orchards in this study are standard bush orchards, traditional, or semi-standard bush. So, the two-hectare radius sample could be used per farm. Larger farms grew multiple types of apple, therefore one trap was used per apple type, based on the end consumption (juice, dessert, or cider). The mapping programme QGIS (QGIS Development Team, 2015), was used to ensure each trap was centred within a two hectare radius of orchard. At each site, three traps were placed in a central orchard location, using GIS to measure the central location, per apple type in 2016, between 25th June to 15th September. This was to ensure peak adult moth flying seasons were included for all three moth pest species, as recommended by Agralan. Pest traps were placed at least 10m apart from one another, so to avoid any interference from pheromones, and the traps were hung with wire at head height (approx. 1.6m) in the apple tree canopy (Alford et al., 1979). Traps were checked every four weeks where male moths were counted, and the sticky pads and pheromone renewed. If another pheromone trap was known to be in the same orchard by the farmer, and within 2ha, pheromone traps were *not* erected at these sites – to avoid underestimating pest moth abundances.

For more details of pest moths in orchards and history of the use of pheromone traps and thresholds of pests needed before being considered a pest outbreak, please see Chapter 2, *section 2.3.7*.

4.4 DATA ANALYSIS

All analysis for Chapter 4 followed the same structured and statistical packages as data analysis for Chapter 3. Short details will be provided here but full details can be found in Chapter 3.

4.4.1 Model simulation using GLMMs

A 'global' model was constructed with the response variable for all models as the insectivore community metrics, prey type or pest abundance and all independent variables considered, *a priori,* to be ecologically important are the same as Chapter 3 including: observer, day number, woody cover, insecticide use, orchard size (ha), temperature (°C), farm management types, and types of apples. Continuous fixed effect variables were standardised within R using scale function. This global model was put through the same model averaging process as in Chapter 3.

Overdispersion was tested using the dispersion parameter and model interactions were tested for using AIC in model interactions and model non-interactions in anova.

Correlation was tested in the same way as Chapter 3, using a correlation coefficient on continuous variables in the model as well as a variance inflation factor (VIF) test on the fitted model. Any variable with a VIF value over 5 was removed from the model to simplify and improve accuracy (Chapter 3).

4.4.1.1 Model averaging process

The global model was put through the same information theoretic (IT) approach used as in Chapter 3, where 'top models' are chosen using AIC Delta < 6 AIC values (Harrison et al., 2017). A model average can be acquired with model average parameters including standard errors and confidence intervals taken using the top models.

The 'conditional' model average is chosen, as in Chapter 3, over the 'full' average as the conditional average is used during construction of model average confidence intervals. The full averages for this chapter analyses have been reported in *Appendices M, O, P, Q*.

4.4.2 Insectivores abundance, species richness and diversity

Abundance, species richness and diversity of insectivores were calculated using GLMMs. Poisson errors were used for the modelling of count data in abundance and species richness models and Guassian errors were used for diversity. The response variable for each model was the insectivore community metric (abundance, species richness and diversity), with increased restrictions for maximum likelihood. Overdispersion was tested for but none was found, thus there was no need to use negative binomial or zero-inflated errors in this analysis (Crawley, 2007; Harrison, 2014). Interactions were tested between farm management and woody cover (Chapter 2, *section 2.3.4*). Orchard size was significantly correlated to woody cover, so orchard size was removed from the global model, as the proximity and amount of woody cover was deemed more ecologically important *a priori* than orchard size. The random effect had a nested structure of visit number within farm ID to account for repeated data.

The IT approach, described above was used for this analysis, starting with a global model where likely interactions were also testing to include in the global model, fitting all sub-sets of the model and then ranked using AIC.

4.4.3 Density of insectivores

Orchard insectivore densities were calculated using the data collected from all bird density in Chapter 3 using Distance 7.0 Alpha (Thomas et al., 2010). Insectivores were isolated through selecting each insectivore species in the model data filters and put through the same model definition properties as total bird density analysis in Chapter 3. These model definition properties for each detection function were half-normal key function with simple polynomial series expansion. Six intervals of 12.5m were chosen and truncated in Distance at 75m instead of 100m, as in Chapter 3. Full details on density estimates are found in Chapter 3, *section 3.5.2*.

4.4.4 Predation rates

Predation rate response variables were constructed through binding together number of preys attacked with number of preys not attacked, using the ‘cbind’ function in R, library (lme4). These response variables produce the probability that a sentinel prey item was attacked during the 24 hours period it was exposed to predators. Mammal, arthropod, and bird attack rates were formed with the cbind function in R (R Core Team, 2015). The dependant variable 'prey type' is the probability that the sentinel prey will be attacked by birds, arthropods, mammals and total predation. Initially, analysis showed strong prey type differences. Initial analysis included all prey types and then a focus on prey type 1 (dough) as predation rates for prey type 2 and 3 (plasticine) were much lower than dough, as also seen by Lövei & Ferrante (2016) and Sam et al. (2015). This is most likely due to the energy return received from predating on dough rather than plasticine (Sam et al., 2015), mimicking natural avian reactions to live prey. Probabilities of predation were calculated out of the model odds ratio using back transformation (Crawley, 2013).

The GLMM model structure used binomial errors to assess predated against non-predated prey type 1 (Crawley, 2013), with 'farmer' as the random effect, using the link function 'logit' used in binomial error structures. Simple GLMMs were created, using the same process as previous analysis (Chapter 3) and then fixed effect variables were added that were important, *a priori*, to reach a global model for the model averaging process. Four model averages and confidence intervals are produced to include farm management and insectivore community metrics of abundance, species richness, and diversity as the dependant variables. Fixed effect variables included in the global model are apple type, day number and orchard size. Unlike for insectivore community composition models, woody cover was not kept in the final global model because it is highly correlated with orchard size, and when testing significance between orchard size and woody cover in explaining the global model, woody cover was not significant. For this reason, woody cover was omitted.

Only one mammal attack was seen on artificial caterpillars in total so was no included during analysis. Arthropod and bird predation rates were investigated but only bird attack rates were analysed and reported. This was because there was no significant influence of including farm management as a fixed effect during analysis with arthropods, thus arthropod predation rates were not affected by the type of farm management.

4.4.5 Pest moth analysis

Three separate moth traps, bated with pheromones to attract 3 specific pest moth species, were at each farm location. Moth pests were analysed separately as each moth trap attracted a different moth species and each species caused different types of damage on the apple crop. Repeat measures of farms were taken depending on the types of apples grown. If a farm held cider, dessert, and juice apples then there were 3 moth trap locations on an orchard, one per apple type. The study was over one year so the farmer was used as the random effect. As I was interested in finding out the impact of farm management category (LEAF, organic, IPM and conventional) and insecticide usage (broad-spectrum and specific) to moth pest abundance, these dependant variables were analysed in separate models. Thus, each moth pest had two analyses with i) farm management and ii) insecticide data.

Other dependant variables deemed significant during model selection were day number, day number squared, apple type and orchards size. Day number squared was used to assess whether moth abundance followed a curved pattern, with a peak mid-season, rather than linear through the year, as birds did.

Total insecticide and number of active ingredients were not used as they were correlated with specific and broad spectrum insecticide data. Woody cover and orchard size data was correlated, however orchard size was used in this analysis instead of woody cover as the moth pests I have analysed are not linked to woodland or hedgerow species; their predominate food plant are apple trees *(Malus).*

As count data was used for the abundance of moths, a Poisson error family was chosen within these mixed models. Overdispersion tests found the residual degrees of freedom to be unequal, well outside the required boundaries (<2) to the residual deviance of the model; the variance was not equal to the mean using Poisson error distribution. Thus, to account for this overdispersion, negative binomial error distribution is needed during model construction (Bolker et al., 2009).

4.5 RESULTS

Overall results show that insectivorous bird species were dominant in organic orchards, and so too were predation levels on dough sentinel prey. Pest levels on farms did not show significant difference between each other. The links between daily predation rates with insectivores show the presence of a functioning pest control ecosystem service by insectivores on orchards with the highest biodiversity - organic.

4.5.1. Insectivorous birds on orchards

Insectivores were chosen based on bird species feeding on insects during the breeding season. 39 species of insectivorous birds were recorded on orchards during the study period (*Appendix C*). I looked at abundance, species richness, Shannon diversity and density as measure of insectivore biodiversity.

Furthermore, motion-censored cameras caught a variety of birds attacking artificial prey. Birds caught on camera were found in organic and LEAF farms. Insectivorous bird species that either had direct contact with artificial prey, or were caught on camera, were european robins (*Erithacus rubecula)*, blackbirds (*Turdus merula*), blue tits (*Cyanistes caeruleus)*, great tits (*Parus major)*, jays (*Garrulus glandarius*), long tailed tits (*Aegithalos caudatus*) and a redstart (*Phoenicurus phoenicurus*).

4.5.1.1 Insectivore abundance

From *figure 4.2* and the global model average, it is visible that organic and LEAF farms have significantly higher abundance of insectivorous birds on orchards, evident as these parameter confidence intervals do not span zero. Woody cover shows a negative impact on insectivore abundance and day number has a significant positive effect, so as the day number increased throughout the year, insectivore numbers increased. All other parameters such as apple type, observer and temperature had no significant effect (positive or negative) on insectivore abundance.

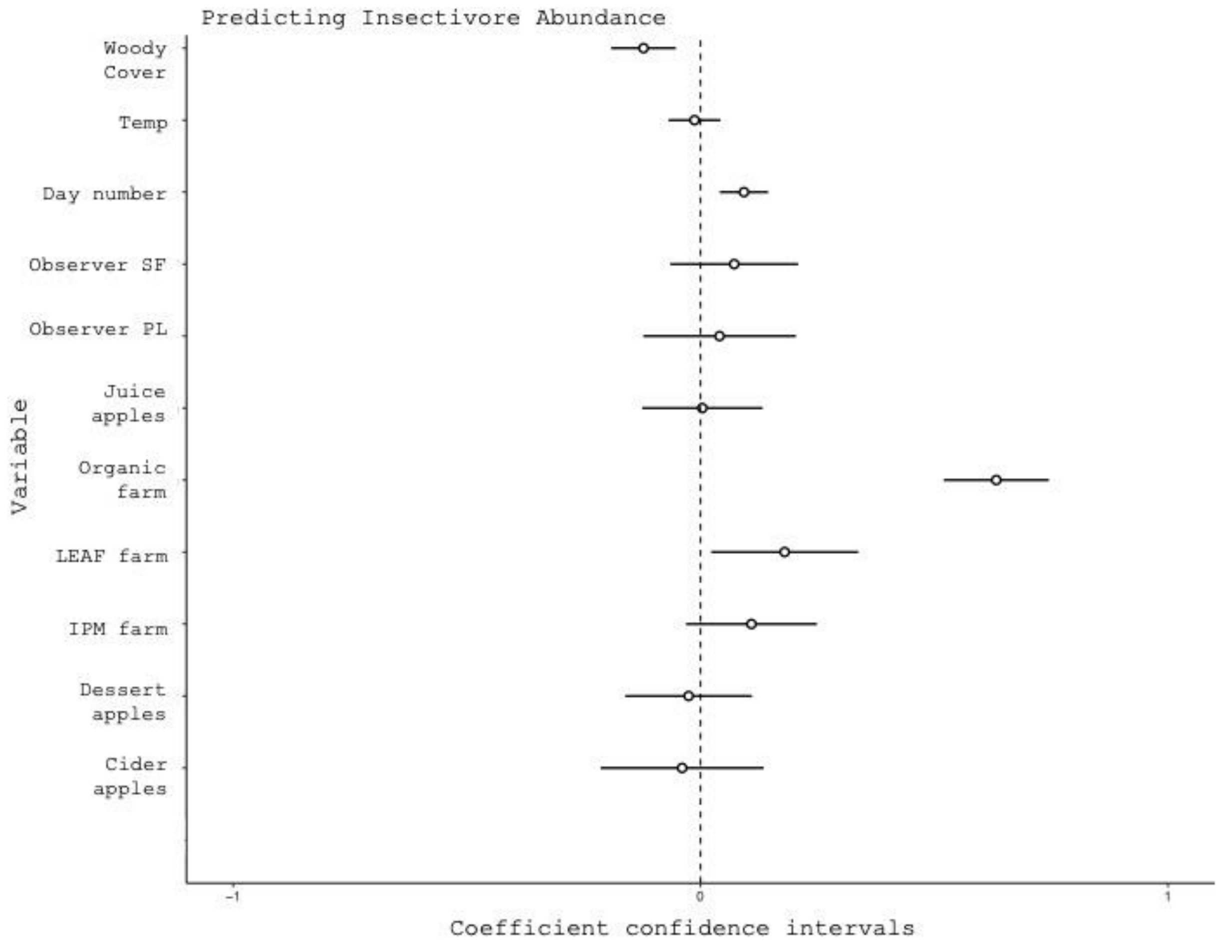


Figure 4.2 Insectivore abundance GLMM 'conditional' model average and confidence intervals using top models, Delta < 6. For 'full' model averages see *Appendix I*.

4.5.1.2 Insectivore species richness

Like abundance, organic orchards significantly affected insectivore species richness, positively. LEAF and IPM farms showed no difference in comparison to the intercept (conventional), in contrast to abundance where LEAF was also significantly influencing in insectivore abundance (*figure 4.3*). Increased woody cover continued to have a significantly negative effect on insectivore species richness. An observer effect is shown here with Observer PL viewing significantly less species richness than the intercept, Observer CP. All other parameters in the model show no significance in insectivore species richness.

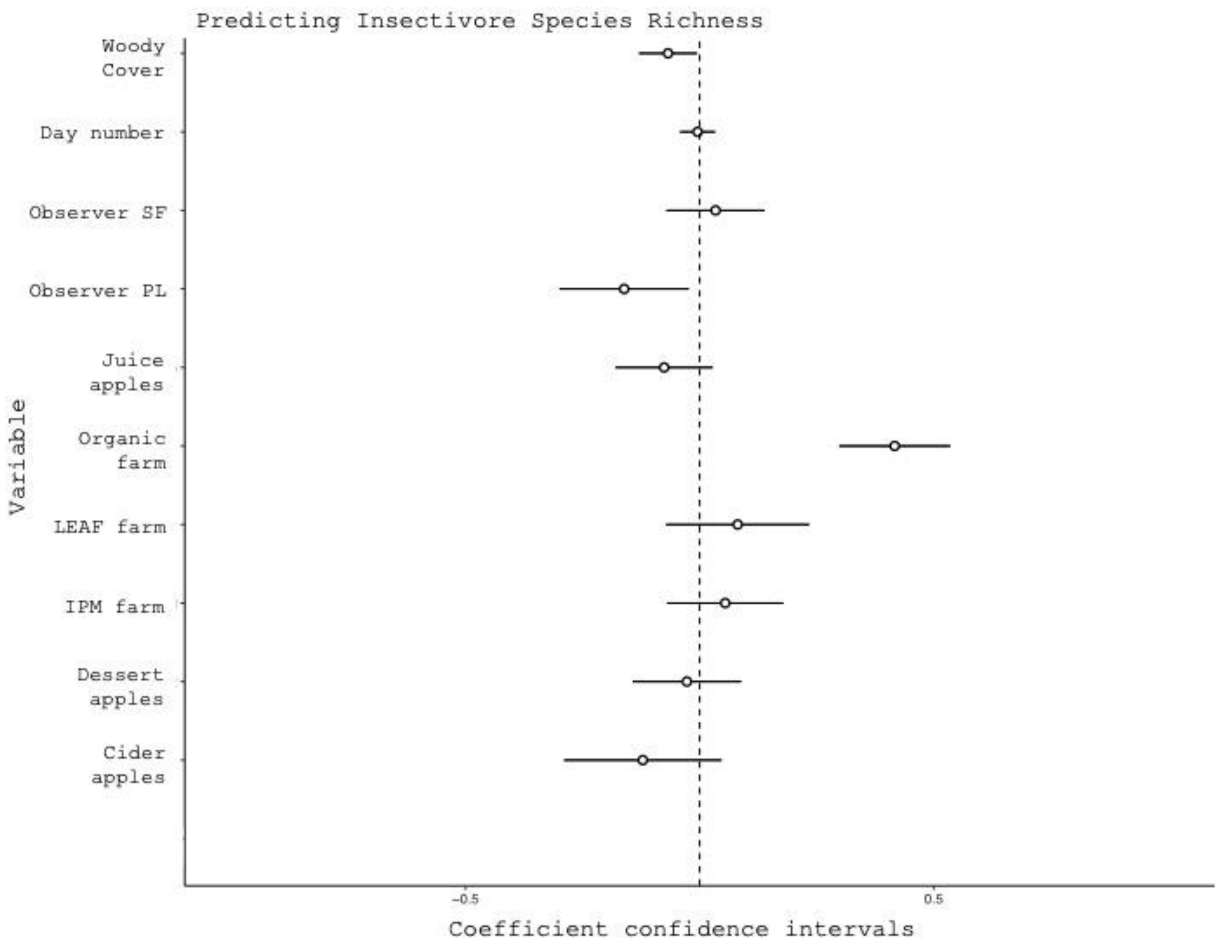


Figure 4.3 Insectivore Species Ricness GLMM 'conditional' model average and confidence intervals using top models, Delta < 6. For 'full' model averages see *Appendix I*.

4.5.1.3 Insectivore diversity

Again, similarly to species richness and abundance, insectivore diversity is significantly higher in organic orchards, in comparison to conventional. LEAF and IPM farms show no significance on insectivore diversity here (*figure 4.4)*. Woody cover presence, again, significantly reduces insectivore diversity on orchards, so too Observer PL. All other parameters have no significant impact on insectivore diversity.

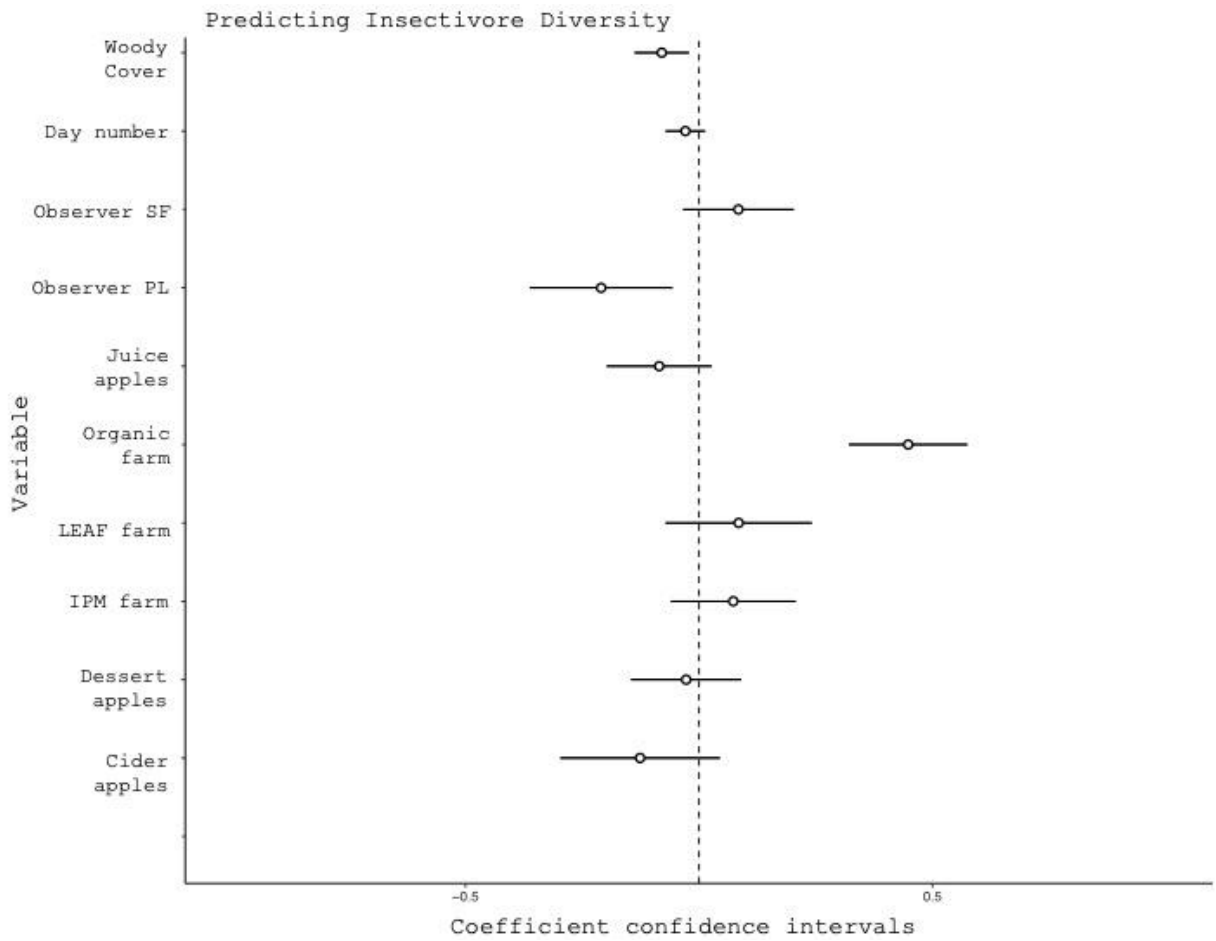


Figure 4.4 Insectivore Diversity GLMM 'condtional' model average and confidence intervals using top models, Delta < 6. For 'full' model averages see *Appendix I.*

4.5.1.4 Insectivore density

The density of insectivorous birds per hectare is shown in *figure 4.5*, with organic farms displaying the highest density of insectivore (13.66) across the four farm management categories, with statistical significance.

Comparisons of LEAF - conventional farms and IPM - conventional farms were not significantly different from each other, resulting from Z tests. Although Z tests shown on *figure 4.5* compare only to conventional farming, IPM and LEAF were both very similar to each other (*Z= -0.01, P = 099)*. Organic was significantly greater than both IPM (*Z=-5.55; P<0.01*) and LEAF (*Z= - 5.2; P<0.01*). Density detection functions (*Appendix J*) support previous results, that organic orchards support high levels of insectivorous birds, using a different monitoring and analysis technique.

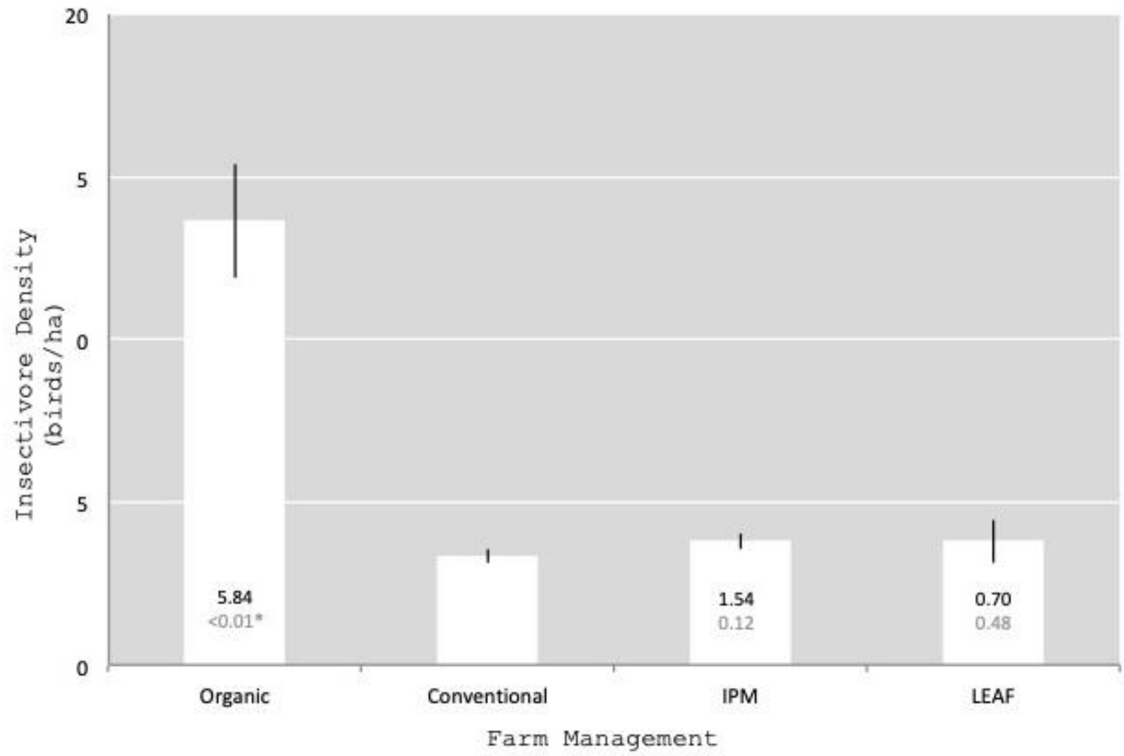


Figure 4.5 Insectivore density averages (organic = 13.66, conventional = 3.32 , IPM = 3.8, LEAF = 3.81) with standard errors, shown as number of birds per hectare, results from Distance analysis. The numbers within the bars indicate the Z (black) and P values (grey) during two-way comparisons, between conventional farming to other three categories (LEAF, IPM and organic).

4.5.2 Sentinel prey

After initial data analysis of predation on plasticine and dough sentinel prey, predation probabilities were higher on green plasticine than white, but both green and white plasticine were much lower than dough sentinel *prey (table 4.1; 4.2*). Due to the impact that prey type had on initial mixed models, the decision was made to only use dough for the remaining analysis, incorporating landscape variables and bird insectivore community metrics.

Table 4.1 Daily Predation Rates (probabilities) for Arthropod and Birds. Attack rates by birds in general were low. Arthropod attack incidences were much higher, just over 50% of dough prey were attacked by arthropods.

Prey Type	Arthropod	Birds
Dough	0.557	0.034
Green Plasticine	0.23	0.027
White Plasticine	0.12	0.008

Table 4.2 Three types of prey and their predation probabilities shows that pastry was predated on most, so was used as the primary sentinel prey during mixed model analysis. Probabilities were calculated in R using back transformation (Crawley, 2007; R Core Team, 2015).

Prey Type	*Farm Management*			
	Conventional	**IPM**	**LEAF**	**Organic**
Dough	**0.02240306**	**0.01368062**	**0.0194303**	**0.1167988**
Green plasticine	0.01792626	0.01092734	0.0155381	0.09529772
White plasticine	0.005147927	0.00312217	0.004454394	0.02899523

4.5.2.1 Dough sentinel prey and farm management

Probability of dough prey being predated on from birds, in relation to farm management group, was modelled. Here, *figure 4.6*, shows organic orchards had significantly higher levels of avian predation compared to conventional orchards, the intercept. LEAF Marque farms had significantly less predation attempts than conventional, and on IPM management farms there was no significant difference in predation. Apple type also shows significant influence over predation levels, with dessert and juice apples showing significantly higher levels of predation, whereas cider shows no significance in increasing predation levels.

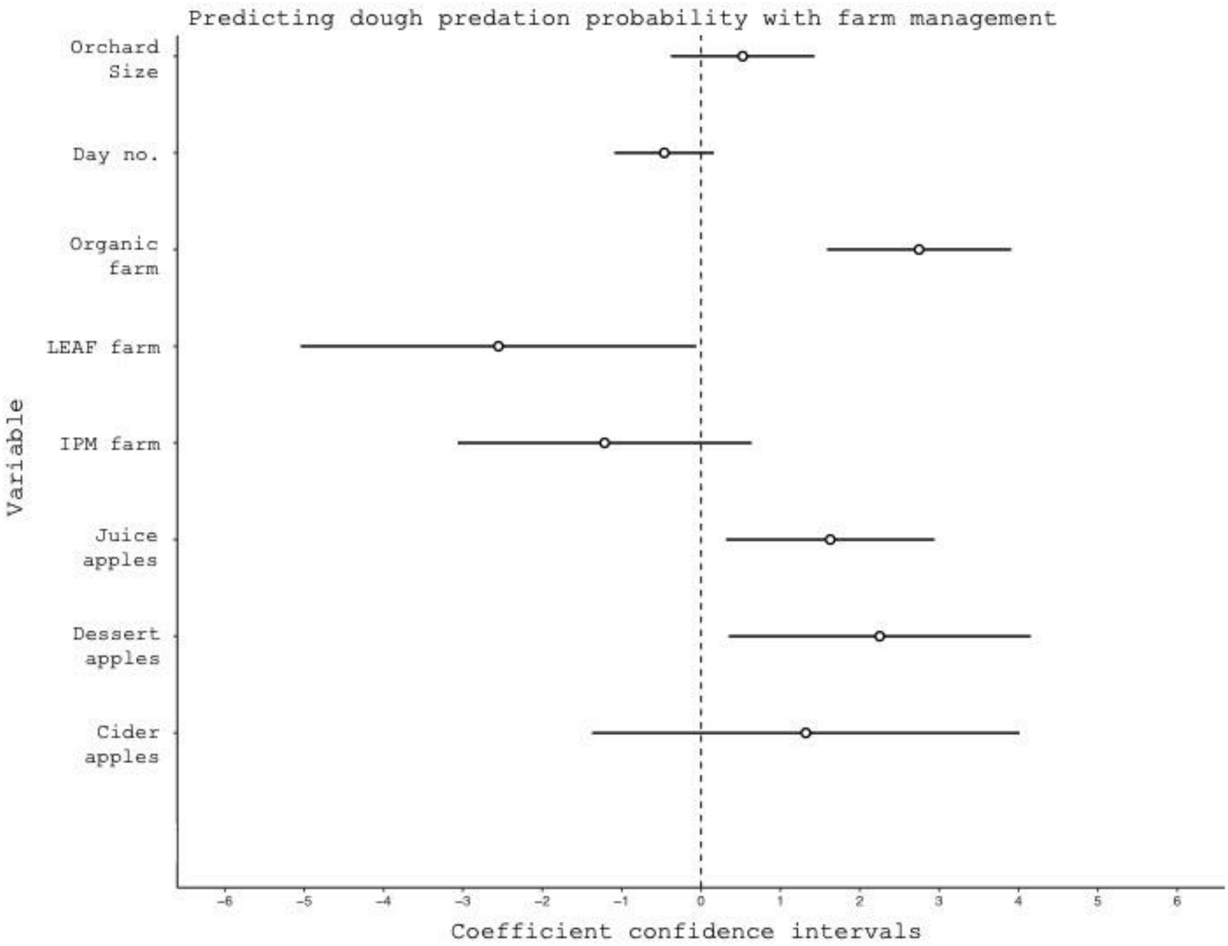


Figure 4.6 Dough predation probability and farm managament GLMM 'conditional' model average and confidence intervals using top models, Delta < 6. For 'full' model averages see *Appendix K*

4.5.2.2 Dough sentinel prey and insectivore abundance

Predation probabilities in dough sentinel prey, significantly increases with the abundance of insectivores (*figure 4.7*). As orchard size increases, predation levels show a significant decrease. Orchards, which sell juice apples, show a significant increase in predation levels. All other fixed effect variables show no significant impact on predation.

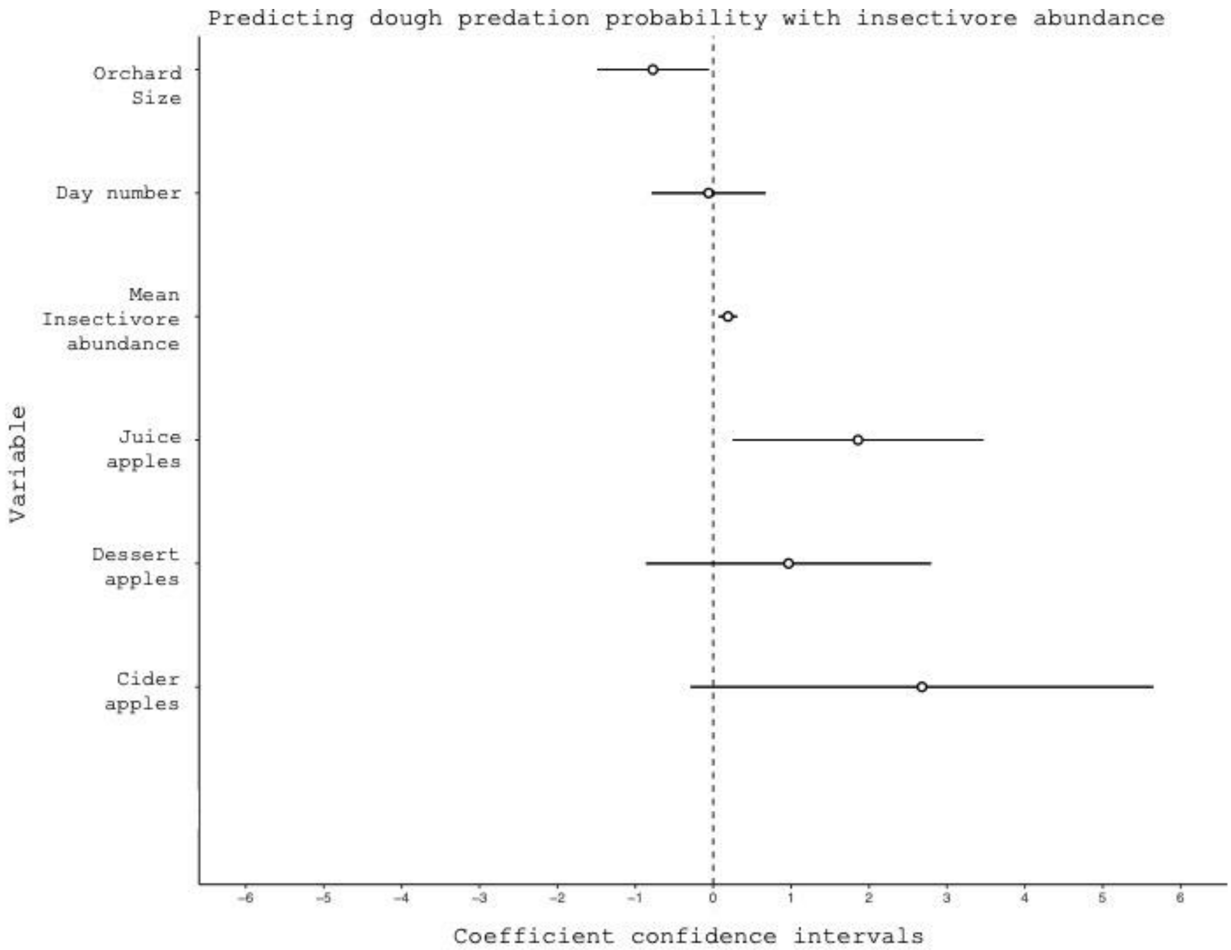


Figure 4.7 Dough predation probability and insectivore abundance GLMM 'conditional' model average and confidence Intervals using top models, Delta < 6. For 'full' model averages see *Appendix K*

4.5.2.3 Dough prey and Insectivore species richness

Like predation and insectivore abundance, predation probability was significantly higher with increased insectivore species richness (*figure 4.8*). Juice apples had significantly higher levels of predation than other apple types and smaller orchards had significantly higher predation rates. All other variable in the global model did not have a positive or negative effect on predation rates in orchards.

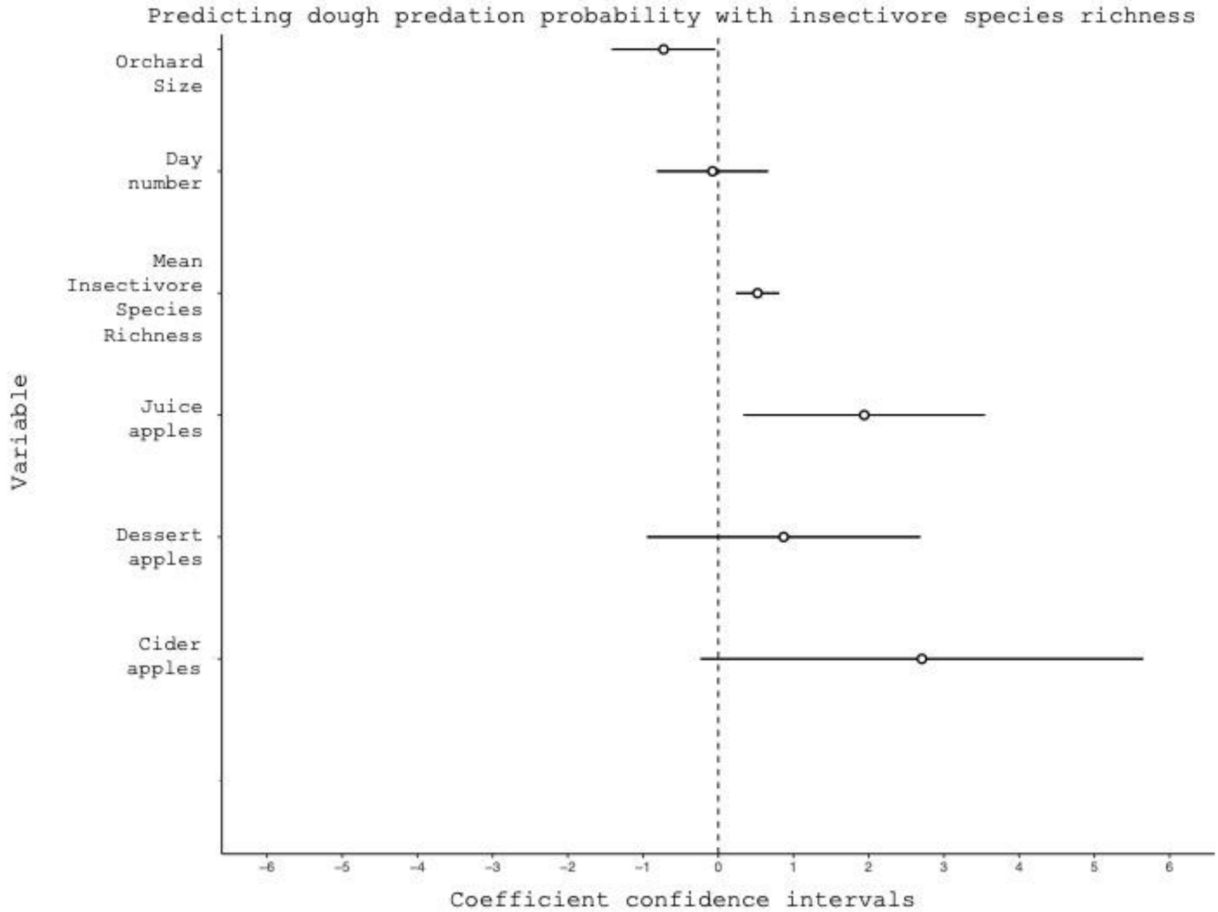


Figure 4.8 Dough predation probability and insectivore species richness GLMM 'conditional' model average and confidence intervals using top models, Delta < 6. For 'full' model averages see *Appendix K*

4.5.2.4 Dough sentinel prey and insectivore diversity

As the insectivore diversity parameter confidence intervals spans zero, there is not strong enough evidence to show that insectivore diversity is an important factor in sentinel prey predation (*figure 4.9*). Cider and juice orchards had significantly more predation in this model and as orchard size increased predation significantly reduced. All other variables did not influence predation.

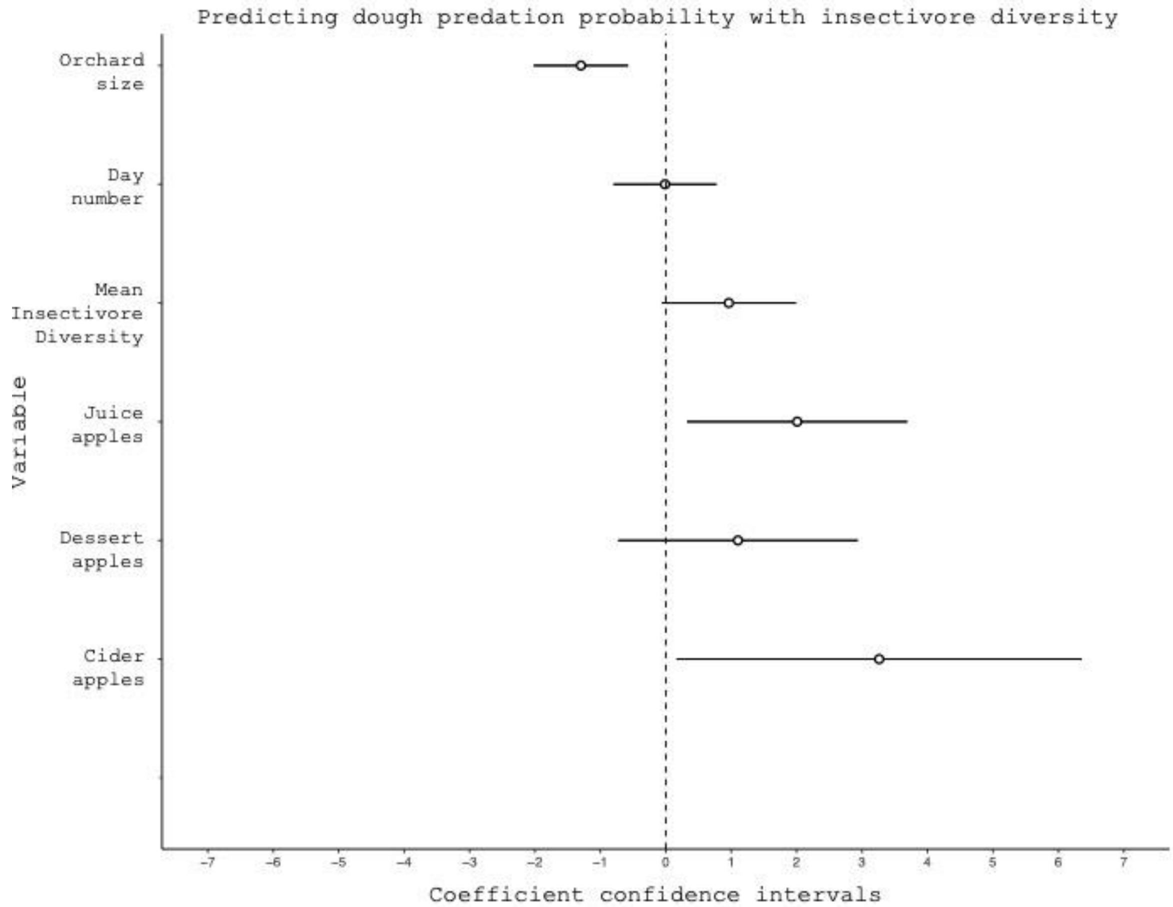


Figure 4.9 Dough predation probability and insectivore diversity GLMM 'conditional' model average and confidence intervals using top models, Delta < 6. For 'full' model averages see *Appendix K*

4.5.3 Pest moths

Pest moths were analysed separately due to the different moth life cycles in each species. For example, apple ermine is known to arrive as adults later in the year than large fruit tree tortrix and codling moth. Codling moth also overwinter under tree bark, providing an additional winter food source for birds (Solomon et al., 1976).

As day number was included in the model, Julian days starting from the first pheromone trap set up date – on 25th July 2016, was used.

Moth data was also analysed with insecticides and farm management separately - there are two GLMM models for each. As moths are directly affected by insecticide sprays, this was an important fixed effect to test for significance and the type of farm management is correlated to insecticide use. Often IPM farms may use very different amounts of insecticide; thus, insecticide amounts were specifically analysed here.

4.5.3.1 Codling moth with farm management and insecticides

There is no significant effect of farm management type on the increase of codling moths in orchards. The model average confidence interval graphs (with management and insecticides) have the same significant parameters (*figure 4.10*). Day number shows a significant negative relationship to codling moth abundance in both models, showing reduced moth abundance as the surveying season goes on all types of farms. Dessert apple was the only type of apple that had significantly less codling moth pests in the models. All other variables including the amount of specific and broad-spectrum insecticides, showed no significant impact on the number of codling moth numbers recorded.

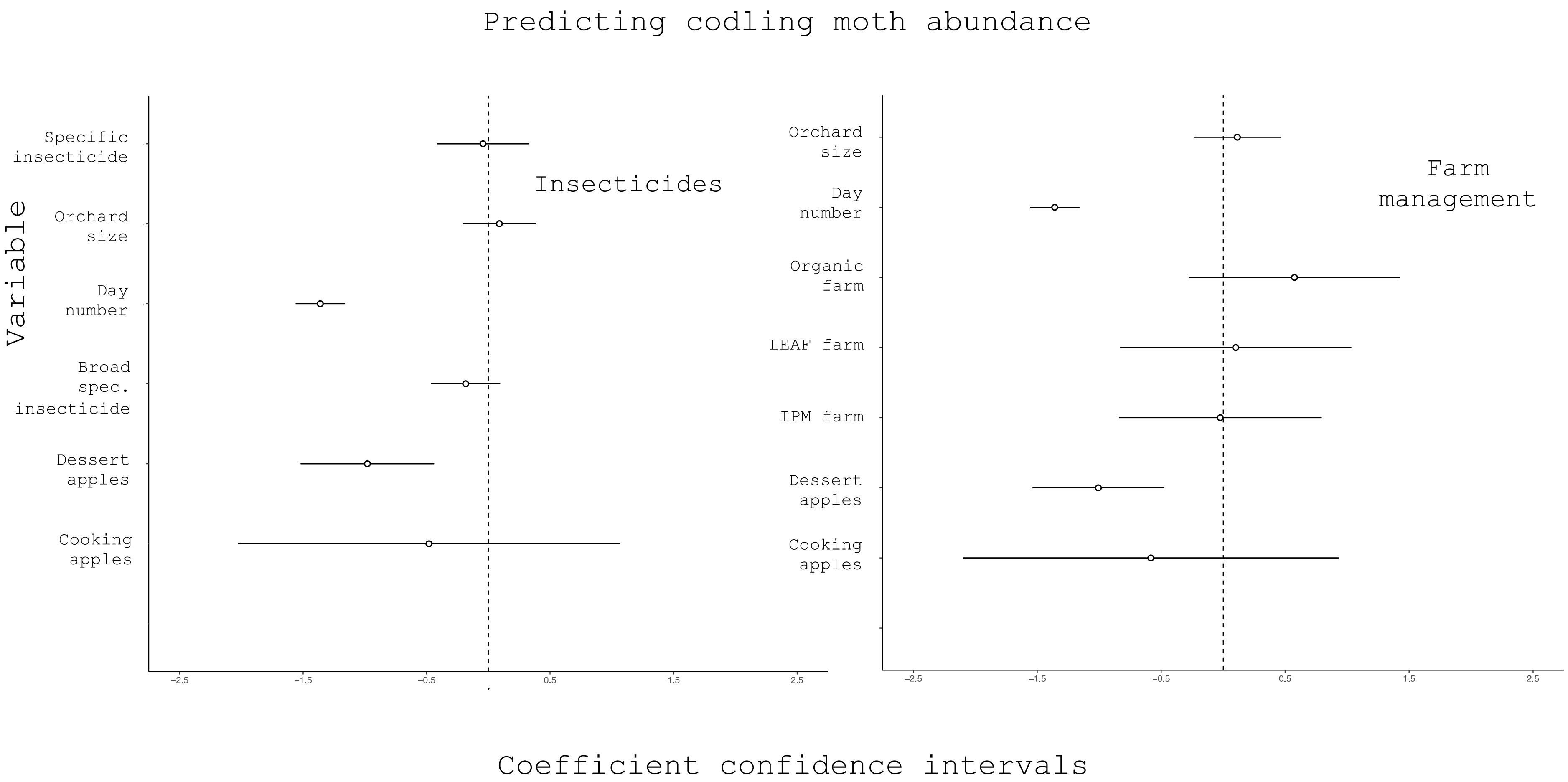


Figure 4.10 Model average confidence intervals using 'conditional' model average results of codling moth abundance with response variables of insecticide application, where broad spectrum and specific insecticide have been used separately (left) and farm management (right). The model intercept is conventional farming with cider apples. 'Full' model averages for codling moth models are in *Appendix L.*

4.5.3.2 Apple ermine moth with farm management and insecticides

From the model average (*figure 4.11*) there is no significant effect of specific insecticide on the abundance of apple ermine pest moth, however a significant positive effect of broad-spectrum insecticide is seen – when more broad scale insecticide was used the abundance of apple ermine pest moth was significantly higher. Organic managed orchards showed a significant negative effect on the abundance of apple ermine pest moths. As orchard size increased apple ermine abundance significantly decreased in the management model, however this was not a significant parameter in the insecticide model. Apple ermine show a significant signal of having a second generation of adults, with increased abundance as day number increases in both models across all farm management types. Type of apple had no significant impact on apple ermine abundance.

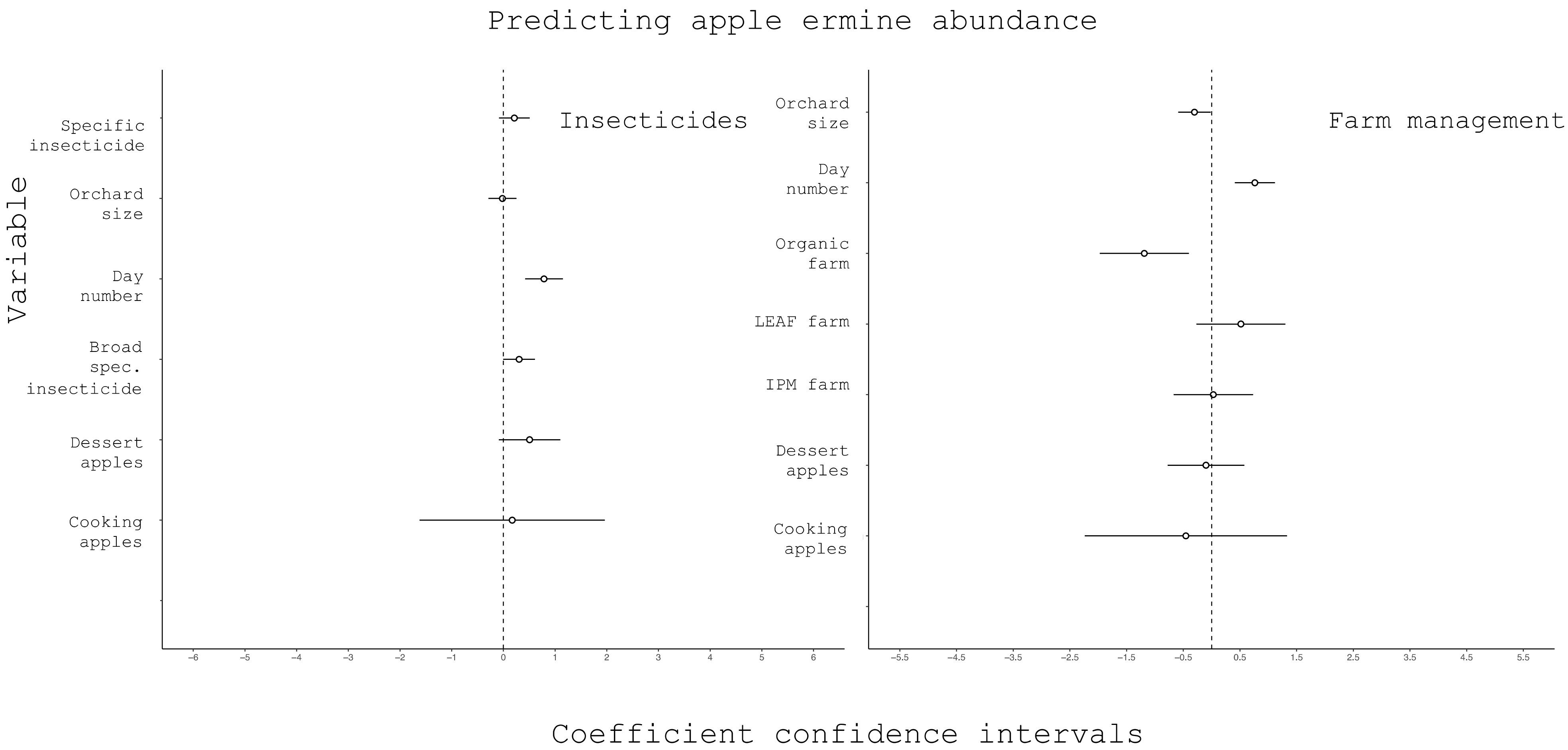


Figure 4.11 Model average confidence intervals of Apple Ermine, using 'conditional' model average results with response variables of insecticide application, where broad spectrum and specific insecticide have been used separately (left) and farm management (right). The model intercept is conventional farming with cider apples. 'Full' model average for apple ermine moth are found in *Appendix M.*

4.5.3.3 Fruit tree tortrix moth with farm management and insecticides

The fruit tree tortrix model average with confidence intervals (*figure 4.12*) shows fruit tree tortrix moth abundance is significantly reduced in numbers with the increased use of specific insecticide, whilst broad-spectrum has no effect on this pest abundance. Orchard size has a significant positive effect on the abundance of fruit tree tortrix in both the management and insecticide models. LEAF farms here show a significant reduction in the abundance of fruit tree tortrix in comparison to the intercept (conventional farming), whilst there is no difference seen between all other farm types and conventional. Day number shows in both models that as time goes on in the season fruit tree tortix moth decreases across all farm management types. Apple type, or any other parameter, has no significant effect on abundance of the fruit tree tortrix pest moths.

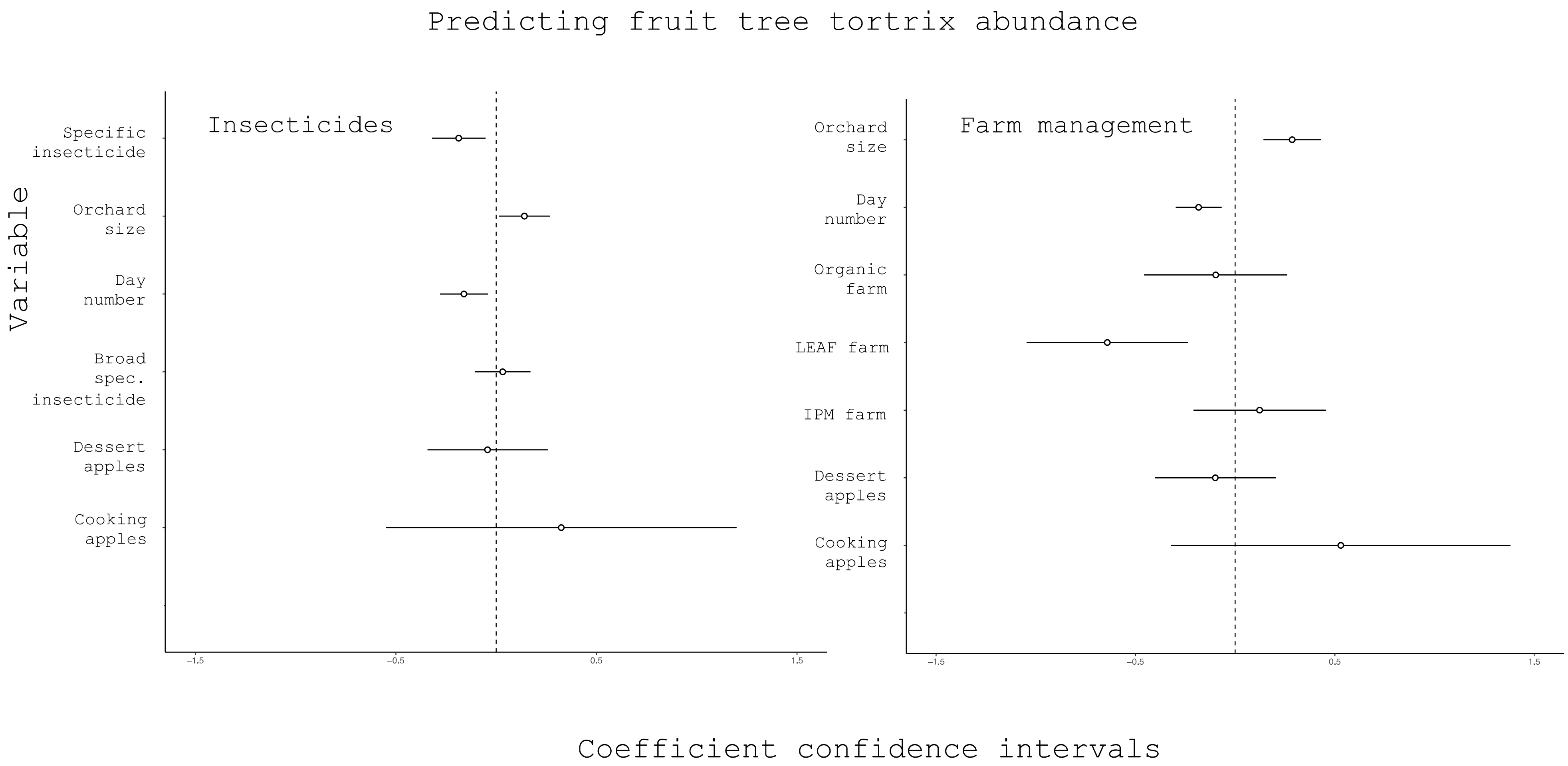


Figure 4.12 Model average confidence intervals of Fruit Tree Tortrix (FTT), using 'conditional' model average results with response variables of insecticide application, where broad spectrum and specific insecticide have been used separately (left) and farm management (right). The model intercept is conventional farming with cider apples. 'Full' model average results for fruit tree tortrix are found in *Appendix N.*

4.6 DISCUSSION

There is a presence of a functioning pest control ecosystem service from insectivorous birds on three economically important *Lepidoptera* pests in UK organic apple orchards, the codling moth, fruit tree tortrix and apple ermine. Organic farming is beneficial to the abundance and species richness of insectivorous birds, in line with general consensus where organic farming benefits biodiversity (Chapter 3; Hole et al., 2005). Firthermore, increased insectivore bird community, supported by organic farming and not landscape features, provides a sufficient pest regulation ecosystem service to keep pest moth levels similar across all four farm managements, including on organic farming. This highlights there are two separate agro-ecosystems at play – one where chemical measures control pests and the other where birds act as the natural pest control.

4.6.1 Insectivore abundance and diversity across farm managements

Like Chapter 3, insectivorous bird abundance, diversity, and species richness followed the same pattern with organic orchards hosting higher levels of avian insectivore biodiversity. The density of insectivores also follows the same trend, with a threefold increase in avian insectivore density on organic orchards. This result follows the majority of studies looking at how organic farming can support biodiversity (Bengtsson et al., 2005; Feber et al., 1997; Hardman et al., 2016; Hole et al., 2005). Studies on apple orchards in particular have had similar results with insectivorous bird abundance increasing in organic orchards when compared to conventional (Katayama, 2016; Mangan, 2017), increasing the natural enemy community of organic orchards. Although IPM and LEAF are meant to be less intense agricultural systems, they do not show a significant increased abundance, species richness or diversity of insectivore birds here, in comparison to conventional. On apple orchards of south-eastern France, the same pattern is seen - organic orchards held the highest amount of natural enemies in comparison to IPM and conventional farming. Our results contradict those of Geiger et al. (2010), who found that organic farming increases plant and carabid species diversity, but not breeding birds. Others have described grassland bird responses to organic farming as 'variable' (Quinn et al., 2012).

However, it is important to note here that IPM farms were self diagnosed at the start of the study so are likely to be very simialr to conventional. Conventional farms may use the same low level 1 IPM techniques but not actively class themselves as IPM, which explains the similarity in results between IPM and conventional, but still doesn't explain the similarity between LEAF and conventional.

Contrary to expectations, and to current studies looking at wider landscape heterogeneity in relation to agricultural biodiversity, our results showed that insectivore abundance, species richness, and diversity decreased with woody cover increases. This result mirrors those of Barbaro et al. (2016), who showed that with increased landscape complexity bird functional diversity decreased. Another similar study from Lemessa et al. (2015) also shows parallels with these insectivore abundance findings, demonstrating how birds did not respond to increased tree cover even in tree-poor landscapes, highlighting how there is enough habitat for insectivorous birds to continue the top-down control of crop pests. One reason organic farming, rather than habitat provision, supported higher insectivore abundances, could be due to the lack of food sources in non-organic farming systems due to chemical pest management reducing natural enemies as well as pest populations (Fountain & Harris, 2015; Laure et al., 2014; Marliac et al., 2015), with evidence that natural enemies are affected more than pests (Bengtsson et al., 2005). Without a food source available, birds – especially insectivores that will be feeding on different life stages of those insects - will not use a "good" habitat area if their feeding requirements are not able to be reached (Solomon et al., 1976).

Many studies give conflicting results, where increased landscape complexity and increased tree cover supports increased bird diversity and species richness (Tscharntke et al., 2008; Tews et al., 2004; Gove et al., 2008; Wretenberg et al., 2010) and where agricultural field edges gave enough edge habitat that avian species richness also increased (Kross et al., 2016). One reason for this result could be due to the general high heterogeneity of the field study area (Chapter 2, *section 2.1*). Unlike Kross's study in the US, Herefordshire has hedgerows surrounding most field edges. However, Benton et al. (2003) suggests the opposite would have been seen if a population based study of bird species surrounding farms was taken rather than farm-scale. Apple orchards, especially organic and cider orchards, can themselves act as bird habitat, therefore the presence of more or less woody cover surrounding each orchard will thus not be a significant variable (Mangan et al., 2017) and has even been recognised in the UK as a UK Biodiversity Action Plan Priority Habitat (Natural England, 2010).

Although it is important for predators to have access to overwintering habitat and a constant supply of prey species (Dennis et al., 1994; Smith et al., 2007), the effects of woody cover was not seen as an important feature of farm landscape on insectivorous abundance. Although this was not the focus of this study, further focus on how natural habitats surrounding, or within farm landscapes, effect bird predator and predation levels should be explored further, especially in relation to Herefordshire where difference in farm landscape features are not extensive, compared to other areas.

4.6.2 Pest control services by insectivore birds

Predation on artificial dough caterpillars from avian insectivores is significantly higher in organic orchards compared to IPM, conventional and LEAF managed farms. This result is in line with findings from Howe et al. (2009). It is known that artificial prey may underestimate true predation rates, so although these results are relative they nevertheless show a significant difference between farm types (Lövei & Ferrante, 2016). With increased insecticides used on farms, there is similar picture with the reduced amount of biological control available (Geiger et al., 2011). Many studies have assumed this, with the decrease in potential service provider it is assumed that ecosystem services are provided (Todd et al., 2011), but here this connection has been tested between the service provider and the actual biological control service provided, measured through predation rate.

Furthermore, predation on artificial dough caterpillars was higher with increases in insectivorous bird abundance and species richness, but not diversity. Insectivores can reduce pests occurring at low densities in agreement with numerous studies (Barbaro et al., 2012; Bereczki et al., 2015; ; Crowder et al., 2010; García et al., 2021; Johnson et al., 2009; Mols & Visser, 2007; Perfecto et al., 2004; Van Bael et al., 2008). In a recent paper, García et al. (2021) demonstrate that using nest boxes within apple orchards occupied by insectivorous birds (blue tits and great tits), pest abundances reduced due to predation pressure increasing, and this chapter supports these findings. Furthermore, Van Bael et al. (2008) found during a meta-analysis that the majority of bird exclosure experiment studies show an increased diversity of predatory birds correlates with increase arthropod predation. This chapter finds that species richness and abundance of insectivores are more important but Van Bael et al. (2008) study did not exclusively look at insectivore birds. Mangan et al. (2017) conversely found that birds on orchards were neither positive nor negative in relation to fruit damage or pest control to apple production.

4.6.2.1 Landscape

The presence of higher hedgerows, coppice and wooded areas was not an important factor to explain avian predation. This result reflects concluding results in a meta-analysis, where vegetation was not found to be a significant predictor in avian arthropod removal (Philpott et al., 2014), but also where no observed difference in predation was seen between simplified habitats of agroforestry systems and forests (Van Bael et al., 2008). Gunnarsson (1996) also found there to be no effect of natural vegetation on avian predation of insect and spider abundance. Similar results are recorded in tropical forests versus forest fragment studies, where lower predation rates of Lepidoptera larvae was evident in the continuous forest sites (Ruiz-Guerra et al., 2012). Finally, Lemessa, et al. (2015) found no relationship between tree cover and pest control by birds.

Contrarily, this result conflicts with other studies that found a significant decrease in predation by birds with increases in distances from forest fragments (Milligan et al., 2016). The majority of studies in Holland et al. (2017) recent literature review also conflict with result findings here on how semi-natural habitats contribute to pollination and pest control provisioning services, however the service providers in the literate review were all insect natural enemies and pollinators, not birds. Likewise, Bereczki et al. (2014) reported forests with high diversity supported higher avian pest control on herbivorous insects.

One reason the opposite is shown here, may be due to increased food sources in large woodland areas, so when woody cover increases, less insectivorous birds were recorded and less predation was seen. Vegetation change on agro-ecosystems is often used as a predictor of biodiversity loss, instead here I see that farm management, especially organic farming, is more important than on-farm or wider landscape in reducing biodiversity loss and ecosystem services. Landscape complexity may be beneficial for many ecosystem services but were not seen to be important in this study's context. This negative or neutral response to pest control from landscape complexity, seen also by Martin et al. (2013), warn of difficulties in finding a simple solution to implement biological control.

4.6.2.2 Arthropods

This study did not look at the abilities of arthropods as a pest control service on apple orchards, because no significant effect of farm management on arthropod predation rates was found, contrary to Porcel et al. (2018) and Dib et al. (2016), who find increases in arthropod diversity with organic management. However, arthropods were actually more active within 24 hours at locating and predation on both dough and plasticine caterpillars than birds were. This finding is in line with Lemessa et al. (2015) who find that arthropods had a higher predatory influence than birds in Ethiopian home-gardens. Furthermore, there are multiple trophic interactions and intra-guild predation complexities (Martin et al., 2015) at play in agricultural landscapes and the introduction of insectivorous birds may reduce other essential predators already present on the orchards (Schmitz, 2007). As arthropods had high predation rates on orchards, these complex interactions should be explored further in future studies and considered when implementing "simple" biological pest control strategies.

4.6.3 Pest moth levels

Contrary to initial predictions, pest levels did not differ significantly across four types of farm management: organic, integrated pest management, LEAF and conventional. These results are mirrored by Feber et al., (1997), who also found pest abundances were not influenced by the farming system. This result was most noticeable in regards to codling moth – one of the most economically important apple pest (Solomon et al., 1976). LEAF marque farms saw a significant reduction in fruit tree tortix moths and organic farms hosted significantly less apple ermine moths, a result mirrored by Dib et al. (2016), who found organic orchards had less aphid pests than IPM and conventional. Multiple studies have found that pests, as well as predatory insects, benefit from organic techniques (Garratt et al., 2011), however, here, adult moth pest abundance is similar across organic and non-organic farms.

A positive response was seen using specific insecticides on fruit tree totrix, which showed a significant reduction in response to those specific insecticides (Thiacloprid, Chlorantraniliprole and Indoxacarb). This is expected due to the nature of specific insecticides that reduce Lepidoptera larvae rather than broad scale insecticides such as Chlorpyrifos and Acetamiprid, which also destroy non-target species that may be beneficial predatory insects (Fountain & Harris, 2015; Wilson & Tisdell, 2000). Broad-spectrum insecticides, instead, significantly increased apple ermine. This could be due to apple ermine’s natural silk webbing defence that is associated with the ermine moth during the whole life cycle, where insecticide regimes find it hard to target the pests protected within the webbing (NACM, 2007). The broad-spectrum insecticides used to control general orchard pests has not worked for apple ermine and, with the future of farming with chemicals so unclear (Hillocks, 2012), a natural enemy pest control system may be more viable than the synthetic alternative.

Dessert apples were the only type of apple that saw a significant reduction in codling moth abundance in both the models that included (i) management and (ii) insecticide model. Although both management and insecticides are not shown to impact codling moth levels in this study, one reason the results show dessert apples hold less pests, could be the use of younger apple trees with high turnover.

Finally, pest abundance showed a mixed response to orchard size, where fruit tree tortix increased with larger orchards, apple ermine decreased with larger orchards, and no significant response from codling moth. From this study it becomes evident that pests respond differently to orchard size and insecticide use, while management, other than LEAF with fruit tree totrix, does not influence pest moth abundance. Uncertainty grows as widespread and affordable chemicals become banned (HSE, 2016) thus, more certainty and economic sense lies within implementing biological control methods into each farm management system (Cardinale et al., 2003).

García et al. (2018) found that not only moth pests decreased when insectivorous birds increased, but so too did other economically important pests to apple orchards such as aphids and apple blossom weevil. This study did not focus on multiple pests due to time limitations, but future studies that expand on this work should include a range of pests and also the damage caused by pests to understand any impact to yield. Mols & Visser (2002) found that through increasing the amount of nest boxes for insectivore birds surrounding orchards, it significantly reduced caterpillar damage to apples. This link to damage caused by pests and subsequent impacts on yield would fill the gap that is classed as an assumption in this study: that an increase in pests equates to a decrease in yield. Chapter 5 goes on to explain that this is not the case; even though adult moth pest levels are similar across farm types, insecticide use increases yield per hectare; implying either pest damage from caterpillars or other pests is greater on organic farms, or other farm management activities encourage higher yields on non-organic farms.

4.6.4 Conclusion

Pest predation levels between farm types mirror the patterns of insectivorous bird communities; higher predation rates are visible on organic farms where biodiversity is greater than other non-organic farm types. This is most likely due to the increased food availability on organic orchards increasing the insectivorous bird communities. With increased use of chemical pest control measures brings a reduction in food sources for insectivorous birds providing this regulating ecosystem service (Solomon et al., 1976; MA, 2005b).

The agro ecosystems between organic and non-organic farms are shown to be very different, where organic is shown to reduce pest levels through a natural pest control service, non-organic orchards have reduced pests using insecticides as a chemical alternative, whilst reducing biodiversity leading to less availability of a natural predation service. It is clear that a no-chemical farming supports a pest control ecosystem service here, rather than a “no or less” scenario of chemical use, suggested to increase natural enemies biodiversity and biological pest control in orchards from Dib et al. (2016). Geiger et al. (2010), suggests that for European farming to combat biodiversity loss and enhance ecosystem services, such as pest control, the shift in non-pesticide and organic farming needs to be over large areas. However, here has shown that even on small areas, organic farming with increased biodiversity, supports higher levels of insectivore bird species that in turn support a pest control ecosystem service to UK organic apple growers.

Increased biodiversity levels provide a higher “insurance policy” in relation to climate change and natural ecosystem pressures (Perfecto et al., 2004), but also in relation to the continued functioning of ecosystems and protection of crops from new and emerging pests, or pests that will inevitably become chemically resistant (Barzman et al., 2015). As birds act as a natural pest controller, due to their diets on pest insects in European farmland (Holland et al., 2006), here suggests that if farmers manage their farm landscape in line with organic principles they could increase the suitability of the landscape in order to restore biodiversity and increase the pest control ecosystem service from birds. Should farmers decide to take these steps away from costly, chemical management – laced with uncertainty – this could be the step towards a greener agricultural revolution that Conway and Tilman envisioned (1997; 2001).

4.7 REFERENCES

Agralan. (n.d.). Agralan Gorwers - The Natural Choice. Retrieved May 1, 2018, from https://www.agralan-growers.co.uk

AHDB. (2018). *Apples Best Practice Guide. Pests and Disease Control*. Retrieved from http://apples.ahdb.org.uk/ipdm.asp#

Alford, B. D. V, Carden, P. W., Dennis, E. B., Gould, H. J., & Vernon, J. D. R. (1979). Monitoring codling and tortrix moths in United Kingdom apple orchards using pheromone traps. *Annals of Applied Biology*, *91*, 165–178.

Barbaro, L., Brockerhoff, E. G., Giffard, B., & van Halder, I. (2012). Edge and area effects on avian assemblages and insectivory in fragmented native forests. *Landscape Ecology*, *27*(10), 1451–1463. http://doi.org/10.1007/s10980-012-9800-x

Barbaro, L., Giffard, B., Charbonnier, Y., van Halder, I., & Brockerhoff, E. G. (2014). Bird functional diversity enhances insectivory at forest edges: A transcontinental experiment. *Diversity and Distributions*, *20*(2), 149–159. http://doi.org/10.1111/ddi.12132

Barbaro, L., Rusch, A., Muiruri, E. W., Gravellier, B., Thiery, D., & Castagneyrol, B. (2016). Avian pest control in vineyards is driven by interactions between bird functional diversity and landscape heterogeneity. *Journal of Applied Ecology*. http://doi.org/10.1111/1365-2664.12740

Bartoń, K. (2017). *MuMIn: Multi-Model Inference. R package version 1.40.0.*

Bates, D., Maechler, M., Bolker, B., & Walker, S. (2015). Fitting Linear Mixed-Effects Models Using lme4. *Journal of Statistical Software*, *67*(1), 1–48. http://doi.org/10.18637/jss.v067.i01

Bengtsson, J., Ahnström, J., & Weibull, A. C. (2005). The effects of organic agriculture on biodiversity and abundance: A meta-analysis. *Journal of Applied Ecology*, *42*, 261–269. http://doi.org/10.1111/j.1365-2664.2005.01005.x

Benton, T. G., Vickery, J. a., & Wilson, J. D. (2003). Farmland biodiversity: Is habitat heterogeneity the key? *Trends in Ecology and Evolution*, *18*(4), 182–188. http://doi.org/10.1016/S0169-5347(03)00011-9

Benton, T. I. M. G., Bryant, D. M., Cole, L., & Crick, H. Q. P. (2002). Linking agricultural practice to insect and bird populations: a historical study over three decades. *Journal of Applied Ecology*, *39*, 673–687.

Bereczki, K., Hajdu, K., & Báldi, A. (2015). Effects of forest edege on pest control service provided by birds in fragmented temperate forests. *Acta Zoologica Academiae*

Scientiarum Hungaricae, *61*(3), 289–304. http://doi.org/10.17109/AZH.61.3.7.2015

Bereczki, K., Ódor, P., Csóka, G., Mag, Z., & Báldi, A. (2014). Effects of forest heterogeneity on the efficiency of caterpillar control service provided by birds in temperate oak forests. *Forest Ecology and Management*, (327), 96–105. http://doi.org/10.1016/j.foreco.2014.05.001

Bolker, B. M., Brooks, M. E., Clark, C. J., Geange, S. W., Poulsen, J. R., Stevens, M. H. H., & White, J. S. S. (2009). Generalized linear mixed models: a practical guide for ecology and evolution. *Trends in Ecology and Evolution*, *24*(3), 127–135. http://doi.org/10.1016/j.tree.2008.10.008

Bommarco, R., Kleijn, D., & Potts, S. G. (2013). Ecological intensification: harnessing ecosystem services for food security. *Trends in Ecology & Evolution*, *28*(4), 230–8. http://doi.org/10.1016/j.tree.2012.10.012

Boyle, M. (2012). *Quantifying Predation Pressure Along a Gradient of Land-Use Intensity in Saba , Borneo*. Imperial College London.

Burnham, K. P., & Anderson, D. R. (2002). *Model selection and multimodel inference: a practical information-theoretic approach* (2nd editio). New York: Springer.

Butler, S. J., Vickery, J. A., & Norris, K. (2007). Farmland Biodiversity and the Footprint of Agriculture. *Science*, *315*(January), 381–384.

Carden, P. W. (1987). Supervised control of apple pests in southern England. *Crop Protection*, *6*(4), 234–243.

Cardinale, B. J., Harvey, C. T., Gross, K., & Ives, A. R. (2003). Biodiversity and biocontrol: emergent impacts of a multi-enemy assemblage on pest suppression and crop yield in an agroecosystem. *Ecology Letters*, *6*(9), 857–865. http://doi.org/10.1046/j.1461-0248.2003.00508.x

Carson, R. (1962). *Silent Spring*. New York: Houghton Mifflin.

Chaplin-Kramer, R., de Valpine, P., Mills, N. J., & Kremen, C. (2013). Detecting pest control services across spatial and temporal scales. *Agriculture, Ecosystems and Environment*, *181*(2013), 206–212. http://doi.org/10.1016/j.agee.2013.10.007

Classen, A., Peters, M. K., Ferger, S. W., Helbig-bonitz, M., Schmack, J. M., Maassen, G., … Bo, K. (2014). Complementary ecosystem services provided by pest predators and pollinators increase quantity and quality of coffee yields. *Proc R Soc B*, *281*.

Clough, Y., Barkmann, J., Juhrbandt, J., Kessler, M., Wanger, T. C., Anshary, A., … Tscharntke, T. (2011). Combining high biodiversity with high yields in tropical agroforests. *Proceedings of the National Academy of Sciences of the United States of America*, *108*(20), 8311–6. http://doi.org/10.1073/pnas.1016799108

Commission, E. (2018). What is organic Farming? Retrieved July 4, 2018, from https://ec.europa.eu/agriculture/organic/organic-farming/what-is-organic-farming_en

Conway, G. (1997). *The Doubly Green Revolution: Food for All in the Twenty-first Century*. Penguin.

Crawley. (2007). *The R Book*. Chichester: John Wiley & Sons Ltd. http://doi.org/10.15713/ins.mmj.3

Crawley, M. J. (2002). *Statistical Computing: An Introduction to Data Analysis Using S-PLUS*. Chichester: John Wiley & Sons Ltd.

Crawley, M. J. (2013). *The R Book* (Second Edi). Chichester: John Wiley & Sons Ltd.

Crowder, D. W., Northfield, T. D., Strand, M. R., & Snyder, W. E. (2010). organic agriculture promotes evenness and natural pest control. *Nature*, *466*(7302), 109–12. http://doi.org/10.1038/nature09183

Cunningham, S. A., Attwood, S. J., Bawa, K. S., Benton, T. G., Broadhurst, L. M., Didham, R. K., … Lindenmayer, D. B. (2013). To close the yield-gap while saving biodiversity will require multiple locally relevant strategies. *Agriculture, Ecosystems and Environment*, *173*, 20–27. http://doi.org/10.1016/j.agee.2013.04.007

Daniels, S., Witters, N., Beliën, T., Vrancken, K., Vangronsveld, J., & Van Passel, S. (2017). Monetary Valuation of Natural Predators for Biological Pest Control in Pear Production. *Ecological Economics*, *134*, 160–173. http://doi.org/10.1016/j.ecolecon.2016.12.029

Dennis, P., Thomas, M. B., & Sotherton, N. W. (1994). Structural features of field boundaries which influence the overwintering densities of beneficial arthropod predators. *Journal of Applied Ecology*, *31*(2), 361–370.

Dib, H., Sauphanor, B., & Capowiez, Y. (2016). Effect of management strategies on arthropod communities in the colonies of rosy apple aphid, Dysaphis plantaginea Passerini (Hemiptera: Aphididae) in south-eastern France. *Agriculture, Ecosystems and Environment*, *216*(2016), 203–206. http://doi.org/10.1016/j.agee.2015.10.003

Dirzo, R., & Raven, P. H. (2003). Global State of Biodiversity loss. *Annu. Rev. Environ. Resour.*, *28*, 137–167. http://doi.org/10.1146/annurev.energy.28.050302.105532

Donald, P. F., Green, R. E., & Heath, M. F. (2001). Agricultural intensification and the collapse of Europe's farmland bird populations. *Proceedings. Biological Sciences / The Royal Society*, *268*(1462), 25–29. http://doi.org/10.1098/rspb.2000.1325

EC. (2017). Integrated Pest Management.

Feber, R. E., Firbank, L. G., Johnson, P. J., & Macdonald, D. W. (1997). The effects of organic farming on pest and non-pest butterfly abundance. *Agriculture, Ecosystems & Environment*, *64*.

Ferrante, M., Cacciato, L. O., & Lövei, G. L. (2014). Quantifying predation pressure along an urbanisation gradient in denmark using artificial caterpillars. *European Journal of Entomology*, *111*(5), 1–6. http://doi.org/10.14411/eje.2014.082

Foley, J. A., DeFries, R., Asner, G. P., Barford, C., Bonan, G., Carpenter, S. R., … Snyder, P. K. (2005). Global Consequences of Land Use. *Science*, *309*(5734), 570–574. http://doi.org/10.1126/science.1111772

Fountain, M. T., & Harris, A. L. (2015). Non-target consequences of insecticides used in apple and pear orchards on Forficula auricularia L . (Dermaptera: Forficulidae). *Biological Control*, *91*(2015), 27–33. http://doi.org/10.1016/j.biocontrol.2015.07.007

García, D., Miñarro, M., & Martínez-sastre, R. (2018). Birds as suppliers of pest control in cider apple orchards: Avian biodiversity drivers and insectivory effect. *Agriculture, Ecosystems and Environment*, *254*(December), 233–243. http://doi.org/10.1016/j.agee.2017.11.034

García, D., Miñarro, M., & Martínez-Sastre, R. (2021). Enhancing ecosystem services in apple orchards: Nest boxes increase pest control by insectivorous birds. Journal of Applied Ecology, 58(3), 465–475. https://doi.org/10.1111/1365-2664.13823

Garfinkel, M., & Johnson, M. (2015). Pest-removal services provided by birds on small organic farms in northern California. *Agriculture, Ecosystems & Environment*, *211*(2015), 24–31. http://doi.org/10.1016/j.agee.2015.04.023

Garratt, M. P. D., Breeze, T. D., Jenner, N., Polce, C., Biesmeijer, J. C., & Potts, S. G. (2014). Avoiding a bad apple: Insect pollination enhances fruit quality and economic value. *Agriculture, Ecosystems & Environment*, *184*, 34–40. http://doi.org/10.1016/j.agee.2013.10.032

Garratt, M. P. D., Wright, D. J., & Leather, S. R. (2011). The effects of farming system and fertilisers on pests and natural enemies: A synthesis of current research. *Agriculture, Ecosystems and Environment*, *141*(3–4), 261–270. http://doi.org/10.1016/j.agee.2011.03.014

Geiger, F., de Snoo, G. R., Berendse, F., Guerrero, I., Morales, M. B., Oñate, J. J., … Tscharntke, T. (2010). Landscape composition influences farm management effects on farmland birds in winter: A pan-European approach. *Agriculture, Ecosystems & Environment*, *139*(4), 571–577. http://doi.org/10.1016/j.agee.2010.09.018

Godfray, H. C. J., Beddington, J. R., Crute, I. R., Haddad, L., Lawrence, D., Muir, J. F., … Toulmin, C. (2010). Food Security: The challenge of Feeding 9 Billion people. *Science*, *327*(812). http://doi.org/10.1126/science.1185383

González-Gómez, P. ., Estades, C. ., & Simonetti, J. . (2006). Strengthened insectivory in a temperate fragmented forest. *Oecologia*, (148), 137–143.

Gove AD, Hylander K, Nemomissa S, S. A. (2008). Ethiopian coffee cultivation - Implications for bird conservation and environmental certification. *Conservation Letters*, *1*, 208–216.

Green, R. E., Cornell, S. J., Scharlemann, J. P. W., & Balmford, A. (2005). Farming and the Fate of Wild Nature. *Science*, *550*(307). http://doi.org/10.1126/science.1106049

Gunnarsson, B. (1996). Bird Predation and Vegetation Structure Affecting Spruce-Living Arthropods in a Temperate Forest. *The Journal of Animal Ecology*, *65*(3), 389. http://doi.org/10.2307/5885

Hardman, C. J., Harrison, D. P. G., Shaw, P. J., Nevard, T. D., Potts, S. G., & Norris, K. (2016). Supporting local diversity of habitats and species on farmland : a comparison of three wildlife-friendly schemes. *Journal of Applied Ecology*, *53*, 171–180. http://doi.org/10.1111/1365-2664.12557

Harrison, X. A., Donaldson, L., Correa-cano, M. E., Evans, J., Goodwin, C., Robinson, B., … Inger, R. (2017). Best practice in mixed effects modelling and multi-model inference in ecology. *PeerJ*.

Hillocks, R. J. (2012). Farming with fewer pesticides: EU pesticide review and resulting challenges for UK agriculture. *Crop Protection*, *31*. http://doi.org/10.1016/j.cropro.2011.08.008

Hole, D. G., Perkins, a. J., Wilson, J. D., Alexander, I. H., Grice, P. V., & Evans, a. D. (2005). Does organic farming benefit biodiversity? *Biological Conservation*, *122*(1), 113–130. http://doi.org/10.1016/j.biocon.2004.07.018

Holland, J. M., Douma, J. C., Crowley, L., James, L., Kor, L., Stevenson, D. R. W., & Smith, B. M. (2017). Semi-natural habitats support biological control , pollination and soil conservation in Europe . A review. *Agronomy for Sustainable Development*, *37*(31). http://doi.org/10.1007/s13593-017-0434-x

Holland, J. M., Hutchison, M. A. S., Smith, B., & Aebischer, N. J. (2006). A review of invertebrates and seed-bearing plants as food for farmland birds in Europe. *Annals of Applied Biology*, *148*, 49–71. http://doi.org/10.1111/j.1744-7348.2006.00039.x

Howe, A., Lövei, G. L., & Nachman, G. (2009). Dummy caterpillars as a simple method to assess predation rates on invertebrates in a tropical agroecosystem. *Entomologia Experimentalis et Applicata*, *131*, 325–329. http://doi.org/10.1111/j.1570-7458.2009.00860.x

HSE. (2016). Changes to authorisations for products containing chlorpyrifos. Retrieved November 16, 2017, from http://www.hse.gov.uk/pesticides/news/information-update-0316.htm

Jenkins, M., Scherr, S. J., & Inbar, M. (2004). Markets for Biodiversity Services: Potential roles and challenges. *Environment*, *46*(6), 32–42.

Johnson, M. D., Levy, N. J., Kellermann, J. L., & Robinson, D. E. (2009). Effects of shade and bird exclusion on arthropods and leaf damage on coffee farms in Jamaica's Blue Mountains. *Agroforestry Systems*, *76*(1), 139–148. http://doi.org/10.1007/s10457-008-9198-2

Jonsson, M., Kaartinen, R., & Straub, C. S. (2017). Relationships between natural enemy diversity and biological control. *Current Opinion in Insect Science*, *20*, 1–6. http://doi.org/10.1016/j.cois.2017.01.001

Kalka, M. B., Smith, A. R., & Kalko, E. K. V. (2008). Bats Limit Arthropods and Herbivory in a Tropical Forest. *Science*, *320*(5872), 71.

Kamil, B. (2017). Multi-Model Inference.

Katayama, N. (2016). Bird diversity and abundance in organic and conventional apple orchards in northern Japan. *Nature*, *6*(September), 1–7. http://doi.org/10.1038/srep34210

Kirk, D. a, Evenden, M. D., & Mineau, P. (1996). Past and current attempts to evaluate the role of birds as predators of insect pests in temperate agriculture. *Current Ornithology*, *13*, 175–269.

Krebs, J. R., Wilson, J. D., Bradbury, R. B., & Siriwardena, G. M. (1999). The second Silent Spring? *Nature*, *400*, 611–612. http://doi.org/10.1038/23127

Kronenberg, J. (2014). What can the current debate on ecosystem services learn from the past? Lessons from economic ornithology. *Geoforum*, *55*(2014), 164–177. http://doi.org/10.1016/j.geoforum.2014.06.011

Kross, S. M., Kelsey, T. R., Mccoll, C. J., & Townsend, J. M. (2016). Field-scale habitat complexity enhances avian conservation and avian-mediated pest-control services in an intensive agricultural crop. *Agriculture, Ecosystems and Environment*, *225*(2016), 140–149. http://doi.org/10.1016/j.agee.2016.03.043

Laure, M., Gaëlle, M., Simon, S., Rault, M., & Yvan, C. (2014). Management strategies in apple orchards influence earwig community. *Chemosphere*, *124*, 156–162. http://doi.org/10.1016/j.chemosphere.2014.12.024

Lemessa, D., Hambäck, P. A., & Hylander, K. (2015). Arthropod but Not Bird Predation in Ethiopian Homegardens Is Higher in Tree-Poor than in Tree-Rich Landscapes. *PloS One*, *10*(5), 1–12. http://doi.org/10.1371/journal.pone.0126639

Letourneau, D. K., Ando, A. W., Jedlicka, J. A., Narwani, A., & Barbier, E. (2015). Simple-but-sound methods for estimating the value of changes in biodiversity for biological pest control in agriculture. *Ecological Economics*, *120*, 215–225.

http://doi.org/10.1016/j.ecolecon.2015.10.015

Loiselle, B. A., & Farji-Brener, A. G. (2002). What's Up? An Experimental Comparison of Predation Levels between Canopy and Understory in a Tropical Wet Forest. *Biotropica*, *34*(2), 327–330.

Low, P. A., Sam, K., McArthur, C., Posa, M. A. C., Hochuli, D. F. (2014). Determining predator identify from attach marks left in model caterpillars: guidelines for best practice. *Entomologia Experimentalis et Applicata 152: 120–126*. http:// doi: 10.1111/eea.12207

Lövei, G. L., & Ferrante, M. (2016). A review of the sentinel prey method as a way of quantifying invertebrate predation under field conditions. *Insect Science*, 1–40. http://doi.org/10.1111/1744-7917.12405

Maas, B., Clough, Y., & Tscharntke, T. (2013). Bats and birds increase crop yield in tropical agroforestry landscapes. *Ecology Letters*, *16*(12), 1480–1487. http://doi.org/10.1111/ele.12194

Mace, G. M., Norris, K., & Fitter, A. H. (2012). Biodiversity and ecosystem services: a multilayered relationship. *Trends in Ecology & Evolution*, *27*(1), 19–26. http://doi.org/10.1016/j.tree.2011.08.006

Mangan, A. M., Pejchar, L., & Werner, S. J. (2017). Bird use of organic apple orchards: Frugivory, pest control and implications for production. *PloS One*, *12*(9), 1–15. http://doi.org/10.1371/journal.pone.0183405

Mäntylä, E., Alessio, G. A., Blande, J. D., Heijari, J., Holopainen, J. K., Piirtola, P., & Klemola, T. (2008). From Plants to Birds: Higher Avian Predation Rates in Trees Responding to Insect Herbivory. *Plos One*, *3*(7), 1–8. http://doi.org/10.1371/journal.pone.0002832

Mäntylä, E., Klemola, T., & Laaksonen, T. (2011). Birds help plants: A meta-analysis of top-down trophic cascades caused by avian predators. *Oecologia*, *165*(1), 143–151. http://doi.org/10.1007/s00442-010-1774-2

Marliac, G., Penvern, S., Barbier, J., Penvern, S., Barbier, J., & Capowiez, Y. (2015). Impact of crop protection strategies on natural enemies in organic apple production. *Agronomy for Sustainable Development*, *35*(2), 803–813. http://doi.org/10.1007/s13593-015-0282-5

Martin, E. A., Reineking, B., Seo, B., & Steffan-Dewenter, I. (2015). Pest control of aphids depends on landscape complexity and natural enemy interactions. *PeerJ*, *3*. http://doi.org/10.7717/peerj.1095

Martin, T. E. (1987). Artificial nest experiments: effects of nest appearance and type of predator. *The Condor*, *89*, 925–928.

Mazerolle, M. J. (2017). AICcmodavg: Model selection and multimodel inference based on (Q)AICI. Retrieved from https://cran.r-project.org/package=AICcmodavg.

Millenium Ecosystem Assessment. (2005a). *Ecosystems and Human Well-Being: Opportunities and challenges for Business and Industry*. Washington, DC.

Millenium Ecosystem Assessment. (2005b). *Millenium Ecosysetm Assesment - Biodiversity Regulation of Ecosystem Services*. Washington, DC.

Milligan, M. C., Johnson, M. D., Garfinkel, M., Smith, C. J., & Njoroge, P. (2016). Quantifying pest control services by birds and ants in Kenyan coffee farms. *Biological Conservation*, *194*(2016), 58–65. http://doi.org/10.1016/j.biocon.2015.11.028

Mols, C. M. M., & Visser, M. E. (2002). Great tits can reduce caterpillar damage in apple orchards, 888–899.

Mooney, K. A., Gruner, D. S., Barber, N. A., Van Bael, S. A., Philpott, S. M., & Greenberg, R. (2010). Interactions among predators and the cascading effects of vertebrate insectivores on arthropod communities and plants. *PNAS*, *107*(16), 7335–7340. http://doi.org/10.1073/pnas.1001934107

NACM. (2007). *Growers Update: Apple Ermine Moth*. Hereford.

Natural England. (2010). *Traditional orchards: orchards and wildlife*. *Technical Information Note* (Vol. TIN020).

Oksanen, J., Blanchet, F. G., Friendly, M., Kindt, R., Legendre, P., McGlinn, D., … Wagner, H. (2017). *vegan: Community Ecology Pack. R package version 2.4-5*. Retrieved from https://cran.r-project.org/package=vegan

Peisley, R. K., Saunders, M. E., & Luck, G. W. (2016). Cost-benefit trade-offs of bird activity in apple orchards. *PeerJ*, 1–20. http://doi.org/10.7717/peerj.2179

Perfecto, I., Vandermeer, J. H., Bautista, G. L., Nuñez, G. I., Greenberg, R., Bichier, P., & Langridge, S. (2004). Greater predation in shaded coffee farms: The role of resident neotropical birds. *Ecology*, *85*(10), 2677–2681. http://doi.org/10.1890/03-3145

Philpott, S. M., Soong, O., Lowenstein, J. H., Pulido, A. L., Lopez, D. T., Flynn, D. F. B., & DeClerck, F. (2014). Functional richness and ecosystem services: Functional bird predation in tropical on arthropods agroecosystems. *Ecological Applications*, *19*(7), 1858–1867. http://doi.org/10.1890/08-1928.1

Porcel, M., Andersson, G. K. S., Pålsson, J., & Tasin, M. (2018). organic management in apple orchards: higher impacts on biological control than on pollination. *Journal of Applied Ecology*. http://doi.org/10.1111/1365-2664.13247

Potts, S. G., Biesmeijer, J. C., Kremen, C., Neumann, P., Schweiger, O., & Kunin, W. E. (2010). Global pollinator declines: trends, impacts and drivers. *Trends in Ecology & Evolution*, *25*(6), 345–53. http://doi.org/10.1016/j.tree.2010.01.007

Puech, C., Baudry, J., Joannon, A., Poggi, S., & Aviron, S. (2014). organic vs. conventional farming dichotomy: Does it make sense for natural enemies? *Agriculture, Ecosystems*

& Environment, *194*, 48–57. http://doi.org/10.1016/j.agee.2014.05.002

QGIS Development Team. (2015). QGIS Geographic Information System. Open Source Geospatial Foundation Project. Retrieved from http://qgis.osgeo.org

Quinn, J. E., Brandle, J. R., & Johnson, R. J. (2012). The effects of land sparing and wildlife-friendly practices on grassland bird abundance within organic farmlands. *Agriculture, Ecosystems & Environment*, *161*, 10–16. http://doi.org/10.1016/j.agee.2012.07.021

R Core Team. (2015). R: A Language and Environment for Statistical Computing. Vienna, Austria: R Foundation for Statistical Computing. Retrieved from https://www.r-project.org/

Reddy, G. V. P., & Guerrero, A. (2001). Optimum timing of insecticide applications against diamondback moth Plutella xylostella in cole crops using threshold catches in sex pheromone traps. *Pest Management Science*, *57*(1), 90–94. http://doi.org/10.1002/1526-4998(200101)57:1<90::AID-PS258>3.0.CO;2-N

Richards, S. A., Whittingham, M. J., & Stephens, P. A. (2011). Model selection and model averaging in behavioural ecology: the utility of the IT-AIC framework. *Behavioural Ecology and Sociobiology*, *65*, 77–89. http://doi.org/10.1007/s00265-010-1035-8

Riedl, H., & Croft, B. A. (1974). A study of pheromone trap catches in relation to codling moth (Lepidoptera: Olethreutidae) damage. *The Canadian Entomologist*, (106), 525–537.

Royal Society. (2009). *Reaping the Benefits: Science and the sustainable intensification of global agriculture*. London. Retrieved from http://www.asbp.org.uk/uploads/documents/resources/Andrew Smith - CERAM ASBP 27th Feb.pdf

Ruiz-Guerra, B., Renton, K., & Dirzo, R. (2012). Consequences of Fragmentation of Tropical Moist Forest for Birds and Their Role in Predation of Herbivorous Insects. *Biotropica*, *44*(2), 228–236.

Safonkin, A. F., & Triseleva, T. A. (2005). Fecundity of Large Fruit-tree Tortrix Archips podana Scop. (Lepidoptera: Tortricidae) on Different Host Plants. *Biology Bulletin Bulletin*, *32*(2), 207–210.

Samnegård, U., Alins, G., Boreux, V., Bosch, J., García, D., Happe, A. K., … Hambäck, P. A. (2019). Management trade-offs on ecosystem services in apple orchards across Europe: Direct and indirect effects of organic production. *Journal of Applied Ecology, 56*(4), 802–811. https://doi.org/10.1111/1365-2664.13292

Sam, K., Remmel, T., & Molleman, F. (2015). Material affects attack rates on dummy caterpillars in tropical forest where arthropod predators dominate: an experiment using clay and dough dummies with green colourants on various plant species.

Entomologia Experimentalis et Applicata, 1–8. http://doi.org/10.1111/eea.12367

Schmitz, O. J. (2007). Predator Diversity and Trophic Interactions. *Ecology*, *88*(10), 2415–2426.

Shave, M. E., Shwiff, S. A., Elser, J. L., & Lindell, C. A. (2018). Falcons using orchard nest boxes reduce fruit-eating bird abundances and provide economic benefits for a fruit-growing region. *Journal of Applied Ecology*, (January), 1–10. http://doi.org/10.1111/1365-2664.13172

SIR. (2018). The Pests: Codling Moth Life Cycle. Retrieved May 31, 2018, from https://www.oksir.org/the-pests/codling-moth/lifecycle/

Smith, J., Potts, S. G., Woodcock, B. a., & Eggleton, P. (2007). Can arable field margins be managed to enhance their biodiversity, conservation and functional value for soil macrofauna? *Journal of Applied Ecology*, *45*(1), 269–278. http://doi.org/10.1111/j.1365-2664.2007.01433.x

Solomon, M., Glen, D., Kendall, D., & Milsom, N. (1976). Predation of overwintering larvae of codling moth (Cydia pomonella (L.)) by birds. *Journal of Applied Ecology*, *13*(2), 341–352. Retrieved from http://www.jstor.org/stable/10.2307/2401784

Tews, J., Brose, U., Grimm, V., Tielbörger, K., Wichmann, M. C., Schwager, M., & Jeltsch, F. (2004). Animal Species Diversity Driven by Habitat Heterogeneity / Diversity : The Importance of Keystone. *Journal of Biogeography*, *31*(1), 79–92. http://doi.org/10.1046/j.0305-0270.2003.00994.x

Thomas, L., Buckland, S. T., Rexstad, E. a., Laake, J. L., Strindberg, S., Hedley, S. L., … Burnham, K. P. (2010). Distance software: Design and analysis of distance sampling surveys for estimating population size. *Journal of Applied Ecology*, *47*(1), 5–14. http://doi.org/10.1111/j.1365-2664.2009.01737.x

Tilman, D., Balzer, C., Hill, J., & Befort, B. L. (2011). Global food demand and the sustainable intensification of agriculture. *PNAS*, *108*(50). http://doi.org/10.1073/pnas.1116437108

Tilman, D., Fargione, J., Wolff, B., D'Antonio, C., Dobson, a, Howarth, R., … Swackhamer, D. (2001). Forecasting agriculturally driven global environmental change. *Science (New York, N.Y.)*, *292*(5515), 281–4. http://doi.org/10.1126/science.1057544

Tscharntke, T., Sekercioglu, C. H., Dietsch, T. V., Sodhi, N. S., Hoehn, P., & Tylianakis, J. M. (2008). Landscape Contraints on Functional Diversity of BIrds and Insects in Tropical Agroecosystems. *Ecology*, *89*(4), 944–952.

UK Moths. (2018). UK Moths. Retrieved June 11, 2018, from http://ukmoths.org.uk

Van Bael, S. A., Philpott, S. M., Greenberg, R., Bichier, P., Barber, N. A., Mooney, K. A., & Gruner, D. S. (2008). Birds as predators in tropical agroforestry systems. *Ecology*, *89*(4), 928–934. http://doi.org/10.1890/06-1976.1

Venables, W. N., & Ripley, B. D. (2002). *Modern Applied Statistics with S.* (Fourth). New York: Springer. Retrieved from http://www.stats.ox.ac.uk/pub/MASS4

Wilson, C., & Tisdell, C. (2000). *Why farmers continue to use pesticides despite environmental, health and sustainability costs* (No. 53). *Economics, Ecology and the Environment*. Brisbane, Australia.

Wilson, J. D., Morris, A. J., Arroyo, B. E., Clark, S. C., & Bradbury, R. B. (1999). A review of the abundance and diversity of invertebrate and plant foods of granivorous birds in northern Europe in relation to agricultural change. *Agriculture, Ecosystems & Environment*, *75*(1–2), 13–30. http://doi.org/10.1016/S0167-8809(99)00064-X

Appendix H
Apple moth pest descriptions

Codling Moth (*Cydia pomonella)* – Common throughout Britain and a key pest to apples, their direct attack on fruits occurs through larval habits of burrowing into fruits (AHDB, 2018). The caterpillars feed inside apples (Malus), quince (Cydonia oblonga), pear (Pyrus) and other wild and cultivated fruit, causing visible damage to fruit. Adult Codling Moth occur mainly in July and August, often with a second generation in September and October (Moths UK, 2018).

Large Fruit Tree Tortrix (*Archips podana)* – The larvae feed on foliage, flowers and fruit of deciduous trees, including apple (UK Moths, 2018). However, primarily they are known to use apple trees as their preferred host plant (Safonkin & Triseleva, 2005). Defoliation of apple trees causes reduction of apple yields. Main flight period is June – July, but second generation may arise into September (UK Moths, 2018).

Apple Ermine (*Yponomeuta malinellus*) – Ermine larvae feed in “blotch” type mines on apple (*Malus*) leaves, and then feed in a silken web in May and early June, covering entire branches in severe cases (UK Moths, 2018; NACM, 2007). Defoliation caused by leaf mining web-covering causes apple yield to reduce. The adult flight period is July and August (UK Moths, 2018).

Light brown Apple Moth (*Epiphyas postvittana)* – Introduced in 1930’s, this species has spread, from south west of England northwards, and is now regular in many parts of the UK – it is polyphagous and should be considered when identifying larvae on any plant (UK Moths, 2018).

Summer Fruit Tortrix (*Adoxophyes orana)* – This moth is a recent arrival in this country and is a pest particularly in apple orchards. The larvae feed on a variety of fruit, especially apples. The moth is bivoltine, appearing in June and August-September (UK Moths, 2018).

Winter moth (*Operophtera brumata)* – Common in most of Britain, this moth occurs late autumn through to January or February. The larvae feed on a range of trees and shrubs, as well as moorland species such as heather (Calluna). Sometimes the larvae occur in great numbers, reaching pest status and occasionally completely defoliating small trees (UK Moths, 2018).

Appendix I

Insectivore bird GLMM 'Full' model average summary

Generalised linear mixed effects full model average summary examining insectivorous bird community metrics as the response variable (abundance, species richness and Shannon diversity) across two years 2015 and 2016 in apple orchards of the study area. Investigation of the response variables in relation to farm management, observer, apple type, Woody Cover, day number and bird box presence. The intercept is conventional farm management with CP as the observer. Values in bold are significant (P=<0.05). VIF is the variance inflation factor value derived from squared GVIF^(1/(2*Df)), where any variable above 5 was considered correlated and removed from the global model before dredging.

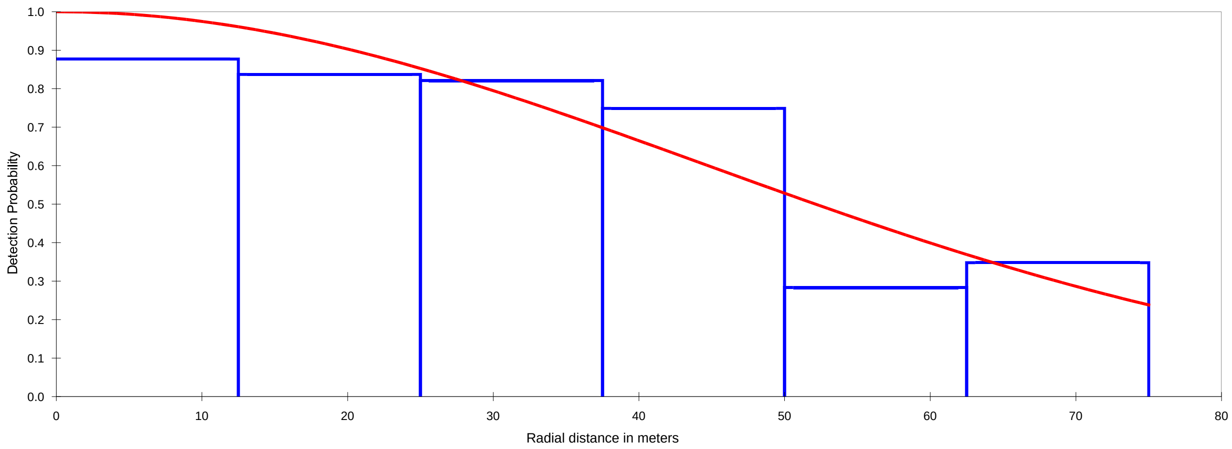
		Insectivore Abundance				**Insectivore Species Richness**				**Insectivore Diversity**			
Fixed Effects		Value	Standard Error	*P*-value	*VIF*	Value	Standard Error	*P*-value	*VIF*	Value	Standard Error	*P*-value	*VIF*
Farm Management	Conventional (Intercept)	1.895	0.064	**<0.001**	1.392	1.563	0.076	**<0.001**	1.44	1.356	0.084	**<0.001**	1.376
	IPM	0.109	0.071	0.125	1.392	0.054	0.063	0.389	1.44	0.076	0.067	0.257	1.376
	LEAF	0.181	0.080	0.025	1.392	0.081	0.078	0.298	1.44	0.098	0.079	0.217	1.376
	Organic	0.633	0.057	**<0.001**	1.392	0.417	0.060	**<0.001**	1.44	0.445	0.064	**<0.001**	1.376
Observer	PL	0.005	0.033	0.871	1.318	-0.161	0.070	0.022	1.182	-0.199	0.078	0.011	1.153
	SF	0.01	0.035	0.786	1.318	0.034	0.053	0.525	1.182	0.090	0.060	0.137	1.153
Apple types	Dessert	-0.006	0.036	0.859	1.893	-0.008	0.033	0.820	2.155	-0.001	0.011	0.944	1.871
	Juice	0.001	0.032	0.977	1.666	-0.038	0.053	0.474	1.685	-0.007	0.028	0.808	1.621
	Cider	-0.010	0.049	0.831	1.772	-0.063	0.086	0.469	1.568	-0.015	0.051	0.768	1.638
Environmental/ Landscape impacts	Day number	0.186	0.053	**<0.001**	1.553	-0.002	0.02	0.907	1.051	-0.003	0.015	0.864	1.044
	Woody Cover	-0.244	0.07	**<0.001**	2.164	-0.105	0.079	0.185	2.484	-0.099	0.090	0.273	2.344
	Temperature	-0.007	0.031	0.824	1.713	-	-	-	-	-	-	-	-

Appendix J

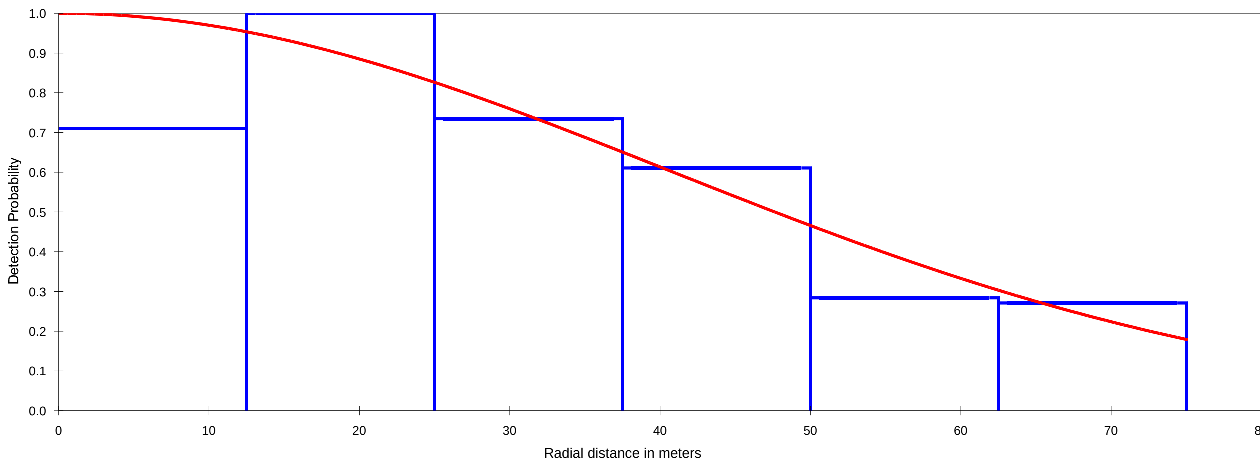
Insectivore density detection plots

Insectivore detection probability plots using the model with lowest AIC (15319.55, no of parameters 12). The model used half normal and simple polynomial distribution and constructed in Distance version 7.0.

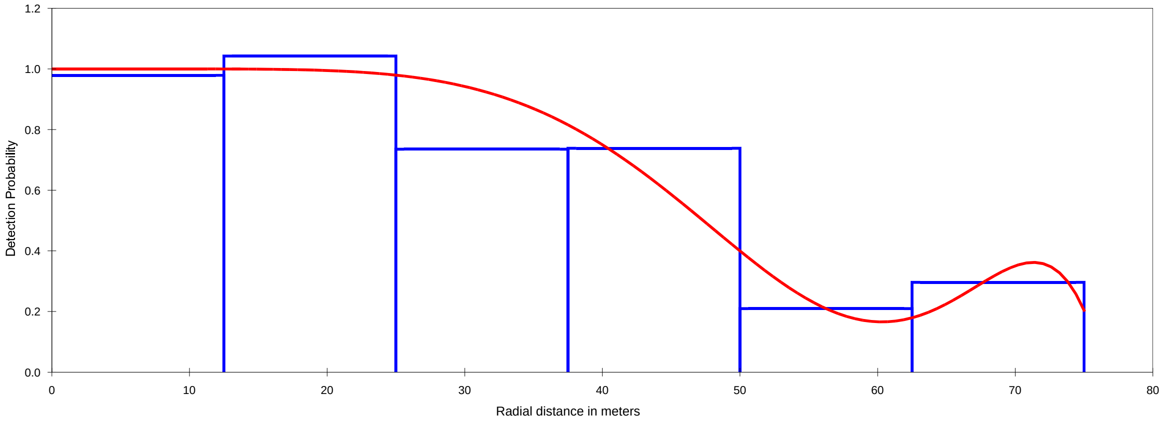
Conventional detection probability plot



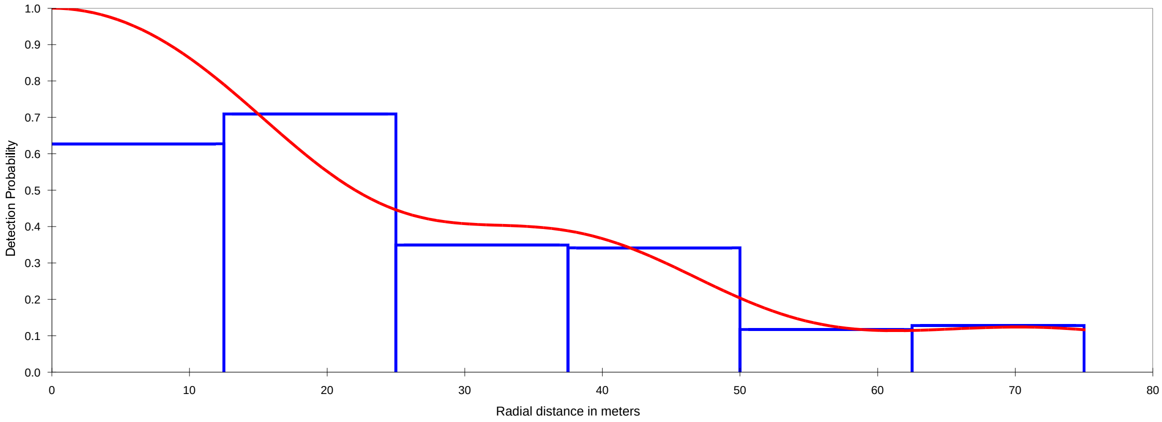
IPM detection probability plot



LEAF detection probability plot



Organic detection probability plot



Appendix K

Prey predation probability GLMMs ‘full’ model average summaries

Generalised linear mixed effects full model average summary examining dough prey predation probabilities as the response variable. Investigation of the response variables was separated into four models, to analyse fixed effect variables of: farm management, insectivore abundance, insectivore species richness and insectivore diversity separately. Each glmm included fixed effect variables: apple type, orchard size and day number. The intercept is conventional farm management with CP as the observer in farm management model. Values in bold are significant ($P=<0.05$). VIF is the variance inflation factor value derived from squared GVIF^(1/(2*Df)), where any variable above 5 was considered correlated and removed from the global model before dredging.

		Prey predation and farm management				**Prey predation and insectivore abundance**				**Prey predation and insectivore species richness**				**Prey predation and insectivore diversity**			
Fixed Effects		Value	Standard Error	*P*-value	*VIF*	Value	Standard Error	*P*-value	*VIF*	Value	Standard Error	*P*-value	*VIF*	Value	Standard Error	*P*-value	*VIF*
Farm Management	Conventional (Intercept)	-3.56	0.634	**<0.001**	2.076	-	-	-	-	-	-	-	-	-	-	-	-
	IPM	-1.204	0.941	0.205	2.076	-	-	-	-	-	-	-	-	-	-	-	-
	LEAF	-2.594	1.293	0.046	2.076	-	-	-	-	-	-	-	-	-	-	-	-
	Organic	2.786	0.603	**<0.001**	2.076	-	-	-	-	-	-	-	-	-	-	-	-
Insectivore community metrics	Intercept	-	-	-	-	-4.399	0.495	**<0.001**	-	-4.384	0.495	**<0.001**		-4.498	0.514	**<0.001**	-
	Mean insectivore abundance	-	-	-	-	1.193	0.387	**0.002**	2.283	-	-	-	-	-	-	-	-

	Mean insectivore species richness	-	-	-	-	-	-	-	-	1.472	0.413	**<0.001**	1.927	-	-	-	-
	Mean insectivore diversity	-	-	-	-	-	-	-	-	-	-	-	-	0.429	0.402	0.288	1.385
Apple types	Dessert	2.012	1.132	0.077	2.335	0.372	0.736	0.616	1.777	0.326	0.696	0.641	2.238	0.454	0.805	0.575	2.247
	Juice	1.449	0.814	0.077	1.484	1.71	0.936	0.069	1.442	1.846	0.901	**0.042**	1.369	1.86	0.973	0.058	1.353
	Cider	0.363	0.935	0.7	1.322	2.066	1.764	0.244	1.628	2.153	1.753	0.222	1.977	2.913	1.791	0.106	2.045
Environmental/ Landscape impacts	Day number	-0.229	0.329	0.488	1.685	-0.028	0.35	0.935	1.45	-0.039	0.364	0.916	1.008	-0.005	0.378	0.989	1.006
	Orchard size	0.215	0.393	0.587	4.871	-1.298	0.885	0.144	3.752	-1.197	0.848	0.160	3.222	-2.595	0.732	**<0.001**	2.515

Appendix L

Codling moth pest abundance GLMMs 'full' model average summaries

Generalised linear mixed effects full model average summary examining codling moth abundances as the response variable. Investigation of the response variables was separated into two models, to analyse fixed effect variables of: farm management and insecticides application separately. Each glmm included fixed effect variables: apple type (cooking, cider and dessert – juice was classed as the same management as dessert in this analysis and one farm's central orchard was Bramley cooking), day number squared (to understand if there is a mid season peak in abundance) and day number. The intercept is conventional farm management, with cider apples in farm management model and cider apples in insecticides model. All continuous variables are standardised and values in bold are significant (P=<0.05). VIF is the variance inflation factor value derived from squared GVIF^(1/(2*Df)), where any variable above 5 was considered correlated and removed from the global model before dredging. Day number squared was removed under the VIF tests.

		Pest moth abundance with insecticides				**Pest moth abundance with farm management**			
Fixed Effects		Value	Standard Error	*P*-value	*VIF*	Value	Standard Error	*P*-value	*VIF*
Farm Management	Conventional (Intercept)	-	-	-	-	2.084	0.226	**<0.001**	1.313
	IPM	-	-	-	-	-0.003	0.155	0.983	1.313
	LEAF	-	-	-	-	0.014	0.180	0.938	1.313
	Organic	-	-	-	-	0.082	0.258	0.753	1.313
Insecticides	Broad spectrum insecticide volume	-0.079	0.13	0.545	1.369	-	-	-	-
	Specific insecticide volume	-0.011	0.096	0.910	1.656	-	-	-	-
Apple types	Cider (intercept)	2.094	0.185	**<0.001**	1.203	2.084	0.226	**<0.001**	1.122
	Cooking	-0.478	0.779	0.544	1.203	-0.575	0.760	0.455	1.122
	Dessert	-0.978	0.272	**<0.001**	1.203	-1.002	0.266	**<0.001**	1.122
Environmental/ Landscape impacts	Day number	-1.361	0.100	**<0.001**	1.014	-1.36	0.09	**<0.001**	1.016
	Orchard size	0.024	0.088	0.784	1.092	0.033	0.11	0.736	1.729

Appendix M

Apple ermine pest abundance GLMMs ‘full’ model average summaries

Generalised linear mixed effects full model average summary examining apple ermine moth abundances as the response variable. Investigation of the response variables was separated into two models, to analyse fixed effect variables of: farm management and insecticides application separately. Each glmm included fixed effect variables: apple type (cooking, cider and dessert – juice was classed as the same management as dessert in this analysis and one farm’s central orchard was Bramley cooking), day number squared (to understand if there is a mid season peak in abundance) and day number. The intercept is conventional farm management, with cider apples in farm management model and cider apples in insecticides model. All continuous variables are standardised and values in bold are significant (P=<0.05). VIF is the variance inflation factor value derived from squared GVIF^(1/(2*Df)), where any variable above 5 was considered correlated and removed from the global model before dredging. Day number squared was removed under the VIF tests.

		Pest moth abundance with insecticides				**Pest moth abundance with farm management**			
Fixed Effects		Value	Standard Error	*P*-value	*VIF*	Value	Standard Error	*P*-value	*VIF*
Farm Management	Conventional (Intercept)	-	-	-	-	2.418	0.277	**<0.001**	1.392
	IPM	-	-	-	-	0.030	0.352	0.933	1.392
	LEAF	-	-	-	-	0.518	0.393	0.060	1.193
	Organic	-	-	-	-	-1.189	0.394	**<0.001**	1.392
Insecticides	Broad spectrum insecticide volume	0.221	0.189	0.245	1.232	-	-	-	-
	Specific insecticide volume	0.103	0.148	0.486	1.785	-	-	-	-
Apple types	Cider (intercept)	2.240	0.169	**<0.001**	1.325	2.418	0.268	**<0.001**	1.423
	Cooking	0.052	0.51	0.919	1.325	-0.033	0.256	0.904	1.423
	Dessert	0.159	0.288	0.583	1.325	-0.007	0.945	0.941	1.423
Environmental/ Landscape impacts	Day number	0.784	0.181	**<0.001**	1.028	0.762	0.177	**<0.001**	1.042
	Orchard size	-0.004	0.007	0.946	1.042	-0.224	0.182	0.219	1.981

Appendix N
Fruit tree tortrix pest abundance GLMMs 'full' model average summaries

Generalised linear mixed effects full model average summary examining apple ermine moth abundances as the response variable. Investigation of the response variables was separated into two models, to analyse fixed effect variables of: farm management and insecticides application separately. Each glmm included fixed effect variables: apple type (cooking, cider and dessert – juice was classed as the same management as dessert in this analysis and one farm's central orchard was Bramley cooking), day number squared (to understand if there is a mid season peak in abundance) and day number. The intercept is conventional farm management, with cider apples in farm management model and cider apples in insecticides model. All continuous variables are standardised and values in bold are significant (P=<0.05). VIF is the variance inflation factor value derived from squared GVIF^(1/(2*Df)), where any variable above 5 was considered correlated and removed from the global model before dredging. Day number squared was removed under the VIF tests.

		Pest moth abundance with insecticides				**Pest moth abundance with farm management**			
Fixed Effects		Value	Standard Error	*P*-value	*VIF*	Value	Standard Error	*P*-value	*VIF*
Farm Management	Conventional (Intercept)	-	-	-		4.267	0.137	**<0.001**	1.411
	IPM	-	-	-		0.123	0.166	0.465	1.411
	LEAF	-	-	-		-0.642	0.203	**0.002**	1.411
	Organic	-	-	-		-0.098	0.18	0.591	1.411
Insecticides	Broad spectrum insecticide volume	0.008	0.037	0.832	1.208	-	-	-	-
	Specific insecticide volume	-0.181	0.074	**0.015**	1.730	-	-	-	-
Apple types	Cider (intercept)	4.146	0.062	**<0.001**	1.317	4.267	0.137	**<0.001**	1.384
	Cooking	0.034	0.175	0.845	1.317	0.157	0.337	0.642	1.384
	Dessert	-0.005	0.051	0.929	1.317	-0.029	0.095	0.757	1.384
Environmental/ Landscape impacts	Day number	-0.155	0.066	**0.019**	1.031	-0.183	0.058	**0.002**	1.055
	Orchard size	0.119	0.078	0.130	1.090	0.286	0.072	**<0.001**	1.804

Supplementary Materials 1

Rationale for choice of sentinel prey used

The use of artificial sentinel prey was chosen for Chapter 4's predation experiment. This decision was based on the following information:

Predation rate can be over or underestimated due to a variety of factors, as described in Sam et al. (2015). In nature, live herbivorous caterpillars cause plants to release volatiles as cause leaf damage is caused, which chemically attracts predators (Gentry & Dyer, 2002; Mantyla et al., 2008). Therefore, artificial prey may underestimate the natural predation rate from birds, as they are not causing the same chemical attraction. In contrast, live caterpillars have defence strategies, such as dropping from leaves or camouflage techniques, which reduce their detection from predators. Artificial prey lacks these strategies and therefore their predation rates may be overestimate natural predation rates. Further differences in prey types, and reasons for the artificial choice, are listed below.

- The use of meal worms has been used to infer predation rate in a range of habitats, but the taxa of predators was assumed on occasions where meal worms disappeared (Boyle, 2012).

- Plasticine allows attack marks to be identified (Howe et al. 2009) so camera traps or bird exclosures are not essential.

- Plasticine will not disintegrate in light or heavy rain. Pastry is more likely to but the decision to place for 24 hours during low rain predictions was taken.

- Garfinkel & Johnson used live codling moth larvae during winter in their experiment (2015). They camera trapped a proportion of predation test sites to identify key predators species and used bird exclosures to control for non-bird predators. However, I was not able to release use live pests in my working orchard system.

- Elina Mäntylä tried experimenting using mealworms in Finland, however birds were not interested in predating meal worms in her study system (Mäntylä, E. *Pers Comms,* 2016).

- Exclosures are needed if live prey are used, in order to account for non-bird predators (Ebeling, A. *Pers Comms*, 2016). These exclosures can be time consuming and costly.

Thus, there are both benefits and dis-benefits of using artificial prey but here I have set out the practical reasons that favoured the use of artificial prey over live prey.

Supplementary Materials 2

Determining Colour Choice of Sentinel Prey Experiment

Behavioural tests in a Cambridge aviary were undertaken to decide on which colour plasticine birds were most attracted to or if birds avoided a certain colour.

Location: Madingley Woods Lab, Cambridge, 29th Feb – 2nd March 2016

Methods: Behavioural reactions to prey colours using non-toxic plasticine "Newplast", Devon TQ126RY. Four blue tits were captured for behavioural tests for a separate Cambridge Evolutionary Ecology study. I was able to use the three birds to randomly test the reactions to different coloured plasticine. Prey colours were randomly chosen to be presented 10 times, to one bird at a time, and the reaction to the prey presentation was recorded.

Results: Birds were equally attracted to green and cream plasticine caterpillars.

Conclusion: These results allowed me to use both colours during the sentinel prey experiments. Green clay caterpillars were placed on leaves, to mimic the appearance and behaviour of fruit tree tortix caterpillars; and white clay caterpillars were placed on the apple tree trunk, behind loose bark if available, to mimic the appearance and behaviour of Codling moth larvae.

CHAPTER 5

Can a natural pest control service on apple orchards support high yield and economic value per tree?

5.1 ABSTRACT

Through providing ecosystem services, biodiversity can be managed within agriculture to support natural services such as pest control. Our understanding of how farming systems that support natural pest control compare with a synthetic alternative from intensive agriculture, in terms of value, is not well understood. Here, I compare the yield and yield value of organic and non-organic orchards to understand decision making when choosing between two methods of pest control: a natural pest control system (organic farming) or the synthetic alternative (conventional, LEAF and IPM farming).

Cider apple yield per hectare was significantly lower on organic orchards in comparison to non-organic orchards (except LEAF), but increased insecticide use does not fully explain this; farm management variation is found to be more important than insecticide use showing that other practices within each farming system harbour higher yields and yield value than insecticides alone do. Although wild insectivorous bird variables had no negative or positive impact on yield and yield value per tree in non-organic orchards, insecticide use also has no statistically detectable positive impact on yield and yield value in non-organic orchards.

With increased use of broad-spectrum insecticides on non-organic farms, yield value per hectare of non-organic farms is in-different to organic yield value per hectare, due to the increased market price for organic produce, and increased production costs of using higher volumes of insecticides. This highlights the continued need to financially support organic growers who support increased biodiversity and ecosystem services that organic orchards foster.

Finally, LEAF farms do not support the farmer financially due to the reduced yield and yield value compared to both organic and other non-organic farms – one of the first scientific studies providing negative results on a national farm management programme aimed at improving farmer prosperity.

5.2 INTRODUCTION

5.2.1 The value of Ecosystem Services and its trade-offs

The loss of biodiversity due to human-driven agricultural expansion, logging and development has caused major concerns for species declines, with extinction rates increasing to similar levels of the last five global mass-extinction events in the last 500 million years (Barnosky et al., 2011; Tilman et al., 2017). Agriculture threatens more species than any other human activity (Balmford et al., 2005; Tilman et al., 2017). To keep up with the growing food demands arable croplands alone are predicted to expand by 18%, that is 268 million hectares of non-agricultural lands being converted to match dietary demands (Tilman, 1999). Agricultural interventions will be needed in order to reduce the industry's negative impact on planetary boundaries, and instead act as the significant step toward the sustainable development of our planet (Campbell et al., 2017).

Economic benefits from farming systems through increases in yield, financial support through government led strategies, and financial incentives - available through selling produce to the organic market - are major factors in farm management decision making processes. A choice must also be made on whether farmers are able to adopt less intense agricultural systems to reduce agriculture's negative impacts to nature (Crowder & Reganold, 2015; Reganold & Wachter, 2016). Financial values are imperative to farmers as farm income must be financially and ecologically sustainable. Valuation methods are often promoted to increase the awareness of biodiversity in this financial decision-making processes. In environmental economics, ecosystem services are goods from nature that people value, that increase human welfare (Bateman et al., 2011). As biodiversity is now recognised as an essential but diminishing resource, it fits the definition of economics "the efficient use of scarce resources" lending itself to being analysed economically (Edwards & Abivardi, 1998: 240).

Insectivorous birds are a provider of pest control services in this study and functional biodiversity measure (Penvern, 2019) that are "by and large invisible and underappreciated" (Wenny et al., 2011: 8). Considering 50% of bird species predominately eat invertebrates and an additional 25% eat invertebrates occasionally (Sekercioglu, 2006), their importance as a pest control service provider is great and is outlined in Chapter 4. However, there is still a challenge to quantify their importance in a meaningful way as the literature lacks ornithological research on ecosystem services and valuation (Wenny et al., 2011). This work is necessary to understand current farm management choices that can hinder or support wild bird communities on farmlands.

A supportive farm management choice for birds is organic farming (Chapter 1) and large amounts of literature has so far focussed on win-win scenarios for biodiversity on organic orchards that support multiple ecosystem services. One example, most closely related to this study, is by Martínez-Sastre, Miñarro & García (2020) who found no trade-offs existed between biodiversity groups and ecosystem functions. Only positive effects of animal biodiversity on pest control and pollination services in apple orchards are seen, claiming a "win-win scenario for animal-biodiversity and ecosystem services" (Martínez-Sastre, Miñarro & García, 2020: 1). However, often the trade-off is more visible when introducing yield into comparisons and equations. Organic farming has notoriously lower crop production (yield), and is usually the trade-off noticed and acknowledged in the ecosystem service provision of organic farming (Samnegård et al., 2019; Seufert et al., 2012; Bengtsson, 2015; Crowder & Reganold, 2015; Mäder et al., 2002; Jouzi et al., 2017). Garibaldi et al. (2016) finds the level of agricultural intensity to be an important predictor of yield, where yield reductions range from 5% to 34% lower on organic compared to conventional farms, and this is dependent on the quality of organic management and crop type. Many more examples of such trade-offs and the relationships between biodiversity, ecosystem services and economic value are highlighted by Paul et al. (2020).

5.2.2 Current research and gaps in knowledge

The positive impacts of ecosystem services in agriculture have been widely researched, advocated, and incorporated into different types of farm management practices to reduce crop damage and/or increase crop yield that will financially benefit the farmer. Direct comparisons of an ecosystem service reliant system supported by increased biodiversity, such as pest control delivered by birds, to an intensive system that replaces this ecological system with a synthetic one – i.e., chemical control - is less demonstrated.

Recent literature has started to focus on these links between three trophic levels of predator species, their prey and plant growth (crop yield), through an ecosystem service value lens (Wenny et al., 2011; García et al., 2018). Some of these studies have been highlighted here. Yet only a handful of studies have been able to estimate the value of biological pest control for farmers (Östman et al., 2003; Bengtsson, 2015), and even less so from birds in orchards. This is surprising given the decades of intensive agricultural research available on the chemical control of pests (Bengtsson, 2015). Furthermore, the use of marketable yield values and predation estimates from birds on working farms, instead of experimental fields, is not well documented (Cardinale et al., 2012; Letourneau et al., 2015; Letourneau et al., 2009). Importantly, the presence of predation pressure by birds in orchards has been investigated by García (2021) who showed how nest boxes in orchards were shown to increase predation pressures on apple pests by great tits (*Parus major*) and glue tits (*Cyanistes caeruleus*), which reduced pest damage and increased crop yields. This chapter looks to add to this literature and highlights other important research to consider in this section.

Conversely, Bengtsson (2015) found that conventional farming had a larger positive impact on barley yields than the farms which used natural enemies as a biological control of aphids, with yields being significantly higher on conventional farms. Although conventional farming yielded higher crops than organic methods, they still found that biological control was related to yield but most of the variation was explained by human inputs within the conventional farm management systems. This is one of the first studies to segregate human inputs with natural alternatives – biological inputs from nature – to analyse the contribution that regulating services form a relationship with human inputs, to provide the final ecosystem service of food provision (Bengtsson, 2015; Mace & Bateman, 2011). Bengtsson (2015) explains how the final ecosystem service (yield) should be valued in similar studies, as yield is easily understood by all stakeholders: it can change in response to other ecosystem processes involved and is already valued by society (Lele et al., 2013).

A more recent study considered apple yields when comparing organic and IPM orchards in Europe to find that organic orchards have a 43% decrease in yield on average, but with a large variation - some higher producing organic orchards can produce more than IPM (Samnegård et al., 2019). The same study found that natural enemies (arthropod abundances) were related to apple production, however the study does not consider bird abundances as a natural enemy provider. Samnegård et al. (2019) make a poignant conclusion from their study: if increases in ecosystem service provision do not positively or negatively impact crop production, then it is important to continue with the integration of ecosystem service provision on farms as it can still assist with pest control and is environmentally beneficial.

The relationship between biodiversity, ecosystem services and economic value is complex and although natural pest control services can be valuable for sustainable agriculture (Letourneau et al., 2009), whether this is valuable *enough* to farmers is not well known, especially understanding value from the farmer's perspective (Segura et al., 2004; Zhang et al., 2018; Martínez-Sastre et al., 2020). Increased on-farm biodiversity will often lead to economic disadvantages, leading to the lack of practical uptake of organic farming practices that could ultimately lead to provision of on-farm ecosystem services (Paul et al., 2020). Furthermore, there is usually an academic focus on ecosystem processes rather than farm outcomes whether that's profits or yield (Kleijn et al. 2019).

The final ecosystem service valuation, apple yield, has been utilised in this chapter to explore the reasoning behind the choice of farming systems prevalent in UK's apple growing community (Kleijn et al., 2019; Bengtsson, 2015). Furthermore, a focus on yield as the final service is a suitable mode of communication to the farming community when conveying results. Currently, we know that organic farms have increased biodiversity (Chapter 3) and that they provide an increased pest control ecosystem service (Chapter 4), yet non-organic farming with pesticides remains a major farming system today, and reasons for this are likely economical. With a focus on yields and yield value at the farm level this chapter will bridge some of the missing gaps in the literature, highlighted by Kleijn et al., (2019), to shed more light on the reasons why natural farming systems may not be incorporated at a faster rate by farmers.

This study is a comparison between a farming system that relies on ecosystem services for natural pest control (organic) with 3 types of non-organic farming systems that replace this ecological system with a synthetic one (chemical control). To my knowledge, this is not available in the literature and is needed to understand farmer decision making. This chapter connects three trophic levels: birds, their prey and apple yield, through an ecosystem service value lens (Wenny et al., 2011) and expands on Daniels et al. (2017) to understand the differences between yields and yield value between a natural pest control from wild birds and the synthetic alternative. Using the literature above this chapter looks to build on these examples and include yield per tree with pest control ecosystem service provision (Chapter 4).

5.2.3 Objectives of Study

Previously (Chapter 3), I have shown that organic orchards hold a higher biodiversity of birds than other farm managements that use non-organic principles. This supports a higher functional diversity of insectivores that was shown to increase predation of orchard pests under organic management.

This chapter compares yields and yield value at the farm level between farming systems and explores the impacts of chemical inputs as the replacement of wild bird pest control across all farm types.

A wild bird pest control service on yield and yield value is explored amongst non-organic farm systems only. The reason for this is because organic and non-organic systems differ in many respects, other than just increased insecticide or reliance of ecosystem services like pest control from nature.

This chapter sets out to answer three main questions:

1. How do apple yield and yield value vary between farming systems?

2. What role does chemical control play in yield and yield value differences between farming systems?
3. Is there any evidence that wild insectivorous birds provide a service to non-organic farming systems?

5.3 METHODS

This study used 30 farms across Herefordshire and the surrounding bordering counties to investigate apple yields, yield value and insecticide usages in 2015 and 2016. Within those 30 farms, 28 grew cider apples in at least one field, 13 grew dessert apples and 13 grew apples for the juice market, either solely for juice or would sell at least some of their dessert apples to the juice market. Most farms grew a variety of apples for different markets. All apple type yields and prices were included in initial analyses; however, cider was chosen to be the focus throughout the analysis to answer the objectives set out above. Not all farms had dessert apples or sold juice apples on their farm, but the majority have cider orchards.

5.3.1 Cider apple yields

Cider yields were obtained through farmer questionnaires, given at the end of growing seasons in November to December in 2015 and 2016. These yields were measured in bins of apples ready for sale and each bin equates to a tonne of apples. Apples that were not sold were not counted as yield as the farmer would not receive payment for unsold apples. Usually, unsold apples that are not counted or weighed are left on the orchard for winter bird feed.

5.3.2 Cider apple yield value

Apple yield value data was not taken on a per farm basis, as not all farms sold their produce, especially organic farms who were artisanal cider producers. For this reason, in order to create an apple yield value, an average farm-gate apple price per tonne was agreed upon between key stakeholders, including: farmers who bought produce themselves; John Worle, an apple tree nurseryman and apple cider buyer for over 50 years; the National Association of Cider Makers (NACM); and English Apples and Pears Ltd. An average price for years 2015 and 2016 for the study area for both organic and non-organic cider apples were agreed upon during consultations held in 2017 (*table 5.0*).

Table 5.0 – The average price of cider apples sold at farm-gate during the study years.

Year	Conventional Cider £ / tonne	Organic Cider £ / tonne
2015	120	130
2016	122	130

To create the final yield value, production cost was then subtracted from the price apples sold at. For the purpose of this study, only insecticide costs per hectare and cost of application were used as the cost of production. The cost of insecticides increased in 2016 due to a cost-effective broad-spectrum chemical (Chlorpyrifos) becoming unavailable for use by farmers. For organic farms, there was no associated cost of production as they do not spray insecticides.

5.3.3 Tree density

Tree density was determined using Google Earth Pro, version 7.3.4.8573 (Google Earth, 2015;2016). Each point count identified in Chapter 2 was used as the reference point to measure tree density per orchard. A 10m radius was used to construct a circle around each point count, to create the orchard sample area for calculating tree density. If a tree was situated over half inside the 10m^2 circumference the tree was counted, if it fell over half outside the circumference, it was not counted. If a point count itself was near a field boundary the new circumference may include a non-orchard field, therefore these were not included for tree density calculations. Orchards where cider apples were grown on vines were too small to detect per tree, so these fields were discarded from analysis. From Google Earth aerial imagery (Google Earth, 2015;2016), cider trees of organic and conventional orchards can be easily counted, dependent on the year. The year of aerial image chosen was based in this order:

1. Year of aerial image closest to the survey years (2015 & 2016).
2. Clearest image available if 2015/2016 were unclear or trees too small to zoom into.

5.3.4 Chemical control

Insecticide spray records, including price per hectare, were obtained through farmer questionnaires. Litres and total price per hectare, per farm of insecticide were used in data analyses. All records were kept anonymous and have been analysed without farm names for farm data confidentiality. Fertilisers are not included in this study but it is important to keep in mind they also affect apple yield on non-organic farms (Garratt et al., 2011). Copper, used on organic farming as a fungicide to reduce fungal diseases such as apple scab, was not included in analyses. For the purpose of this study, I focussed on insecticide-use only.

5.3.5 Data analysis

To answer the questions set out in the objectives of study, this analysis section has been separated into three, to break down how each question has been answered. All analyses were completing using the statistical programme 'R' version 3.4.3 (R Core Team, 2015). I used a variety of R packages during analysis including lme4 (Bates et al., 2015). The R package "ggplot2" was used to create the graphs for visualisation in each result (Wickham, 2016).

How do apple yield and yield value vary between farming systems?

Simple linear models were constructed to compare yield and yield value per hectare and per tree between farming systems. Cider yields and yield value for each year was nominated as the response variable in models with farm category as the independent (predictor) variable. Cider yields were kept as separate years and separate simple linear models in R were constructed per year. A random effect was not necessary, as only the relationship of cider yields in each farm category were to be examined, to understand if there was a significant difference on yields between organic and non-organic farms. There were no missing values in the data frame that needed to be omitted (Crawley, 2013). Tukey's post hoc analysis is performed on the model to understand where the differences lie within farm categories.

What role does chemical control play in yield and yield value differences between farming systems?

The relationship between insecticide volume used and yield produced in tonnes per tree and per hectare was analysed in separate linear models for 2015 and 2016 in the R library "lme4". Firstly, simple linear models were constructed using cider yield per hectare / tree as the response variable and litres of insecticide used per hectare as the independent variable. A second model was constructed which included farm management as an additional independent variable. Both models were compared to understand if insecticide use was sufficient to explain any yield and yield value differences between farm category types. Comparison of models was made using one-way analysis of variance (anova), which performs a Chi-squared test to test significance, where the lower AIC is the better model fit, and the p value given determines the significant difference between models. If the anova is significant, Tukey's post hoc analysis is again performed on the model to give differences between categories. Lastly, chemical control use and costs per hectare were investigated between years using linear models with only chemical control of each year within the model.

Is there any evidence that wild insectivorous birds provide a service to non-organic farming systems?

To look at the impact of wild birds on yields, I focused on differences across only non-organic farm management types. Removing organic farms from the dataset allows the consideration of differences within conventional farms only, and excludes any impacts seen from natural pest regulation from organic farms. This decision is based on the theory that organic systems differ to conventional systems in a multiple of ways. Chapter 4 discovered the ecosystem service driven difference (i.e., the natural pest control on organic orchards), and on top of this difference, chemical inputs such as pesticides, fertilisers and herbicides will differ between organic and conventional systems as well as the structure of orchards, number and age of trees.

Using linear mixed effect models in R "lme4" package, I first fitted models of yield and yield value in relation to a range of plausible predictor variables, then added predictor variables that describe insectivorous wild bird community to these models to construct a Global model, ready for top model selection before model averaging. Overdispersion was tested on Global models using the dispersion parameter and model interactions were tested for using AIC in model interactions and model non-interactions in anova. No overdispersion was found in the models of this chapter. Correlation was tested in the same way as previous chapters, using correlation coefficient on continuous variables in the model as well as a variance inflation factor (VIF) test on the fitted model. Variables with a VIF value over 5 were removed from the model to simplify and improve accuracy. The use of VIF in this chapter resulted in the creation of separate models for each bird community variables: insectivore abundance, species richness and diversity. Linear mixed effect top model selection and model averaging process follows the same structure and statistical packages as data analysis for Chapter 3 and 4. Short details and any changes made to Chapter 3 methodology has been provided below.

Mixed model description, model average and model testing processes

LMER Global models were constructed with one response variables for each model, either: a) yield per tree or b) yield value per tree. This chapter analysis use LMER instead of GLMER as the response variables are continuous and the random effect structure has changed from a nested structure "(Farm.ID/Visit)", to just "(Farm.ID)". Like previous chapters, insectivore community metrics were used as independent variables in all models, with the exception of the non-bird models used as a comparison. All independent variables considered, *a priori,* to be ecologically important are similar to Chapter 3 and 4, including: bird community metrics such as insectivore diversity, species richness and abundance; insecticide usage; Woody Cover; orchard size (ha); and farm management type. Continuous fixed effect variables were standardised using R scale function. Orchard size is a description of the entire farm area that is covered in orchards and Woody cover is the amount of natural habitat on farm and adjacent to the orchard edge. Global models were then put through the same information theoretic (IT) approach model averaging process as previous chapters, where 'top models' are chosen using AIC Delta < 6 AIC values

(Harrison et al., 2017). The 'conditional' model average has been reported whilst 'full' model averages for this chapter have been reported in *Appendices O and P.*

Structure of results

Yield and yield value models have been separated by: yield and yield value, bird community metrics, insecticide use and finally a null model with no bird community metrics for comparison. All models have been displayed in *table 5.1.*

Table 5.1 – Model separation visualisation. All full models include farm characteristic independent variables of *woody cover* and *orchard size, insecticide amount* and *farm managements:* IPM, LEAF and Conventional, The same models are repeated for two separate response variables shown as A and B.

	Response variables		**Farm management & independent variables**	**independent variables included as separate models**
Main results & Appendix O	A. B.	Cider apple yield per tree Cider apple yield value per tree	+IPM +LEAF +Conventional +Woody Cover +Orchard size +Insecticide use	insectivore abundance insectivore species richness insectivore diversity
Appendix P	A. B.	Cider apple yield per tree Cider apple yield value per tree	+IPM +LEAF +Conventional +Woody Cover +Orchard size +Insecticide use	No insectivore variables

5.4 RESULTS

Organic farms have lower cider apple yields and yield value per hectare in comparison to IPM and conventional farming, but not lower than LEAF. No difference was seen between non-organic farming systems (LEAF, IPM and conventional).

Although yields were positively correlated with insecticide use on orchards when including organic farms, it doesn't explain all the differences in yield and yield value between farm types (*section 5.4.3)*. Furthermore, in mixed model analysis the amount of insecticide within non-organic orchards is not a significant influence in yield increases (*section 5.4.4)*. An important result to highlight is the difference in yield value between 2015 and 2016 seen in linear models that include insecticides and farm category (*section 5.4.2).* In 2015 yield value per hectare was not significantly different between conventional and organic farms, whereas in 2016 organic was significantly less valuable per hectare. This effect coincides with the insecticide cost increases of 2016, encouraging farmers to use significantly less insecticides in 2016 that did not differ significantly to the zero insecticide use of organic. Yield and yield value per tree tells a different story, where organic has the same or higher yields and yield value per tree than non-organic farms. Finally, the wild bird community metrics have no significant (positive or negative) impact on yields or yield value within non-organic farms.

5.4.1 Tree Density between farm management types

Tree densities differ between farm management systems, with a significant difference in tree density between farm categories reported (one-way ANOVA results: $F(3,552)=136.8$, $p<0.001$) shown in *Figure 5.0*. Post hoc comparisons using the Tukey test were carried out. There was a significant difference between organic and LEAF ($p<0.001$), where organic had 395 fewer trees per hectare than LEAF on average. There was also a significant difference between organic and IPM ($p<0.001$), where organic had 444 fewer tress per hectare than IPM on average. Finally, there is significant difference between organic and conventional ($p<0.001$), where organic had 420 fewer trees per hectare than conventional on average.

Due to these differences between farm categories, there is a need to use tree density within the data analysis of mixed models where yield and yield value are the response variable. Therefore, tree density has been added to every model as yield or yield value *per tree*. As tree density is highly correlated to farm type, using yield per tree and yield value per tree as the response variables avoided collinearity issues during model analysis.

Yield per tree is calculated simply as: yield per hectare / tree density per hectare. *Yield value per tree* is calculated as: yield value per hectare / tree density per hectare.

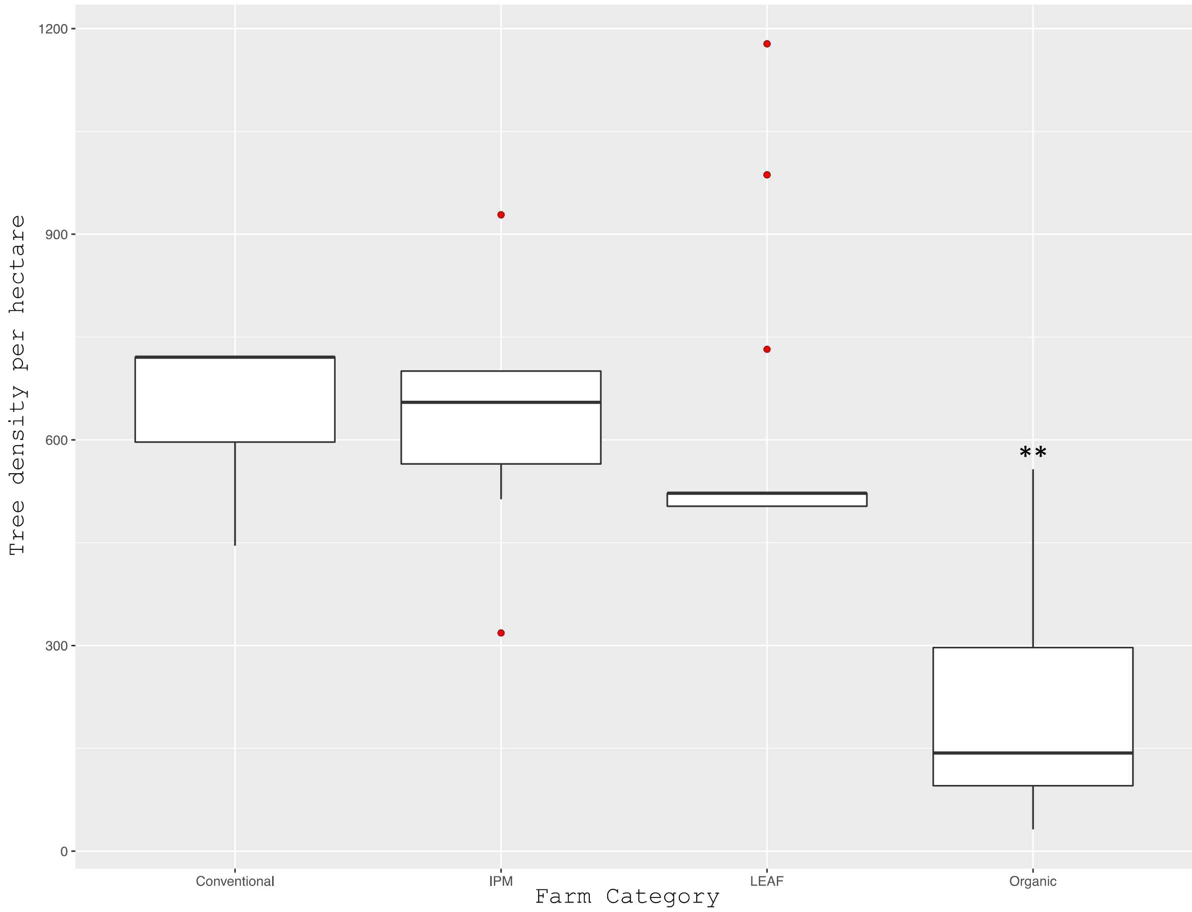


Figure 5.0 Box plot for average number of apple trees per hectare on conventional, IPM, LEAF and organic orchards. Significant farm category differences are indicated with "**". Linear regression and post hoc analysis results (Tukey p<0.001) show organic orchards have significantly less trees per hectare than all other groups.

5.4.2 How do apple yield and yield value vary between farming systems?

Cider apple value was collected using data for years 2015 and 2016 for both organic and conventional cider apples prices per tonne. *Table 5.0* simply shows 2016 prices slightly increased from 2015 for conventional apples, yet the price remained the same between years for organic apples. The price per tonne for organic apples is 8.3% higher than conventional apples in 2015, and just 6.6% higher than conventional in 2016.

Cider apple yield per hectare

A simple linear model used farm category as the single predictor variable with cider apple yield (*figure 5.1a and 5.1b*) as the response variable using the lme4 library in R (R Core Team, 2015). The difference between farm types and yield is significant between organic farms and IPM ($p < 0.05$ in 2015 and 2016) as well as organic and conventional ($p < 0.05$ in 2015 and 2016), but not LEAF ($p > 0.05$). Post hoc analysis comparisons showed that organic was significantly less than conventional (*Tukey $p < 0.001$ in 2015; $p = 0.012$, 2016*), and IPM (*Tukey $p < 0.001$ in 2015; $p = 0.002$ in 2016*), but not different to LEAF (*Tukey $p = 0.144$ in 2015; $p = 0.180$ in 2016*). All non-organic yields per hectare did not significantly differ in both years (*all $p>0.05$*).

Cider apple yield value per hectare

In the same format as yield, farm category was added to a simple linear model as the fixed effect variable with the response variable of cider apple yield value (*figure 5.2a and 5.2b*). The same overall results are portrayed; the amount of income organic apples generate is significantly less per hectare than non-organic ($p < 0.05$ in 2015 and 2016), but not LEAF. Post hoc Tukey analysis results concluded that organic income generation per hectare was significantly less than conventional (*Tukey $p < 0.001$ in 2015, 2016*), and IPM (*Tukey $p < 0.001$ in 2015, 2016*) but was not significantly different to LEAF (*Tukey $p = 0.342$ in 2015; $p = 0.168$ in 2016*). All non-organic yield value per hectare did not significantly differ in both years (*all $p>0.05$*).

Cider apple yield and yield value per tree results

Linear model results for total yield and yield value per tree are shown in *Figure 5.3a and 5.3b.* Organic yield is not significantly different to conventional yields in model results ($p = 0.056$), and yield value is significantly higher on organic farms compared to conventional ($p = 0.002$). Post hoc comparisons using the Tukey test were carried out. Firstly, there is significant difference in the yield value between organic and conventional per tree (yield: Tukey *$p <0.001$),* where **organic has higher yield value than conventional, per tree.** There is also significant difference between organic and LEAF yield and yield value (yield: Tukey *$p= 0.049$; yield value: Tukey $p <0.001$*), where organic has more yield and value per tree than LEAF on average. There is significant difference between IPM and conventional where IPM had significant increase in yield and yield value per tree (*yield: Tukey $p = 0.0014$; yield value: Tukey $p = 0.002$*). Finally, there is significant difference between LEAF and IPM yields and yield value (*yield*: Tukey *$p <0.001$; yield value: Tukey $p <0.001$*), where LEAF has less yield and value than IPM on average.

All other farm categories show no difference between them, including yields of organic and conventional (*Tukey $p = 0.219$*), yield and yield value between organic and IPM (*yield: Tukey $p = 0.599$*; *yield value: Tukey $p = 0.999$*), and yield and yield value between LEAF and conventional (*yield: Tukey $p = 0.934$*; *yield value: Tukey $p = 0.893$*).

Figure 5.1a – Box plot from linear regression model results for cider apple yields in tonnes per hectare on conventional, IPM, LEAF and organic orchards in 2015. Linear regression and post hoc analysis results show organic orchards have significantly less yield than conventional (*Tukey p<0.001*) and IPM (*Tukey p<0.001*) but are not different to LEAF (*Tukey p = 0.144 in 2015*). Significant farm category differences between non-organic and organic are indicated with "**".

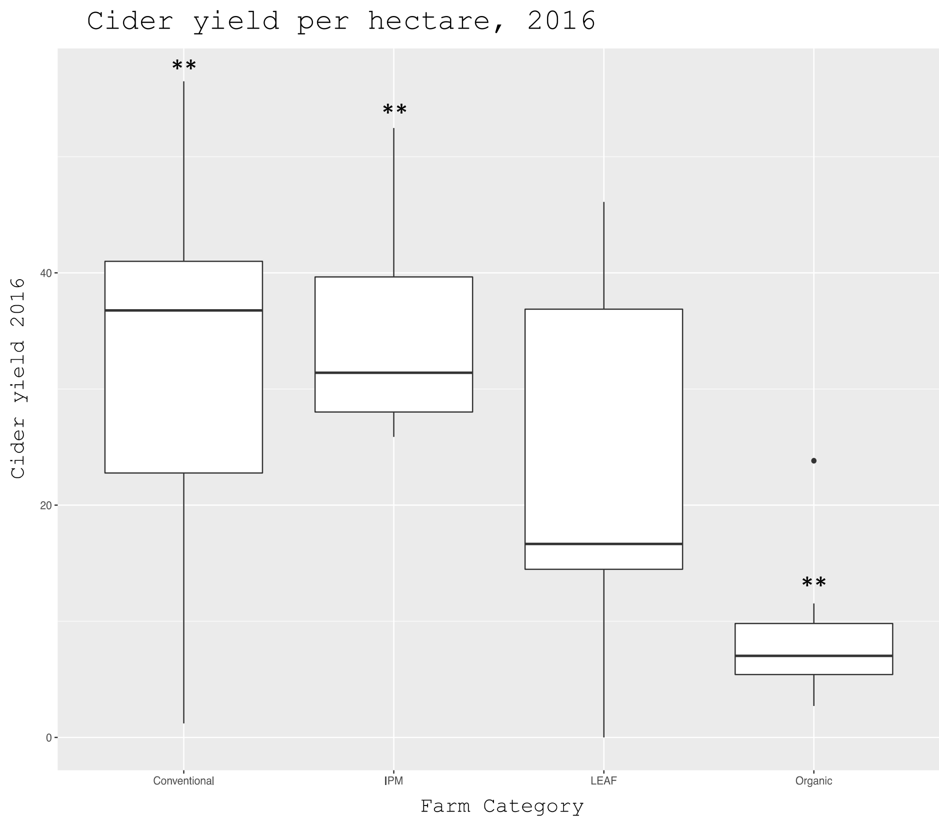


Figure 5.1b - Box plot from linear regression model results for cider apple yields in tonnes per hectare on conventional, IPM, LEAF and organic orchards in 2016. Linear regression and post hoc analysis results show organic orchards have significantly less yield per hectare than conventional (*Tukey p = 0.012*) and IPM (*Tukey p = 0.002*), but not different to LEAF (*Tukey p = 0.180*). Significant farm category differences between non-organic and organic are indicated with "**".

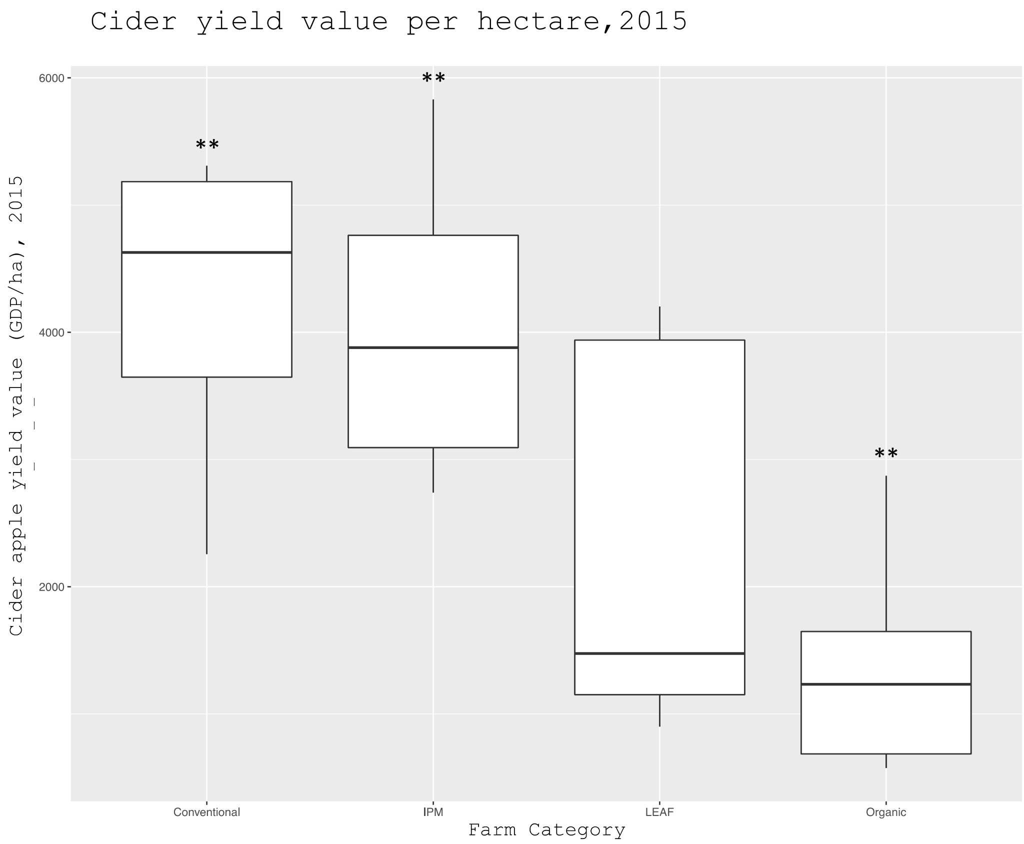


Figure 5.2a Box plot from linear regression model results for cider apple yield value in GDP per hectare on conventional, IPM, LEAF and organic orchards in 2015. Linear regression and post hoc analysis results show organic orchards have significantly less yield value per hectare than conventional (*Tukey p < 0.001*) and IPM (*Tukey p < 0.001*), but not different to LEAF yield value (*Tukey p = 0.342*). Significant farm category differences between non-organic and organic are indicated with "**".

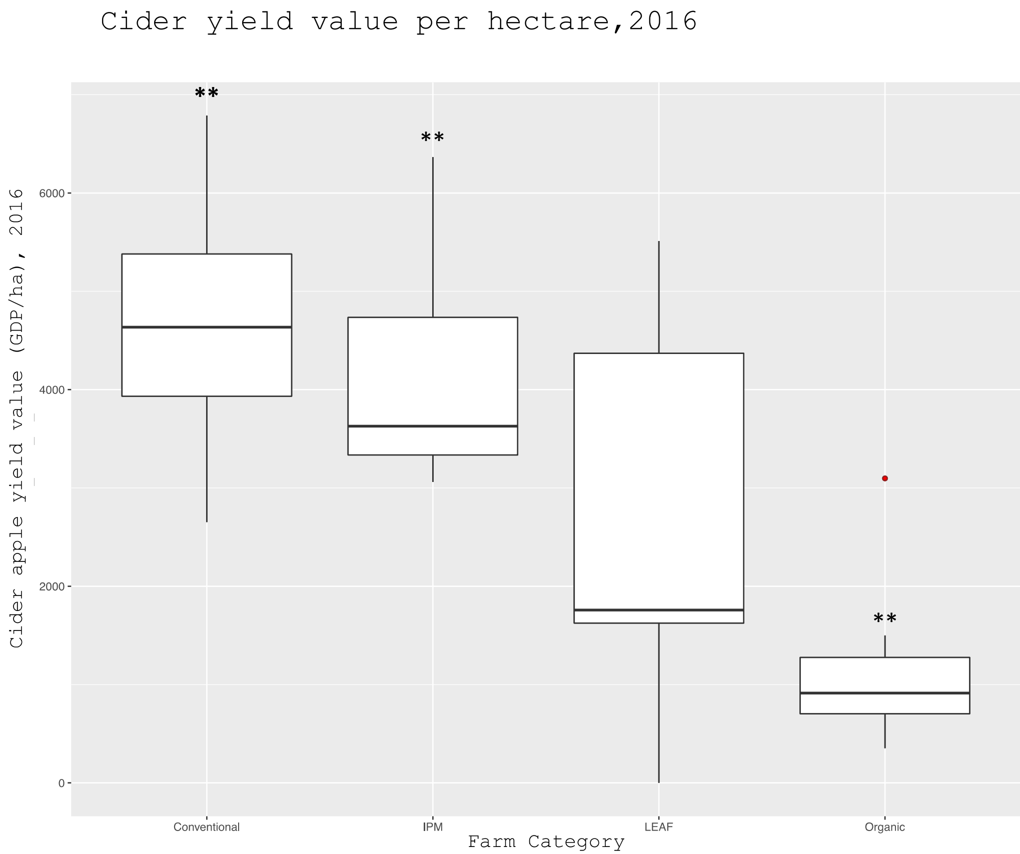


Figure 5.2b Box plot from linear regression model results for cider apple yield value in GDP per hectare on conventional, IPM, LEAF and organic orchards in 2016. Linear regression and post hoc analysis results show organic orchards have significantly less yield value per hectare than conventional (*Tukey p < 0.001*) and IPM (*Tukey p < 0.001*), but not different to LEAF yield value (*Tukey p = 0.168*). Significant farm category differences between non-organic and organic are indicated with "**".

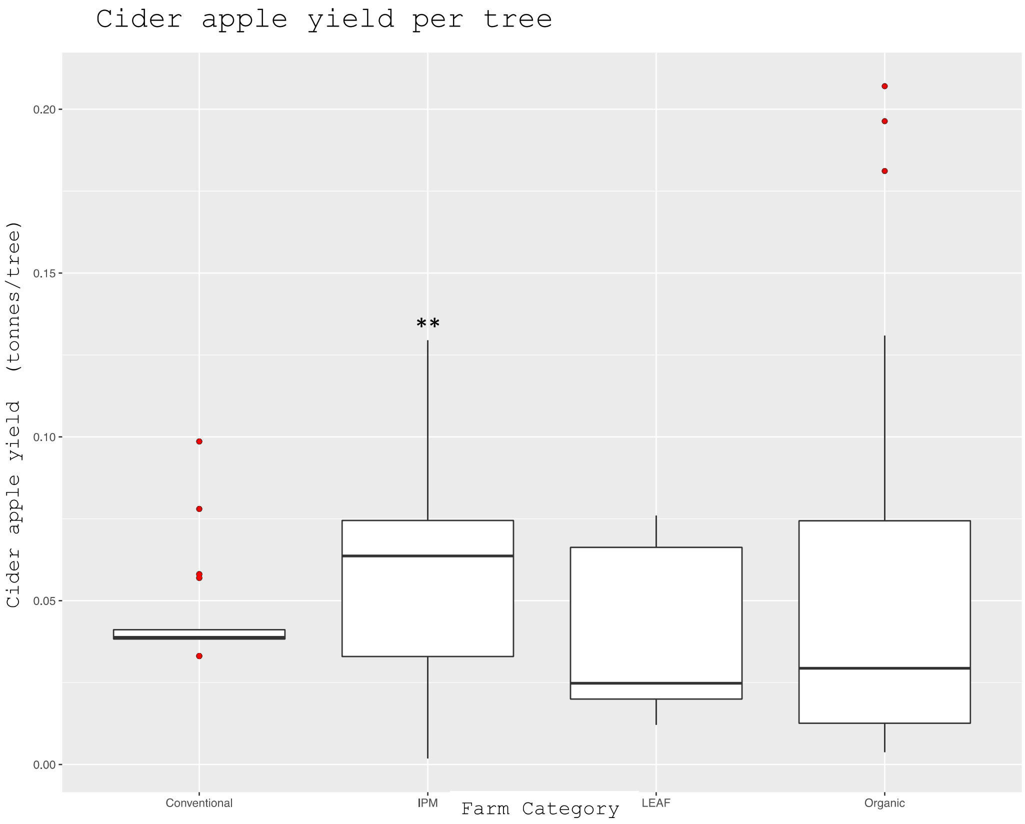


Figure 5.3a Box plot from linear regression model results for cider apple yield in tonnes per tree on conventional, IPM, LEAF and organic orchards across 2015/16. Linear regression and post hoc analysis results show organic orchards have similar yield per tree to conventional (*Tukey p = 0.219*), LEAF (*Tukey p = 0.049*) and to IPM (*Tukey p = 0.599*). IPM has significantly more yields per tree than conventional (*Tukey p < 0.0014*) and to LEAF yield (*Tukey p < 0.001*). Main significant farm category differences are indicated with "**".

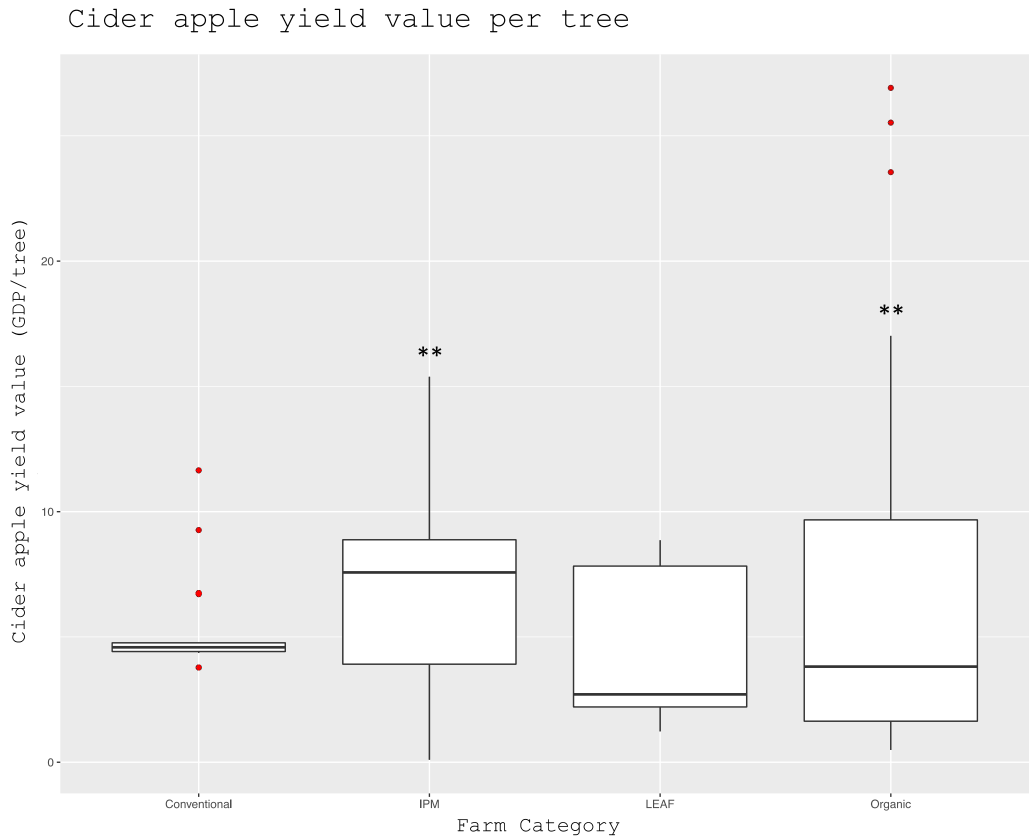


Figure 5.3b Box plot from linear regression model results for cider apple yield value in GDP per tree on conventional, IPM, LEAF and organic orchards across 2015/16. Linear regression and post hoc analysis results show organic orchards have higher yield value per tree to conventional (*Tukey p < 0.001*), and to LEAF (*Tukey p < 0.001*), but similar yield value to IPM (*Tukey p = 0.999*). IPM has significantly more yield value per tree than conventional (*Tukey p = 0.002*) and to LEAF yield (*Tukey p < 0.001*). Main significant farm category differences are indicated with "**".

5.4.3 What role does chemical control play in yield and yield value differences between farming systems?

Farm management's insecticide use differs between years

Insecticide use differed between years and between farm categories. Linear model regression results for insecticide use per hectare per farm category results show that in 2015, insecticide use varies significantly between farm categories (*One-way ANOVA results:* $F(3,23)=13.78$, $p < 0.001$, *figure 5.4a*). Tukey post hoc analysis applied on the linear model showed organic was significantly different to all other farm management types (*Tukey* $p= 0.002$ *for organic – LEAF,* $p< 0.001$ *for organic – conventional and organic - IPM).* All non-organic farm types were not different to each other in terms of insecticide use per hectare in 2015 (all variations of non-organic farms *Tukey* $p > 0.05$)

In 2016, although there is significant difference between farm categories and insecticide use (*One-way ANOVA results:* $F(3,23)=13.93$, $p < 0.001$, *figure 5.4b*), Tukey post hoc analysis shows **that conventional farms did not differ significantly to organic farms (zero insecticide) (*Tukey* $p = 0.579$ *organic - conventional*).** Organic farms still used significantly less insecticides than IPM and LEAF (*organic – LEAF Tukey* $p <0.001$; *organic – IPM Tukey* $p =0.003$). All non-organic farm types were not different to each other in terms of insecticide use per hectare in 2016 (all variations of non-organic farms *Tukey* $p > 0.05$).

In a comparison of total insecticides per hectare between 2015 and 2016, where only total use or total costs per year were compared. It was shown that in total, in the year of 2015 significantly more insecticides were used on average across all farm types (excluding organic), ($p<0.001$), that costs significantly more per hectare ($p<0.001$). Displayed by the difference in y axis between *figure 5.4a and 5.4b.*

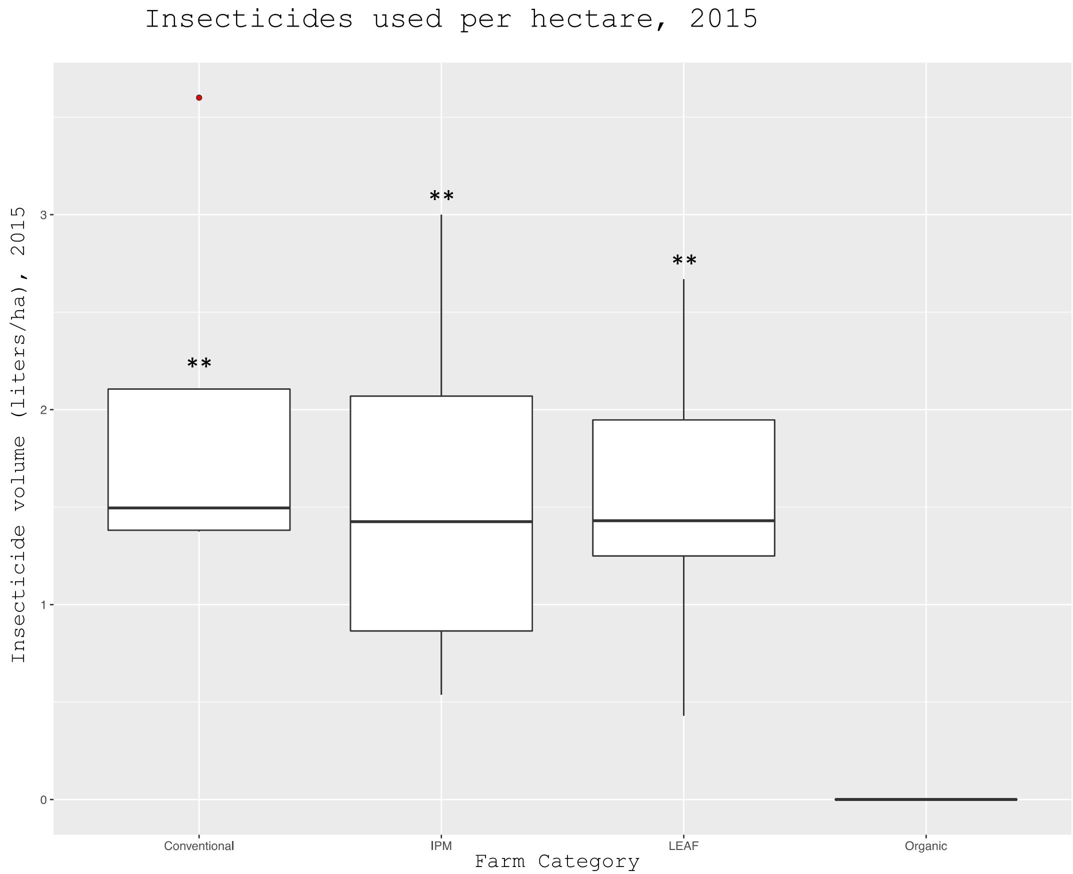


Figure 5.4a Box plot from linear regression model results for insecticide used per hectare on conventional, IPM, LEAF and organic orchards across in 2015. Linear regression and post hoc analysis results show organic orchards are significantly different to all non-organic orchards with the lowest insecticide use (zero), (*p=0.002 for organic – LEAF, p<0.001 for organic – conventional and organic - IPM*). Main significant farm category differences from organic are indicated with "**".

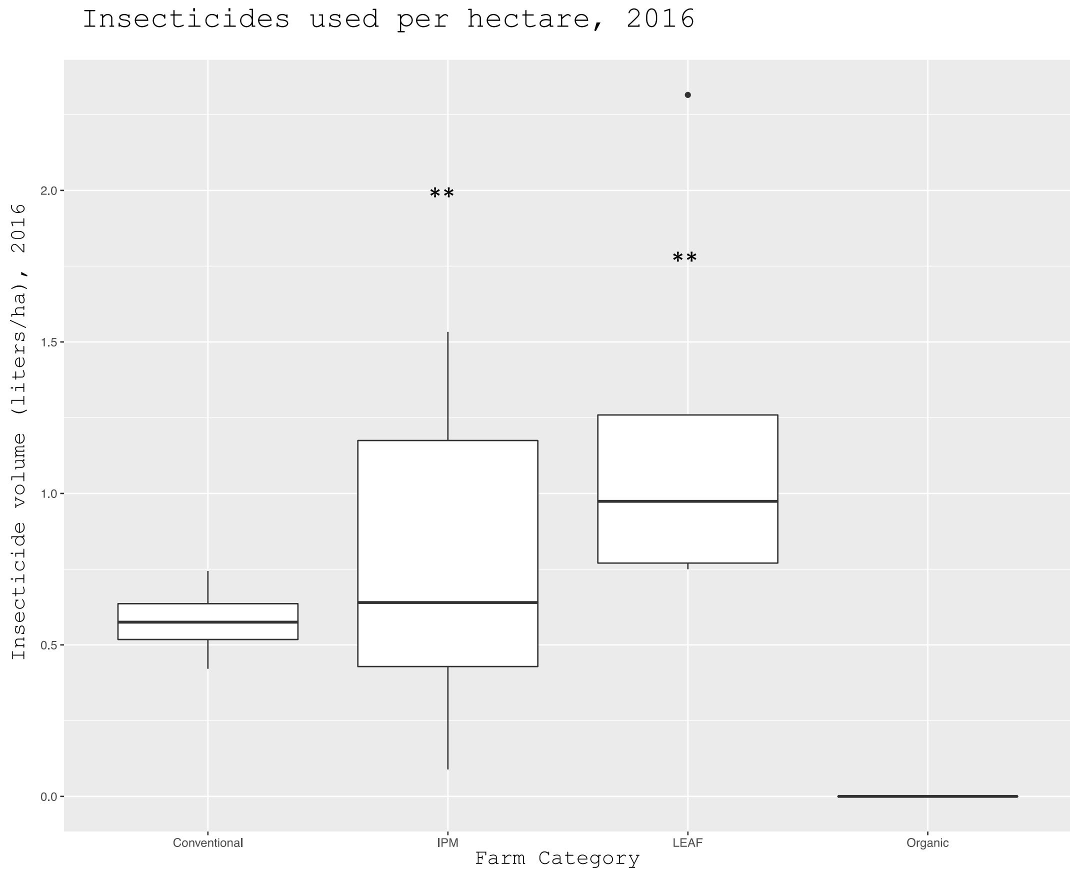


Figure 5.4b Box plot from linear regression model results for insecticide used per hectare on conventional, IPM, LEAF and organic orchards across in 2016. Linear regression and post hoc analysis results show organic orchards are not significantly different to conventional orchards (*p = 0.579*). LEAF and IPM remain significant and use more insecticide than organic (zero) (*p<0.001 for LEAF- organic, p=0.003 for IPM-organic*), even though overall all non-organic farms use less insecticides in 2016 than they did in 2015 (*p<0.001*). Main significant farm category differences from organic are indicated with "**".

Yield and yield value simple linear model overview

Simple linear models show that increased use of insecticides on orchards significantly increased *yield per hectare* in 2015 (*$p < 0.001$, figure 5.5a)* and 2016 *($p = 0.044$, figure 5.5b*), and *yield value per hectare* in 2015 (*$p < 0.001$*, *figure 5.6a* in 2015, but did not increase *yield value* per hectare in 2016 (*$p = 0.058$*, *figure 5.6b*). The same linear models were constructed for total yield and yield value years per tree (*figure 5.7a and 5.7b*), with similar results: as insecticide increases, so does apple yield per tree (*$p < 0.001$*) and yield value per tree (*$p < 0.001$*).

Farm Management Category improves model fit

Using analysis of variance to compare insecticide-only models with farm category as an additional predictor variable, it was shown as significant to keep farm category in the *yield* and *yield value* models per hectare after model comparison. When farm category was added to linear models, it significantly improved model fit for *yield* per hectare (*anova: $F(25,22) = 3.774$, $p = 0.0252$* in 2015; *$F(26,23) = 4.927$, $p = 0.0087$* in 2016) and *yield value* per hectare (*anova*: *$F(25,22)=3.456$, $p<0.001$* in 2015; *$F(25,22)=7.88$, $p<0.001$* in 2016).

The farm management variable also improves model fit when looking at yields and yield value *per tree* in simple linear models (*yield per tree* anova*: $F(554,551)= 32.708$, $p < 0.001$* and *yield value per tree* anova: *$F(554,551)= 37.65$, $p< 0.001$*). This implies that insecticide use is not the only management function that impacts *yield* and *yield value per hectare*

Cider apple yield with farm management and insecticides per hectare

Linear model results (*figure 5.5a and 5.5b*) with farm management type as an additional predictor variable to insecticide show that in 2015 organic and LEAF *yield per hectare* have significantly less yield than conventional (organic *$p = 0.042$; LEAF $p = 0.034$)*, but in 2016 only organic has significantly less yield to conventional (*$p = 0.006$*). Tukey post hoc show organic *yield per hectare* in 2015 and 2016 is significantly less than IPM and conventional (both *Tukey $p<0.001$*) but not to LEAF (*Tukey $p = 0.084$ in 2015; $p = 0.3$ in 2016*).

Cider apple yield value with farm management type and insecticides per hectare

Linear model results (*figure 5.6a and 5.6b*) for *yield value per hectare* in 2015 show only LEAF has significantly less *yield value* to conventional (*p = 0.038**) leaving organic not significantly different to conventional per hectare (*p = 0.088*), however in 2016 organic *yield value per hectare* is the only farm management type significantly less than conventional (*p < 0.001*). Tukey post hoc analysis shows **no difference in *yield value per hectare* between any farm category group in 2015** (*all comparisons Tukey p > 0.05*), but in 2016 organic is significantly less than conventional *yield value per hectare* (*Tukey p = 0.002*) and IPM (*Tukey p = 0.007*), but not significantly different to LEAF (*Tukey p = 0.379*).

Cider apple yield and yield value with farm management type and insecticides per tree

Linear model results with farm management and insecticide per tree for yield and yield value *per tree* results (*figure 5.7a and 5.7b*) show very different results: **organic and IPM farms have a higher *yield per tree*** (*p < 0.001 for IPM and organic*) and ***yield value per tree*** (*p <0.001 for IPM and organic*) **compared to conventional farms**. Tukey post hoc comparisons show the same results: organic farms have higher *yield per tree* than LEAF and conventional (*Tukey p <0.001*), but the same *yield per tree* as IPM (*Tukey p= 0.165*). LEAF farms had less *yield per tree* than IPM (*Tukey p < 0.001*), IPM had more *yield per tree* than conventional (*Tukey p < 0.001),* and no difference in *yield per tree* between LEAF and conventional (*Tukey p = 0.98*).

Yield <u>value</u> per tree shows organic trees hold more *yield value* than LEAF, IPM and conventional trees (organic - *conventional and LEAF: Tukey p <0.001; IPM: p = 0.012*). IPM has more yield value per tree than conventional (*Tukey p<0.001*) and LEAF has less value per tree than IPM (*Tukey p <0.001*). LEAF and conventional value per tree did not significantly differ *(Tukey p = 0.997*).

In the figures below, organic farms (purple) are lined up along the *y* axis at zero insecticide use. Some low yield-producing conventional orchards, namely LEAF farms, produce similar yields to high yielding organic orchards.

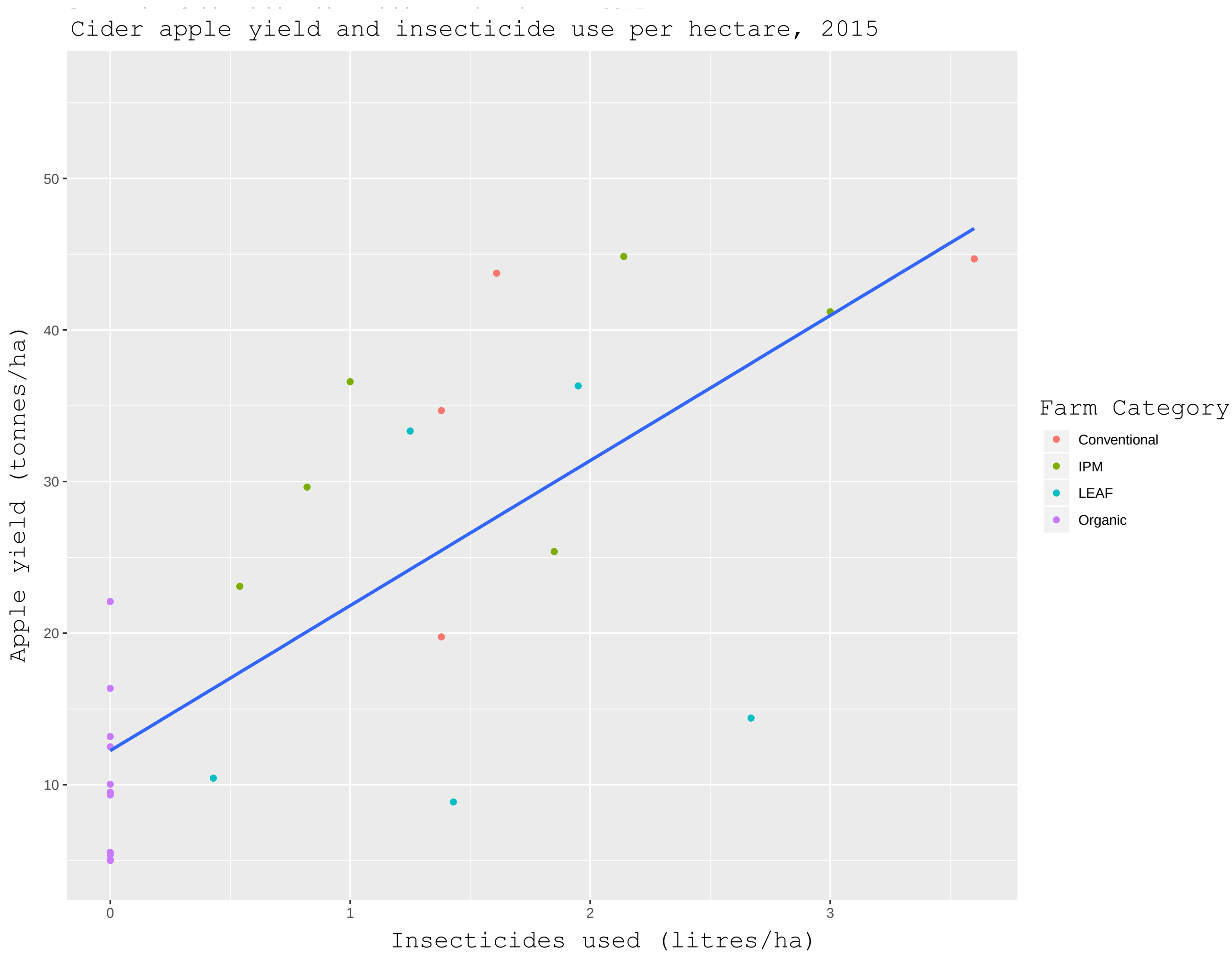


Figure 5.5a – Scatter graph to show linear model results of the relationship between insecticide used in litres per hectare and yield in tonnes per hectare per farm category, in 2015, where increased insecticides increase yield (*p <0.001).* The ab line (method = lm) was used for the regression line. Each point represents a farm and the colour its farm category it belongs to out of conventional, IPM, LEAF and organic)

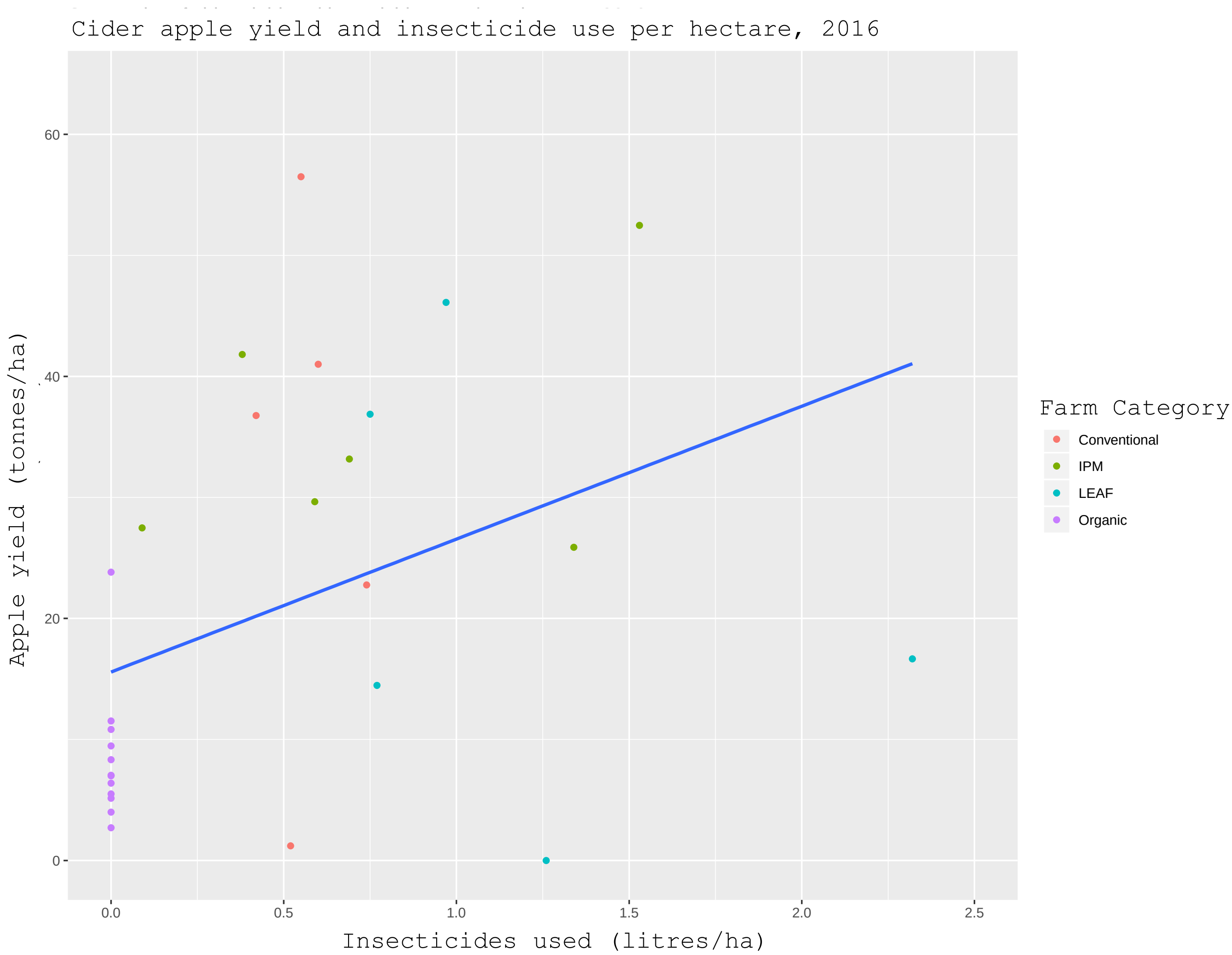


Figure 5.5b Scatter graph to show linear model results of the relationship between insecticide used in litres per hectare and yield in tonnes per hectare per farm category in 2016, where increased insecticides increase yield *(p = 0.044*). The ab line (method = lm) was used for the regression line. Each point represents a farm and the colour its farm category it belongs to out of conventional, IPM, LEAF and organic). Note, one LEAF grower produced no apple cider yield in 2016.

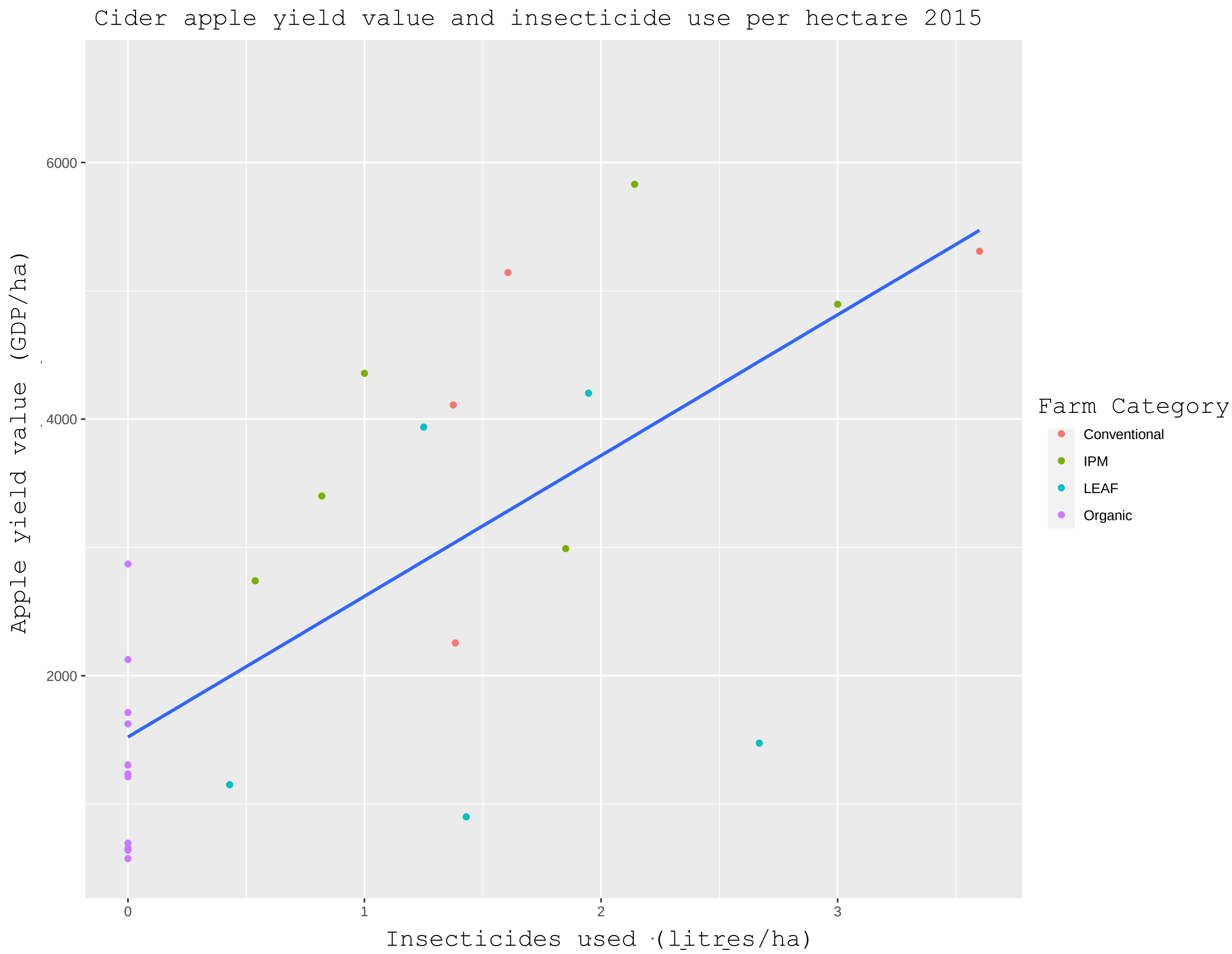


Figure 5.6a – Scatter graph to show linear model results of relationship between insecticide used in litres per hectare and yield value in GDP per hectare per farm category, in 2015, where increased insecticides increase yield *value* (*p <0.001*). The ab line (method = lm) was used for the regression line. Each point represents a farm and the colour its farm management category it belongs to (conventional, IPM, LEAF and organic)

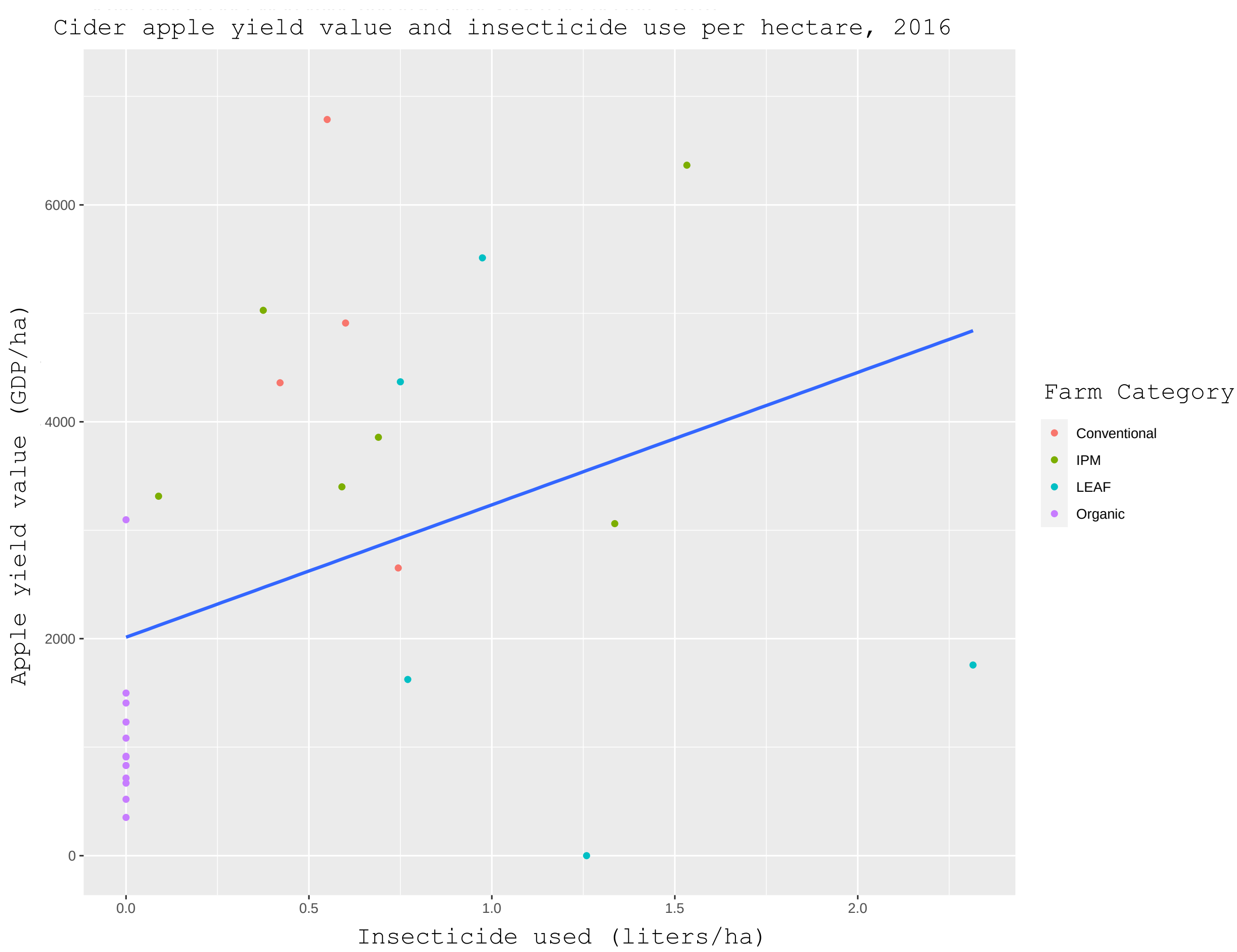


Figure 5.6b Scatter graph to show linear model results of the relationship between insecticide used in litres per hectare and yield value in GDP per hectare per farm category, in 2016, **where increased insecticide <u>does not</u> increase apple yield value (*p = 0.058*).** The ab line (method = lm) was used for the regression line. Each point represents a farm and the colour its farm management category it belongs to (conventional, IPM, LEAF and organic). Note, one LEAF farm had no cider yield in 2016, therefore this yield had no associated yield value.

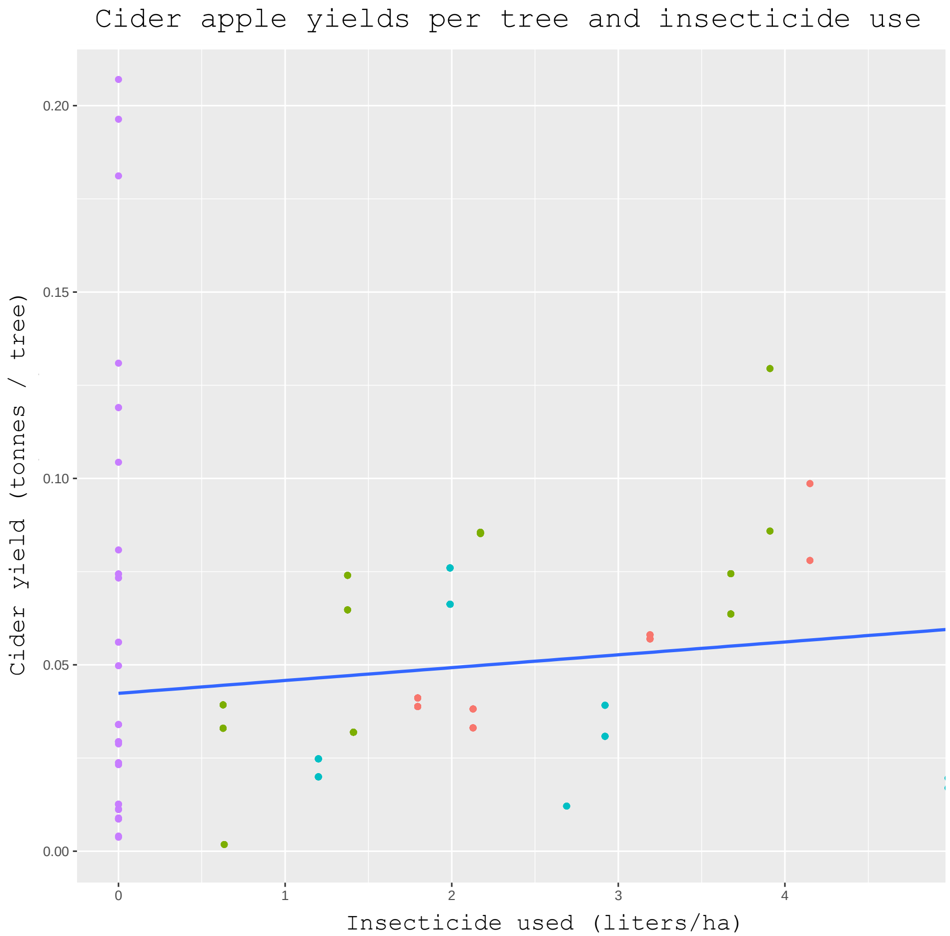


Figure 5.7a Scatter graph to show linear model results of the relationship between insecticide used in litres per hectare and yield of cider apples in tonnes per tree per farm management category across 2015/16. Organic farms had higher yield per tree than LEAF and conventional (*$p< 0.001$*), but the same yield as IPM (*$p=0.165$*). The ab line (method = lm) was used for the regression line. Each point represents a farm and the colour its farm management category it belongs to (conventional, IPM, LEAF and organic).

Figure 5.7b Scatter graph to show linear model results of the relationship between insecticide used in litres per hectare and yield value in GDP per tree per farm management category across 2015/16. Organic farms have more value per tree than LEAF, conventional (*$p < 0.001$*) and IPM trees (*$p = 0.012$*). The ab line (method = lm) was used for the regression line. Each point represents a farm and the colour its farm management category it belongs to (conventional, IPM, LEAF and organic).

5.4.4 Is there any evidence that wild insectivorous birds provide a service to non-organic farming systems?

Generalised linear mixed effect model averages with confidence intervals results have been separated to show the impact of each variable. Within non-organic farms wild birds and insecticides are not significant predictor variables that neither increase or decrease yields and yield value per tree. In many model averages the wild bird community variable is not chosen to be kept within top models, therefore was not included in the model average results. Where wild bird community variables do appear in model averages, all of them have confidence intervals that cross zero, making these insignificant explanatory variables to the yield and yield value per tree. In all model variations, increased insecticide use is not significant in creating higher yield or yield value per tree. Sections 5.5.3.1 to 5.5.3.6 display non-organic farm model average results. These model average results exclude organic farms to allow differences between non-organic orchards themselves to be shown.

5.4.4.1 Cider apple yield per tree with insectivore abundance

There was no model average created for this due to only one model being chosen from top model selection process. Instead, these are mixed effect model results with confidence intervals using the top model chosen from model averaging. From *figure 5.8,* only orchard size is significant, where the larger the orchard the less cider apple yield per tree. With organic farms excluded from the model, IPM and insecticide use does not show to be significant in impacting apple yields per tree. **This means that between non-organic farms themselves, a higher rate of insecticide spray <u>does not</u> increase cider apple yields or yield value per tree**. Although insectivore abundance is included in this model, it too is not significant in impacting apple yields, positively or negatively.

5.4.4.2 Cider apple yield value per tree with insectivore abundance

The model average results for apple cider yield value results in *figure 5.9* show the same trend as yield. Only orchard size shows a significant negative relationship, and all other independent variable have no significant impact on cider apple yield value per tree, including insecticide use and farm management.

Here can be seen that insecticide abundance has been removed from the model average during the top model selection process, therefore does not appear in the results.

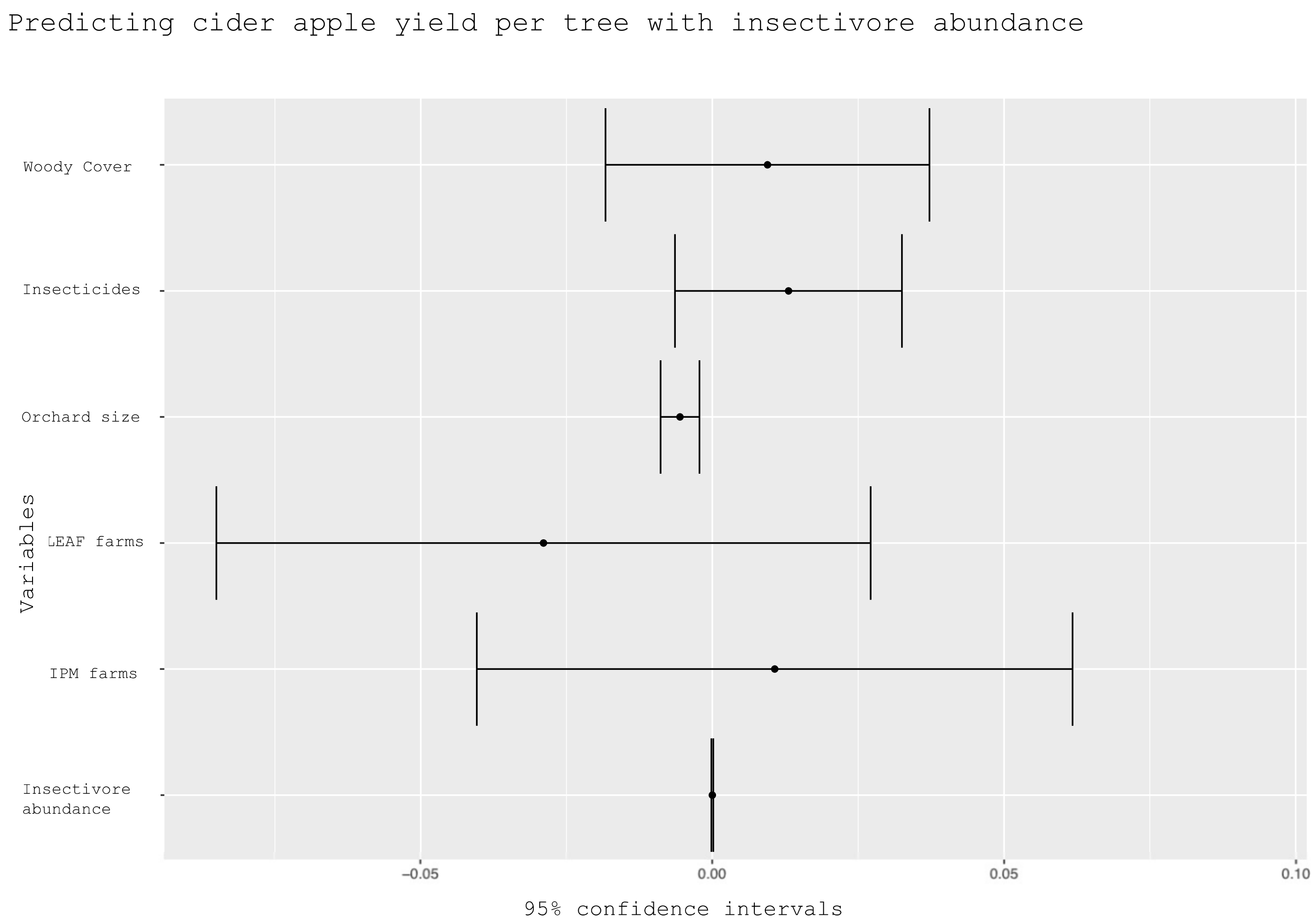


Figure 5.8 – GLMM top model results and confidence intervals for predicting yield with insectivore abundance, without organic farms and with 95% confidence intervals. Only one model was found significant therefore no model average needed. Full model results table can be found in *Appendix O.*

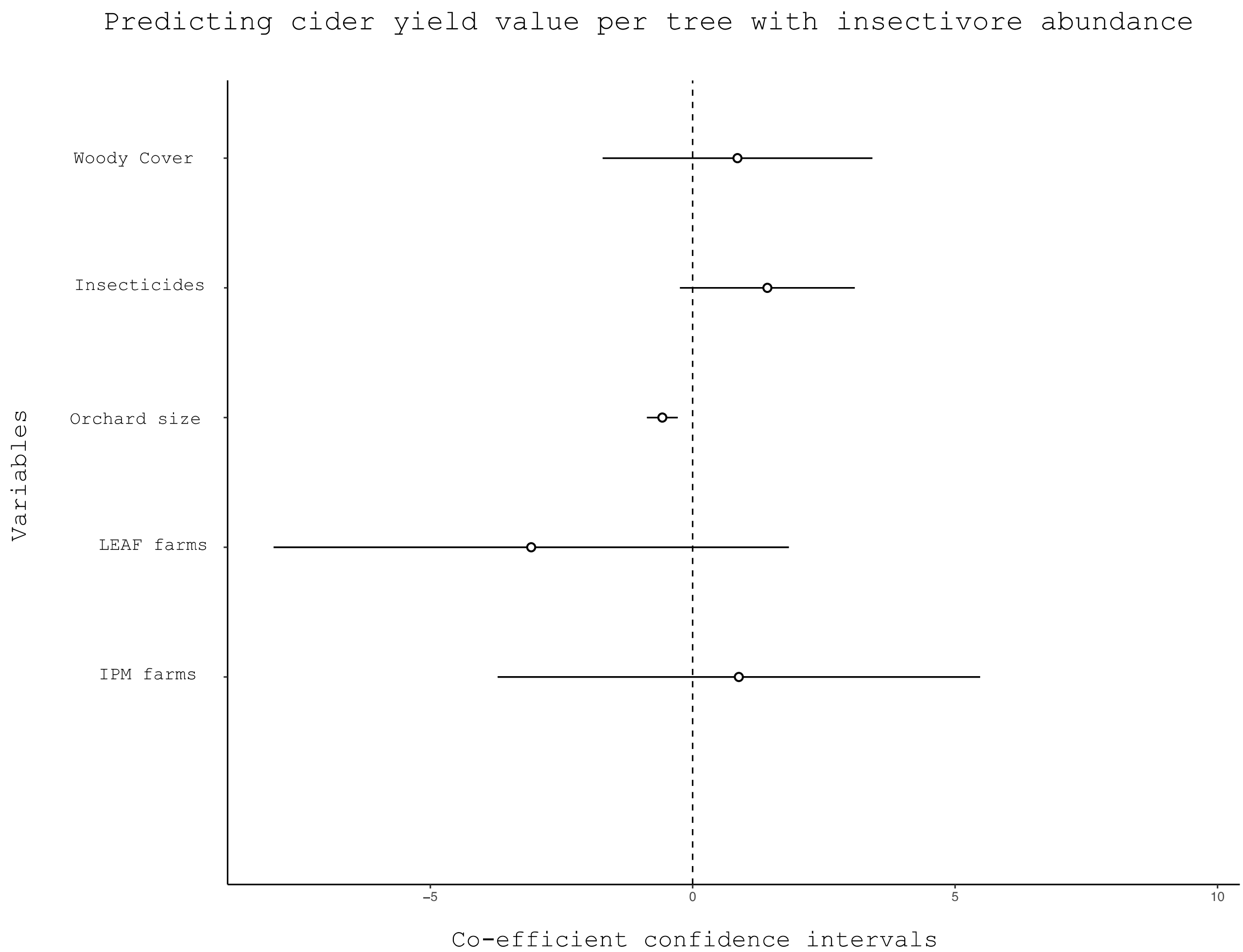


Figure 5.9 – GLMM Model average confidence intervals without organic farms using insectivore abundance to predict yield value. These are 'conditional' model average results, using top models, Delta < 6. Explanatory variables on y axis are on-farm management and farm characteristics. Insectivore abundance <u>did not</u> feature in top models. The model intercept is conventional farming. 'Full' model average is found in *Appendix O.*

5.4.4.3 Cider apple yield per tree with insectivore species richness

This mixed effect model result with confidence intervals used the top model chosen from model selection process of apple yield and insectivore species richness, due to only one top model being given in model selection process. Only orchard size has a significant, negative, effect on cider apple yields per tree (*figure 5.10*). IPM and insecticide use does not show to be significant in impacting apple yields per tree, again. All farm management types are insignificant, as is woody cover. Insectivore abundance is included in this model but is not positively or negatively significant to apple yields.

5.4.3.4 Cider apple yield value per tree with insectivore species richness

The model average results shown in *figure 5.11* shows orchard size is the only significant variable that negatively effects cider apple yields per tree. All other variables did not have a significant effect on cider apple yield value per tree, including insecticide use or other non-organic farm managements. Insectivore abundance has not been included in any top models and therefore the model averaging process has not included this variable in the results.

Predicting cider apple yield per tree with insectivore species richness

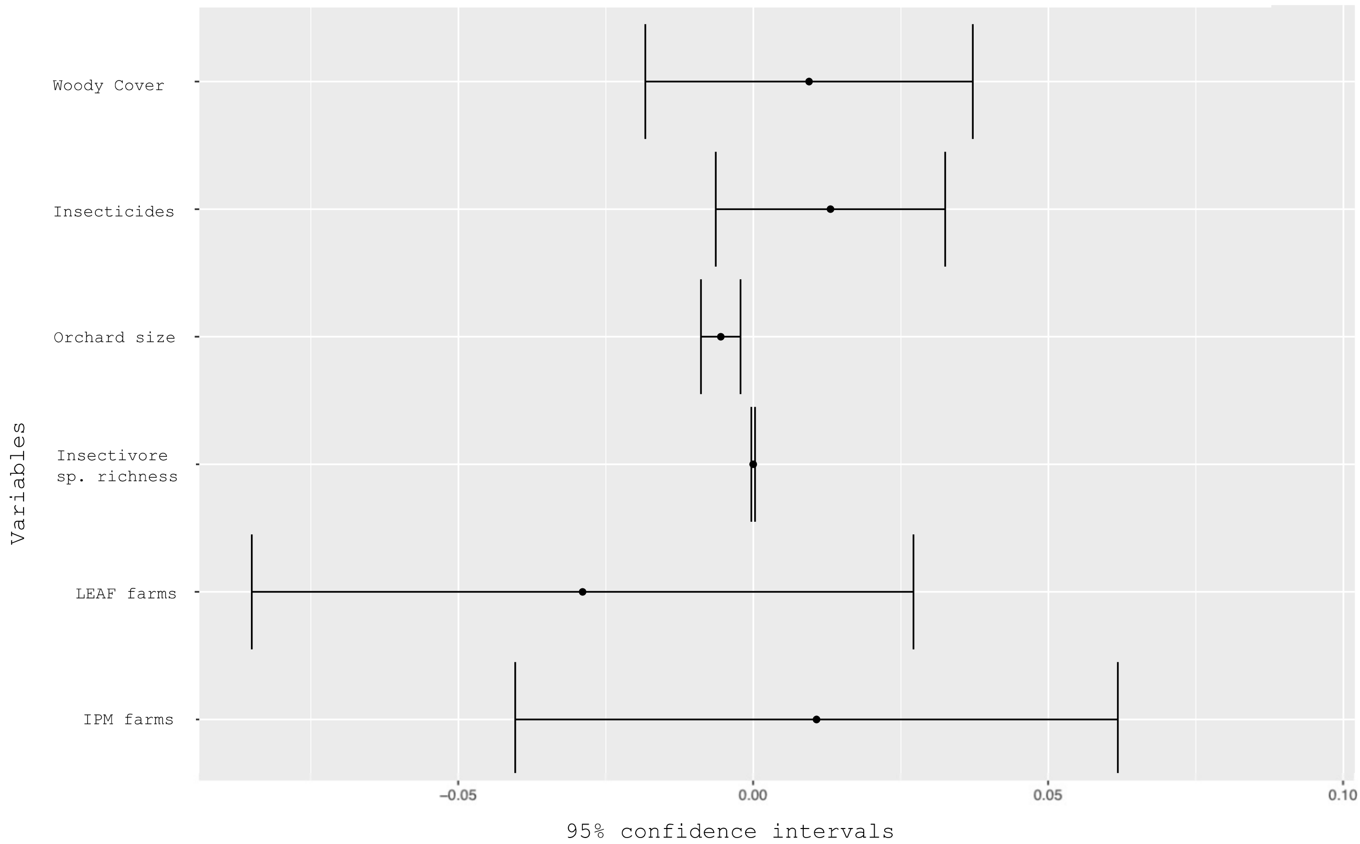


Figure 5.10 – GLMM top model results with confidence intervals for predicting yield with insectivore sp. richness, without organic farms and with 95% confidence intervals. Only one model was found significant therefore no model average needed. Full model results table can be found in *Appendix O.*

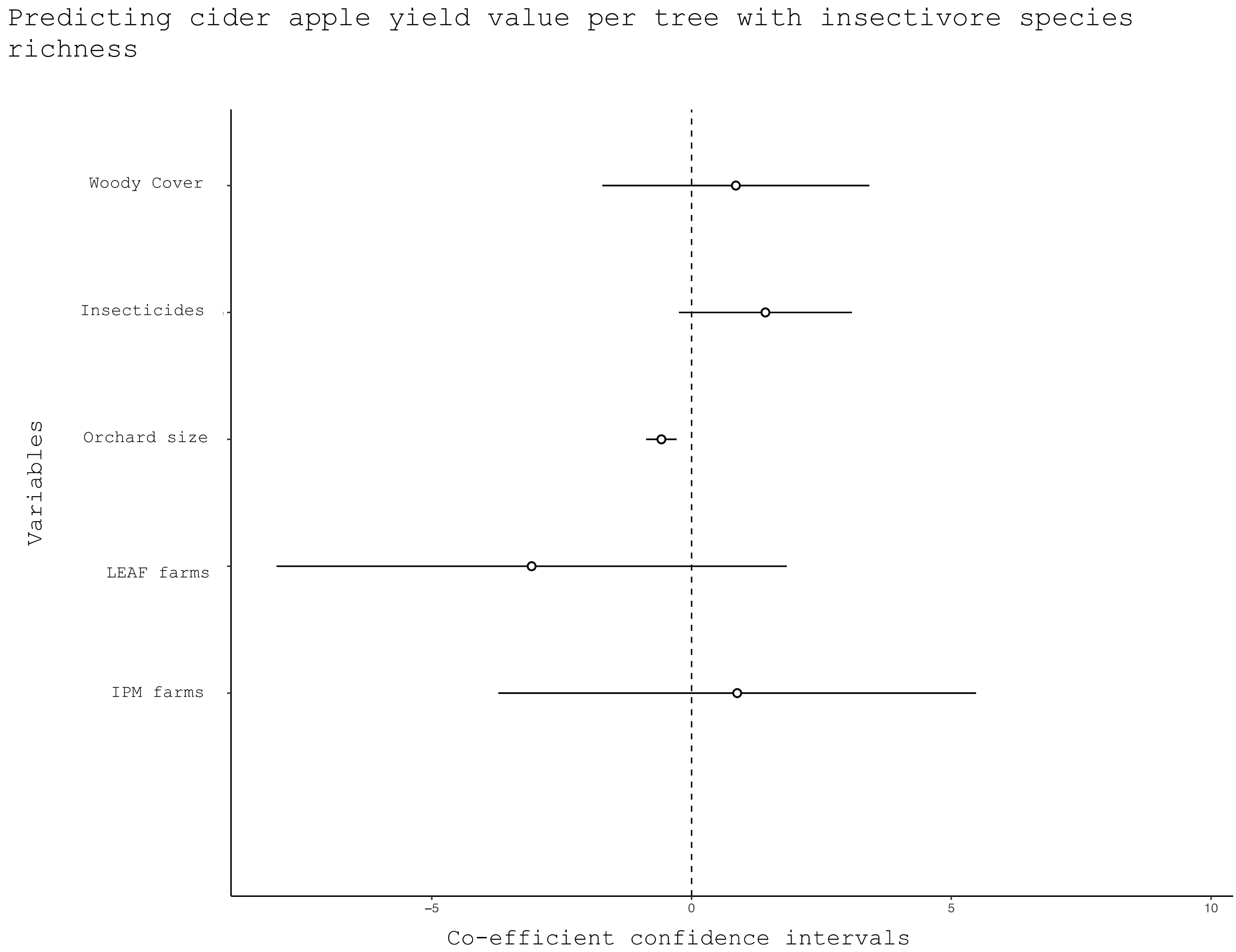


Figure 5.11 – GLMM Model average confidence intervals without organic farms using insectivore sp. richness to predict yield value. These are 'conditional' model average results, using top models, Delta < 6. Explanatory variables on y axis are on-farm management and farm characteristics. Species richness was <u>not</u> included in any top models, therefore not reported here. The model intercept is conventional farming. 'Full' model average is found in *Appendix O.*

5.4.4.5 Cider apple yield per tree with insectivore diversity

These mixed model results with confidence intervals are from the top model chosen from model selection process, *Figure 5.12* shows the model results with 95% confidence intervals that orchard size is negatively significant to cider apple yield per tree. All other variables did not have a significant effect on cider apple yield, including insecticides and other non-organic farm managements. Although insectivore diversity has been included here, it is also not significant in predicting yield. This graph is from the full lmer model results rather than model average due to there being only one top model to choose from, thus no average was able to be taken.

5.4.4.6 Cider apple yield value per tree with insectivore diversity

Figure 5.13 is the model average results using insectivore diversity within non-organic farms. Only orchard size is shown as a significant predictor variable, the larger the orchard the lower the yield value per tree. Although, diversity is including in the top models and model average here, it is still not significant in predicting apple cider yield value. Non-organic farm managements, Woody Cover and insecticide use all are not significant to apple yield value.

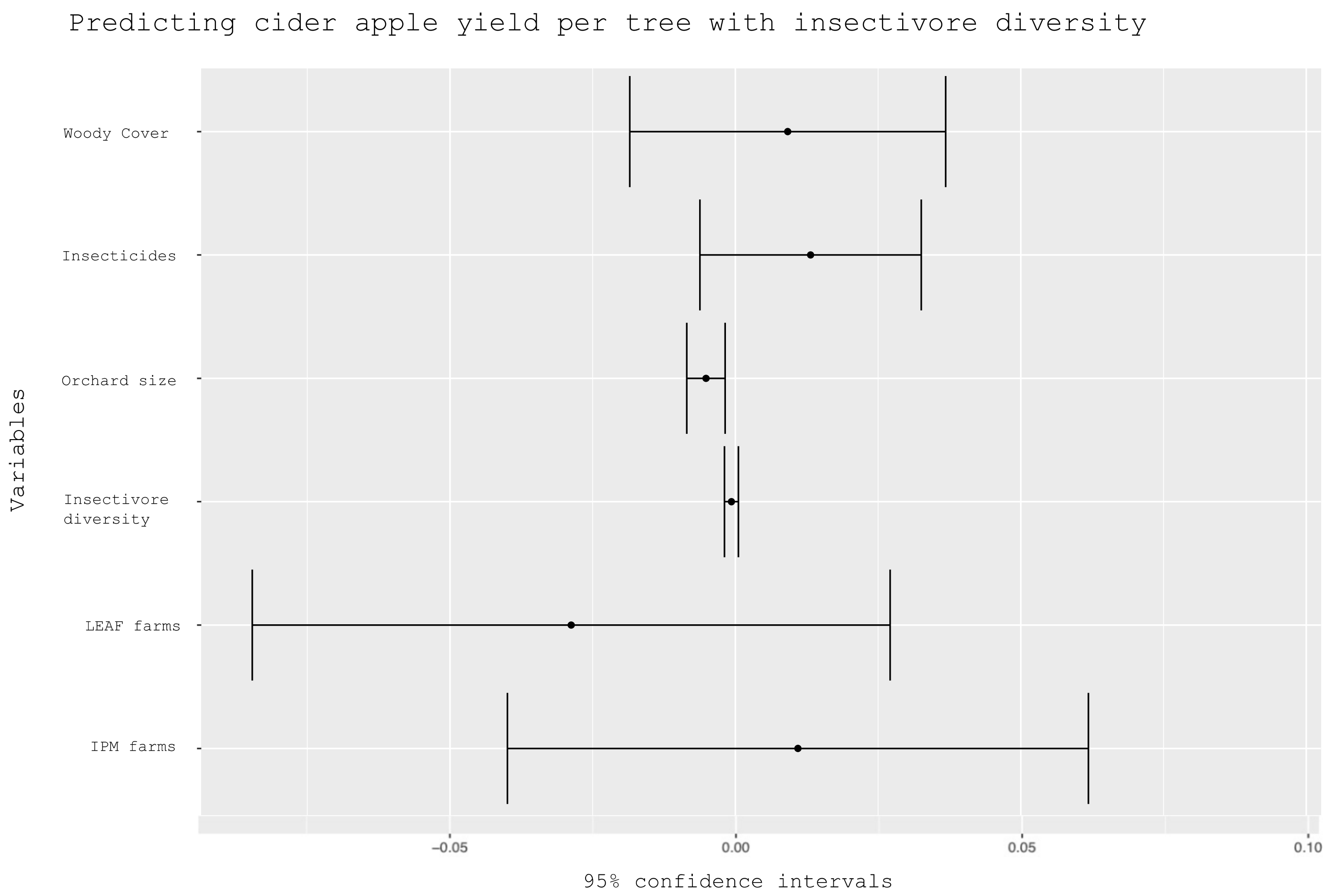


Figure 5.12 – GLMM top model results with confidence intervals for predicting yield with insectivore diversity, without organic farms and with 95% confidence intervals. Only one model was found significant therefore no model average needed. Full model results table can be found in *Appendix O.*

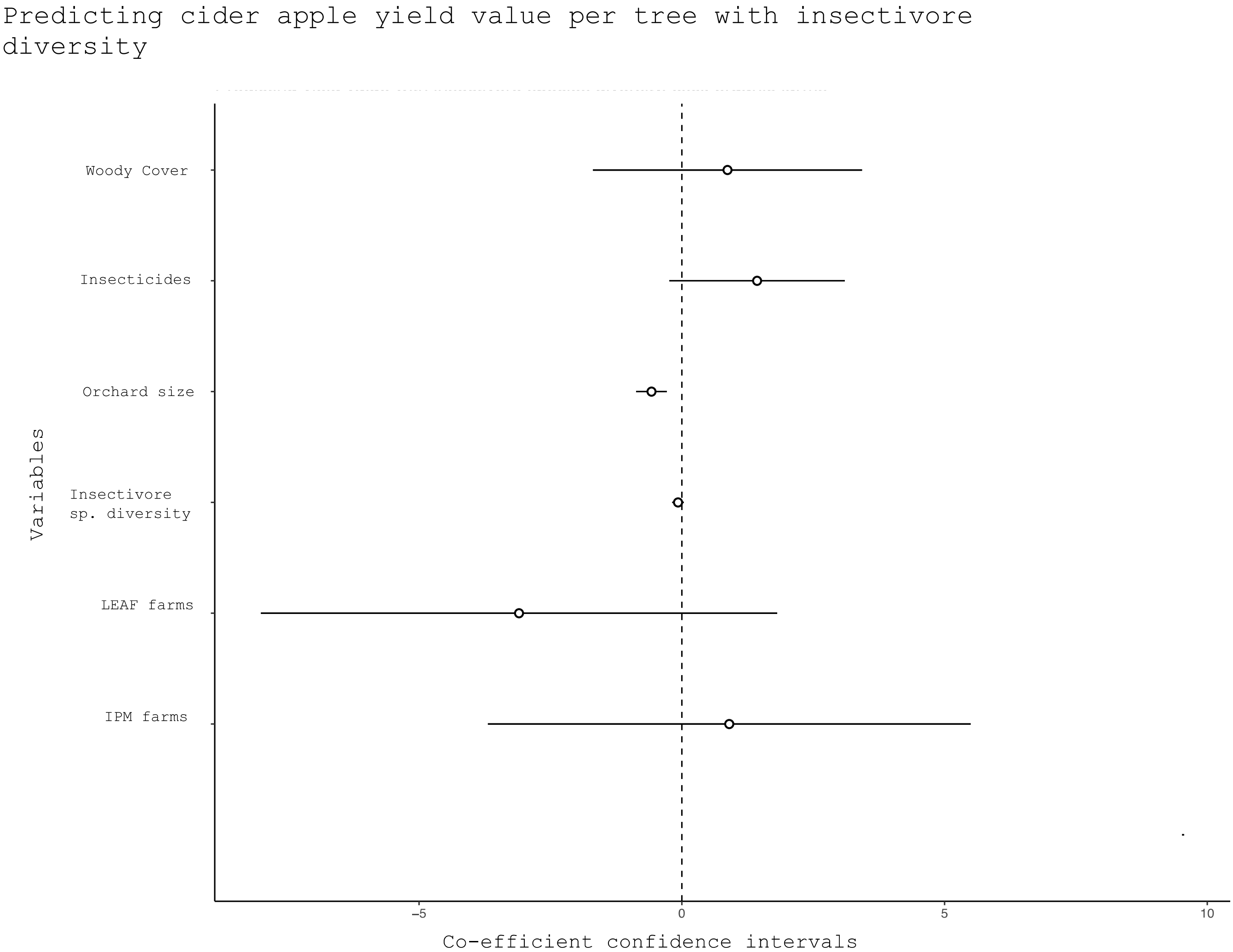


Figure 5.13 – GLMM Model average confidence intervals without organic farms using insectivore diversity to predict yield value. These are 'conditional' model average results, using top models, Delta < 6. Explanatory variables on y axis are on-farm management and farm characteristics. The model intercept is conventional farming. 'Full' model average is found in *Appendix O.*

5.5 DISCUSSION

In this chapter organic farms have been used as a predation proxy during analysis to understand value, in terms of yield and yield value, that biodiversity has on organic orchards. Valuing the pest control ecosystem service that exists on organic orchards and comparing this to the synthetic alternative of pest control used on non-organic farms, is the focus within the chapter. Farms that have higher bird biodiversity, support a high functional diversity of insectivorous birds (Chapter 3), which predate more economically important moth pests than farms that do not support bird functional diversity (Chapter 4) and the impact on how this translates to a value is discovered through this chapter.

To summarise the results of this chapter I have focussed on answering the three questions posed in the objectives of study here:

1) Per hectare, yields on organic farms are lower than conventional and IPM farms, but not different to LEAF.

   Per tree, organic has the same or higher yields than non-organic farms. Organic yield value per tree is higher on organic orchards than all non-organic farms, especially when considering increased use of insecticides in 2015.

2) Chemical use is positively correlated to yields but does not explain all the differences in yield and yield value between farm management types, where farm management improved model fit in both per hectare and per tree models.

   Organic yield value does not differ from all non-organic farming systems in 2015 (due to the costs of larger volumes of insecticides). In 2016, non-organic farms used less insecticides, which meant yield value per hectare of organic becomes significantly less than IPM and conventional, but not to LEAF.

3) There is no evidence to suggest that wild bird affects yield and yield value on non-organic farms, positively or negatively.

This chapter shows that organic farms, that rely on wild birds as their pest control and use no chemical inputs to produce their yields, have statistically lower apple yield per hectare than non-organic systems that use chemical inputs to replace wild bird ecosystem services, except for LEAF farms. This explains the continued reliance on chemical alternatives to natural pest control from birds in UK horticulture because the yield benefits of wild birds are smaller than can be achieved using non-organic farming practices. Furthermore, within non-organic farms themselves, there is no evidence of yield and yield value impacts from wild birds, showing that conventional and IPM management is more important in driving yields and yield value than ecosystem services from nature.

Conversely and importantly, within non-organic farms, insecticide use is not positively significant to yield but does become negatively significant to yield value, where increased insecticide use on non-organic farms reduces yield value per hectare, not dissimilar to yield value of organic. This shows that although insecticides play a role in yield production between organic and non-organic farms, there are other farm management factors at play that are more important to yield and yield value besides insecticide applications: an extremely important result for highlighting that complete chemical reliance in agriculture is not as beneficial as non-organic farms and wider society may perceive.

5.5.1 Cider apple yields increase with non-organic farming per hectare, but not with LEAF; whilst yield value is higher on organic than conventional in 2015.

Importantly, yield per tree does not differ between farming systems, where organic yield per tree is the same as IPM and conventional yields *(Results section 5.4.2)*, but higher yields than conventional when insecticide analysis is included and higher than LEAF orchards with or without insecticides included. Furthermore, organic cider apple yield value is higher, per tree, than all non-organic farm management types with insecticide analysis (*Results section 5.4.3).* The reason for these increases in yield and yield value of organic trees is due to several reasons. Firstly, tree density is significantly lower on organic orchards (*Results section 5.4.1)*, with a more traditional orchard structure that hosts larger spacing between trees and permanent grass swords, usually grazed by livestock, and older trees able to reach veteran stage (Burrough et al., 2010; Pantera et al., 2018). Secondly, veteran trees are stereotypically much taller and wider than non-organic orchard trees, which are usually placed on root stocks to limit tree growth to make maintenance and chemical treatment more accessible on non-organic farms. Therefore, veteran trees can host higher abundances of apples per tree but need more space to do so. Thirdly, the value per tree is shown to be higher due to the reduced chemical inputs of organic management and a higher price of organic apples compared to conventional (organic market price is £130 compared to conventional at £120 in 2015 and £122 in 2016 per tonne, *section 5.3.2*). Similar increases in yield have been shown by Tschumi et al. (2016), who found a 10% increase in crop yield when biodiversity-mediated ecosystem services increased on farm. But like Albrecht et al.

(2020), impacts to yields from enhanced species community metrics, such as that of organic farms, is variable - which is highlighted through this discussion.

Although this analysis between farm management types per tree allows for a fair comparison, farm yield is taken across the whole farm on a hectare-by-hectare basis and not per tree. To make this chapter's interpretation applicable to the farming community, the remaining discussion will focus on a per hectare basis as yield and yield value is the final ecosystem service in question, which is per hectare rather than per tree. However, it is important to keep in mind that it is expected that organic orchards will have lower yields at the orchard level, or per hectare, because of their characteristic low tree density described here.

Cider apple yields per hectare increase significantly in this study with non-organic farming, except LEAF, showing that pest control strategies on organic orchards from the natural pest control of birds are not as productive as non-organic systems, also supported by other studies (Samnegård et al., 2019; Seufert et al., 2012; Bengtsson, 2015; Crowder and Reganold, 2015; Gabriel et al., 2013; Mäder et al., 2002; Jouzi et al., 2017). Conventional and IPM farming systems, that do not rely on a natural pest control system (Chapter 4), provide more yield to farmers than a natural pest control ecosystem service available on organic farms. Seufert et al. (2012) found that organic fruits were at similar yield levels to conventional farming and did not significantly differ. Although this contradicts the findings here per hectare from conventional and IPM that produce significantly higher yields than organic, it resonates with the results from LEAF farms in this study, that do not differ from organic yields per hectare (*Results section 5.4.1*).

When looking at farms individually, four non-organic farms had lower average yields and yield value per hectare than some of the more productive organic farms (*figures 5.5a,b and 5.6a,b)*. These non-organic farms lost considerable profit due to the additional human inputs used, especially in 2015, with a low crop to sell. These results are like Samnegård et al. (2019), who found that organic orchards have 48% less yield than IPM but also saw that some high-yielding organic farms had higher yields than the average IPM orchards. This may become a more likely future scenario due to insecticide resistance (LEAF 2017; Luck et al. 2009), crop failures with intense weather scenarios as seen by the floods in Herefordshire in 2019, and the nature of economic market fluctuations.

The difference between organic with IPM and conventional yields per hectare was an expected result, which is why premiums are assigned to products managed through organic certification schemes. However, LEAF farms were not expected to have similar yield produced as organic per hectare, especially due to the lower tree density of organic orchards (*Results section 5.4.1*). LEAF uses an integrated farm management approach (IFM) to sustainably manage farms over the whole farm to economically support farms whilst also being concerned and aware about pressures on their environment (LEAF, 2016). LEAF farms do not receive a premium for their products sold to market, so lower yields are not compensated for. Although LEAF advertise sustainability, this may not be sustainable for farmers economically, where organic may actually provide more financial incentives, as well as environmental, considering the evidence that supports the practice of organic farming to sustain and protect farmland biodiversity (Tuck et al., 2014), not seen by LEAF farming in this study (Chapter 3). Apple yields from organic growers are either sold at higher price premiums on to the organic market, or they are used as production of artisanal cider on small scale farms (Marsden, J. *Pers. Comms*, 2018). This organic price premium is evident in 2015 where insecticide use is higher than in 2016 *(Results, 5.4.3),* which means higher costs per hectare for non-organic farms. In 2015 the yield value on organic does not differ per hectare to all non-organic farms – showing the costs of insecticides used were too high and brought the value of non-organic farming in line with organic, even though organic yields and tree density are significantly less per hectare. When insecticide use was less in total in 2016, this makes organic value per hectare become less than non-organic orchards, yet LEAF remains at a lower value than organic. This could be due to a higher use of insecticides in 2016 when other non-organic farms used less (*figure 5.4b*), although not significantly different.

The underperformance of LEAF farms was unexpected and has not been researched independently before now. Reed et al. (2017) interviewed LEAF farmers to understand their perceptions of LEAF and how it benefits the farmer, with no mention of yield increase. The access to market is highlighted throughout LEAF memberships but LEAF does not seem to benefit the farmer through delivering "prosperous farming" through IFM approaches (Reed et al., 2017: 9), as this chapter has highlighted. Farmers pay considerable amounts of money to be part of the LEAF Marque scheme but the only benefit seems to be the access to shop floors of larger retailers with 97% of participants in the study reporting this, yet only 23% saw a price premium attached to this (Reed et al., 2017). Reed et al. (2017) is the only published document (at time of writing) that assesses the impacts of LEAF marque farming. However, this document is published on behalf of LEAF themselves and was based solely on farmer interviews who are LEAF demonstration farmers. Business strategy and engagement are key skills that LEAF farms declare they have developed through the scheme, and savings may have been made through efficacy and operational changes rather than yield increase or yield value increased through reduction in insecticide inputs.

5.5.2 Insecticides increase yields but not yield value, and not within non-organic orchards

When comparing organic farms with non-organic, there is a direct correlation with the amount of insecticide used and the apple yield gained per hectare showing that when more insecticides are added higher yields per hectare are observed than birds can do on their own under a natural pest control scenario (*Results section 5.4.3)*. This helps in our understanding of why insecticides continue to be used as the pest control service on non-organic apple orchards, even though birds provide this natural predation service on organic farms, they are not as effective as the chemical alternative. However, when comparing within the mixed models using more predictor variables and without organic orchards in the dataset, there is another key finding; within non-organic farming the use of insecticide does not play a determining factor in yields per tree (*Results section 5.4.4*). Thus, there are other management factors at play on non-organic farms that are not included in this analysis, of which have more impact on yields than insecticides do. These are likely to be other human inputs, such as fertiliser use (Garratt et al., 2011) and herbicides (Bengtsson, 2015). Although the use of insecticides is a major distinguishing factor in each type of management used, and one of the main influences determining crop yield from Bengtsson (2015) study, analysis here has shown that farm category significantly improves the model fit during analysis of variance of linear models (*figure 5.5a – 5.7b*), so other management practices are more important than increased insecticide use.

The difference in yield value per hectare seen between years is important to contextualise here. In 2015, yield value per hectare of all non-organic farms were not significantly different to organic apple yield value per hectare. The reason for this similarity is due to increased use and cost of insecticides in 2015 in comparison to 2016 (*figure 5.4b*). This finding shows that by taking into account the increased value of organic produce sold to market and the cost of insecticide use of non-organic farms, similar yield values between organic and non-organic farming are discovered. Crowder & Reganold, (2015) show comparable results: although yields were lower on organic fields, financial gains were substantial and were significantly profitable when organic premiums were applied.

Organic orchards sprayed significantly less insecticides (zero) than all other farm categories in 2015, however this was not the same for 2016 where there was no significant difference between organic with conventionally management orchards. IPM and LEAF still sprayed significantly higher amounts than organic in 2016 (*figure 5.4a and 5.4b*), albeit lower than they sprayed in 2015. The reason for the reduced use of insecticides by all non-organic farms, especially conventional, in 2016 was likely due lack of the previously widely available, broad-spectrum insecticide chemical called Chlorpyrifos. Chlorpyrifos was banned in the UK after 2015 harvest (HSE, 2016) due to worries surrounding the negative impacts on human health it may cause users. This change in chemical insecticide availability influenced farmer decision making, leading to non-organic farms spraying less chemicals in total in 2016 (with conventional spray at similar levels of organic). The significantly reduced insecticide inputs in 2016 by conventional farms causes the value of conventional yields per hectare to change, from being comparable to yield value of organic in 2015 to significantly higher value than organic in 2016, even though mean apple yields decrease slightly from 39.5 tonnes in 2015 to 37 tonnes per hectare in 2016 (*figure 5.1a and 5.1b*).

Non-organic farmers have shown here how they have adapted their behaviour in response to volatile chemical markets, to use lower levels of chemicals with the risk of reducing yields, to save money from production costs and keep yield value high. This highlights the impact that using chemicals has and the costs of chemicals to farmers, where organic yield value remains less volatile and more stable between years. Increased insecticides use does not increase apple yield value per hectare, as well as having no significant impact on apple yields or yield value per tree (*Results section 5.4.3; 5.4.4).* This connects to the second stage of sustainable intensification theory, "Substitution", outlined by Pretty et al. (2018), where co-production of agricultural and environmental outcomes can be achieved. With less chemical inputs on non-organic farms achieving higher yield value than organic, shows that sustainable intensification may be well received and achieved by the non-organic farming community. It would ensure yield value remains stable due to reduced volatility of insecticide price increases or prohibitions. Nevertheless, wild bird communities were significantly lower across all non-organic orchards (Chapter 3), showing that although reduced insecticides may benefit agricultural profit outcomes, it didn't benefit biodiversity on non-organic farms in that year, but over a longer period this may change (Kleijn et al., 2019; Bullock et al., 2011).

Economically, there is no reasoning for farmers continuing to rely on chemicals; within non-organic management of orchards themselves more insecticide use is *not* better. Yield value is proven to be similar between organic and non-organic in 2015 and an increase in yield might not necessarily be a future aim for some growers, especially with a view on future provision uncertainty of pesticides (Hillocks, 2012; HSE, 2016). Over-reliance of the use of chemical insecticides (and fertilisers, Garratt et al., 2011) in orchards negatively affects natural enemies and pest control providers, such as birds (Chapter 4; Fountain & Harris, 2015; Wilson & Tisdell, 2000), that provide valuable pest control ecosystem services to growers (Cross et al., 2015). Long term reliance on chemical input is not sustainable in terms of pest resilience increases, where increased resistance from pests to chemicals over time leads to decreased agricultural resilience (Lamichhane, 2017; Bengtsson, 2015; Luck et al., 2009; Lewis et al., 1997). The dependent use of insecticides is not ecologically beneficial in the short term either, due to chemical leaching and the negative impacts to the ecosystems such as pollution, eutrophication, and ecosystem service deterioration (Bengtsson, 2015).

Increased costs of chemicals, low rates of apple prices per tonne and potential threats of pest resistance (Wilson & Tisdell, 2000), suggest savings from reduced farm inputs may begin to overtake the importance of high yielding crops, as shown by the decisions of non-organic farmers in this study. The less volatile route of organic farming may become ever more appealing as chemical volatility increases in the future.

Premium prices are paid for organic produce to compensate for yield gaps and as shown in this study, they do compensate and can provide similar yield value per hectare as non-organic when insecticide-use is high. To keep this result from being only relevant when insecticide use is high, organic premiums need to be more substantial to support organic farmer competitiveness in multiple scenarios. Organic premiums should be able to compensate farmers for the ecosystem services provided, environmental outcomes and the avoided environmental damage in compensation for producing less yield (Reganold & Wachter, 2016; Pretty et al., 2018). To reduce yield value gaps between organic and non-organic farming additional support to organic farmers through grants is continually needed (García et al., 2021), and found here to be meaningful.

5.5.3 Wild insectivorous bird communities do not provide a service to farmers through influencing apple yields or yield value on non-organic orchards.

No significant evidence was found to suggest that increased presence of wild insectivorous bird communities play a role in increasing apple yields or yield values of non-organic farms, in fact no impact was found either positive or negative (*Results section 5.5.4)*. Unlike Chapter 4, where we see there is a pest control service available to organic farmers from birds, in this section non-organic farms are assessed separately and organic is removed from analysis as discussed in *Data Analysis section 5.3.5* due to fundamental differences between organic and non-organic other than wild bird community metrics, which includes the significant differences in tree density (Results section 5.4.1). Analysis shows the use of birds as an indicator for apple yields and yield value is not significant and has no yield or yield value service to non-organic farmers. These results contradict a similar study by García et al. (2021), who found a connection between increased insectivorous bird abundances within orchards through ecological intensification practices and apple pest control, which reduced apple damage and ultimately lead to increased apple yields (García et al., 2021; Samnegård et al., 2019). Furthermore, findings such as those in this section, draw conclusions alike to Samnegård et al. (2019), where increasing natural pest control species may not always be a positive influence on yield outcomes, but if it causes no negative impacts to yield then having a more targeted pest control strategy (specific insecticides over the more widely used broad spectrum) will not detriment yield and *will* be more environmentally friendly. Thereby, encouraging insectivorous birds on non-organic orchards will be a positive impact for the environment on orchards, without limiting apple yields and profits. In addition to environmental benefits, there may be long terms benefits noticed by non-organic orchards if insectivore abundances are able to increase to that of organic levels. Mols & Visser (2007) found that IPM farmers can benefit from insectivore presence without changing their management practice, as great tits are reported to reduce caterpillars, even at low densities. The authors hypothesise that due to the services of increased great tit abundances, spraying insecticides later in the season may be obtainable as the insectivore bird species is able to replace the use of synthetic pest control, currently used against low pest densities.

Relating wild bird findings to agricultural inputs, as Seufert et al. (2012) concludes, has contextual differences in yield value where different outcomes are shown when assessing wild insectivorous birds' impact on apple yield value provision, using organic farms as this proxy. The variations in results are dependent on circumstances such as high yielding organic farms and the amount of insecticides used. As discussed, when synthetic pest control is low on non-organic orchards, as they were in 2016, the replacement of synthetic insecticides by wild birds is not valuable, in terms of yield or yield value, and farmers will find it difficult to switch to a less intense system based on economic and financial decision making (*figure 5.6b*). This finding resonates with Bengtsson (2015) and presents worrying results in terms of the case for biodiversity conservation, where the services from insectivorous birds could be put under economic scrutiny. Yet, when synthetic pest control is high, as in 2015, organic yield value is not different from all non-organic orchards per hectare which shows that a wild bird pest control service can perform as well as some of its synthetic replacements, in terms of yield value, when synthetics are used more intensively (*figure 5.6a*). This variation of the results is in line with findings drawn from Pywell et al. (2015), who found that on arable farms with higher ecosystem service provision, higher crop yields were produced, and farmers are likely to see benefits from organic farming.

Although direct positive impacts to yields from the wild bird community itself on non-organic farms are seen in this analysis, only one ecosystem service was assessed, similar to Bengtsson (2015). There are multiple ecosystem services at play, such as pollination that impacts yield and apple quality (Garratt et al., 2014) and soil nutrients that affects yield to an equal extent to pest control (Bommarco, Kleijn, & Potts, 2013), that were not taken into consideration in this study and will interact with each other to provide multiple ecosystem benefits that should be considered in future studies to account for multiple ecosystem services, on top of final crop production.

5.5.4 The inverse relationship of orchard size and yield

Larger orchard size has been shown here to create significantly less yield and yield value per tree in all mixed effect models that exclude organic farms (*Results section 5.4.4*). In this study, orchard size is a description of field size that contains apples with no other crops present. Results of this relationship are supported by the literature (Carletto et al., 2013; Cornia, 1985; Garibaldi et al., 2016; Kremen & Miles, 2012; Ricciardi et al., 2021; Rosset, 2000; Ünal, 2008; and many others highlighted by Fan & Chan-Kang, 2005). Garibaldi et al. (2016) found the effectiveness of ecological intensification of crop lands to be more successful on smaller versus larger holdings, where flower-visiting pollinator density increased yields at a higher rate on smaller holders versus large farms across 33 different crops. This chapter adds to this finding by removing the ecological intensification practices found on organic farms (birds as pest control) from the analysis, smaller non-organic fields are found to have higher yields than larger non-organic farms. This finding demonstrates that an environmental factor, rather than insecticide usage or other management related to organic farming, has more impact on yields within conventionally managed orchards. Although this chapter did not focus on other ecosystem services, such as arthropod natural enemies or pollination, it has been demonstrated by Bianchi et al. (2006) that the impacts from ecosystem services are larger on small farms due to reduced distance from non-crop habitats compared to larger farms. Another example with similar conclusions demonstrates the use of windbreaks increase crop production through creating diverse non-crop or 'edge' habitats (Brandle et al., 2004).

The findings in this chapter are more recently echoed in the meta-analysis by Ricciardi et al. (2021), where smaller farms on average and worldwide, had greater yield whilst harbouring more biodiversity at the field and landscape scales on small-holder dominant countries. However, when controlling for labour resources the same relationship was not found, highlighting how smaller farms may have more available family labour to encourage higher yields. The influence of family labour or increased labour inputs per hectare on smaller farms is a likely explanation, of which both Ricciardi et al. (2021) and Dorward (2013) highlight to be a beneficial variable to measure when looking at yield per hectare.

Orchard size is the most dominant and consistently significant variable in all mixed effect models (*Results section 5.4.4).* When relating these findings to the literature highlighted above, the most plausible reason for this is the proximity to edge habitats and surrounding natural habitats that are providing the apple orchard with ecosystem services (Bianchi et al., 2006; Brandle et al., 2004; Garibaldi et al., 2016; Pywell et al., 2015). However, the insignificant variables of woody cover and wild bird community variables in the model results of this chapter have not been in consensus with this rational. Alternatively, Rosset (2000) finds intercropping to play a role in this relationship on small farm sizes, which were found to be 200-1000% more productive than large farms. This is an implausible explanation for this chapter, which measures yield per tree.

Reasons for this disconnect between explanations seen here and the literature could be the presence of the dominant orchard size variable in the models of this chapter and the similarity of the landscape across all field sites (as described in *The Study System: section 2.1)*. If all field sizes of non-organic fields were of a similar, small size and the landscape was varied across different sites, there may be more chance of seeing an impact of the variable 'woody cover' as a measure of distance to natural habitat and edge species that support ecosystem services. Future studies should aim to focus on unifying field sizes, incorporating farm labour availability, whilst ensuring larger differences in woody cover are represented to understand the underlying landscape or farm labour factors at play on small farms, that could be causing this inverse relationship of orchard size and yield seen.

5.5.5 Conclusion

The focus on yield and yield value in this chapter provides important insights into the extent and limits to which chemicals can benefit yield and yield value compared with wild bird equivalents available on organic farms. Overall, this chapter adds to the literature and fills in missing research gaps to understand the differences in yield and yield value of apples between organic and three non-organic orchards, as recommended by Kleijn et al. (2019). This chapter shows the appeal to farm in a non-organic way if producing more yields is the end goal, but chemical inputs is not the key to producing them. Although wild birds do not increase yields and yield value per hectare in non-organic orchards, chemical inputs should not fully replace a natural ecosystem service as higher insecticide use causes yield value of all non-organic orchards to be like organic farming.
This highlights the need to financially support organic growers, whose farms support and foster increased biodiversity and ecosystem services, for lower yields to be financially compensated through higher yield value.

Although results from this chapter suggest that conventional and IPM farming produces higher apple yields, chemical intensification of agriculture is not the answer. Firstly, since insecticides do not increase cider apple yields when comparing just between non-organic farms. Secondly, due to the destructive nature agriculture intensification has caused worldwide, alongside being a major driver for the crossing of several planetary boundaries (Beillouin et al., 2021; Campbell et al., 2017). Lastly, the volatile nature of the agro-chemical industry with chemicals becoming prohibited and the decreasing demand for cider apples in the industry (NACM, 2018), causes yield value per hectare of organic and high chemical non-organic farms to be in-different to each other.

The use of yield valuation and comparison studies ultimately lie with farmers and whether they choose birds or chemicals as their pest regulation service. A farmer's perspective is important to incorporate in biodiversity conservation; they disproportionately feel the economic losses associated with crop loss as the price to consumers change very little in order to avoid consumption decreases (Letourneau et al., 2015). If low-cost insecticide availability changes, as it did in 2016, production costs will be forced to be reduced further to avoid negative impacts on farm finances. This site specific valuation of yields provides assistance in decision making where economic aspects of biodiversity and ecological functions have not previously been brought in to farm management discussions (Hungate & Cardinale, 2017; Kleijn et al., 2019). Many farmers in this study, and many worldwide, are still unaware of this vast academic literature on the value of ecosystem services and the concept is still new (Kleijn et al., 2019). It is therefore important for the farming community to share research and experience of how lower insecticides used in farm management practices may see successes for yield increases and future benefits to biodiversity over time (Pretty et al., 2018).

Finally, LEAF farms have demonstrated that both apple yield and yield value per hectare across both years is not different to organic orchard yields and yield value. LEAF farms do not have a price premium attached to their produce to account for lower yields than IPM and conventional, like organic does. This is the first scientific study of LEAF which has proven does not support the farmer financially due to the reduced yield and yield value compared to both organic and other non-organic farms, even though LEAF values set out to enhance farmer prosperity.

5.6 REFERENCES

Albrecht, M., Kleijn, D., Williams, N. M., Tschumi, M., Blaauw, B. R., Bommarco, R., … Sutter, L. (2020). The effectiveness of flower strips and hedgerows on pest control, pollination services and crop yield: a quantitative synthesis. *Ecology Letters, 23*(10), 1488–1498. https://doi.org/10.1111/ele.13576

Balmford, A., Green, R. E., & Scharlemann, J. P. W. (2005). Sparing land for nature:exploring the potential impact of changes in agricultural yield on the area needed for crop production. *Global Change Biology*, *11*, 1594–1605. http://doi.org/10.1111/j.1365-2486.2005.01035.x

Barnosky, A. D., Matzke, N., Tomiya, S., Wogan, G. O. U., Swartz, B., Quental, T. B., … Ferrer, E. A. (2011). Has the Earth's sixth mass extinction already arrived? *Nature*, *471*(7336), 51–57. http://doi.org/10.1038/nature09678

Bateman, I. J., Mace, G. M., Fezzi, C., Atkinson, G., & Turner, K. (2011). Economic analysis for ecosystem service assessments. *Environmental and Resource Economics*, *48*(2), 177–218. http://doi.org/10.1007/s10640-010-9418-x

Bates, D., Maechler, M., Bolker, B., & Walker, S. (2015). Fitting Linear Mixed-Effects Models Using lme4. *Journal of Statistical Software*, *67*(1), 1–48. http://doi.org/10.18637/jss.v067.i01

Beillouin, D., Ben-Ari, T., Malézieux, E., Seufert, V., & Makowski, D. (2021). Positive but variable effects of crop diversification on biodiversity and ecosystem services. *Global Change Biology. https://doi.org/10.1111/gcb.15747*

Bengtsson, J. (2015). Biological control as an ecosystem service: partitioning contributions of nature and human inputs to yield. *Ecological Entomology, 40 (Suppl. 1)*, 45-55.

Bianchi, F. J. J. a, Booij, C. J. H., & Tscharntke, T. (2006). Sustainable pest regulation in agricultural landscapes: a review on landscape composition, biodiversity and natural pest control. *Proceedings. Biological Sciences / The Royal Society*, *273*(1595), 1715–1727. https://doi.org/10.1098/rspb.2006.3530

Bommarco, R., Kleijn, D., & Potts, S. G. (2013). Ecological intensification: harnessing ecosystem services for food security. *Trends in Ecology & Evolution*, *28*(4), 230–8. http://doi.org/10.1016/j.tree.2012.10.012

Brandle, J. R., Hodges, L., & Zhou, X. H. (2004). Windbreaks in North American agricultural systems. *Agroforestry Systems, 61*, 65–78. https://doi.org/10.1007/978-94-017-2424-1_5

Bullock, J. M., Aronson, J., Newton, A. C., Pywell, R. F., & Rey-Benayas, J. M. (2011). Restoration of ecosystem services and biodiversity: Conflicts and opportunities. *Trends in Ecology and*

Evolution, 26(10), 541–549. https://doi.org/10.1016/j.tree.2011.06.011

Burrough, A. ., Oines, C. ., Oram, S. ., & Robertson, H. . (2010). *Traditional Orchard Project in England - the creation of an inventory to support the UK Habitat Action Plan*. London, UK. Retrieved from www.naturalengland.org.uk

Busby, E. (2019). *Millions of apples left to rot in UK as Brexit uncertainty worsens EU fruit picker shortage. The Independent.* Retrieved from https://www.independent.co.uk/news/uk/home-news/brexit-no-deal-fruit-picking-apples-national-farmers-union-eu-workers-harvest-a9163781.html [Accessed 22.08.21].

Campbell, B. M., Beare, D. J., Bennett, E. M., Hall-Spencer, J. M., Ingram, J. S. I., Jaramillo, F., … Shindell, D. (2017). Agriculture production as a major driver of the earth system exceeding planetary boundaries. E*cology and Society, 22*(4). https://doi.org/10.5751/ES-09595-220408

Cardinale, B. J., Duffy, J. E., Gonzalez, A., Hooper, D. U., Perrings, C., Venail, P., … Naeem, S. (2012). Biodiversity loss and its impacts on humanity. *Nature, 486*(7401), 59–67. http://doi.org/10.1038/nature11148.Access

Carletto, C., Savastano, S., & Zezza, A. (2013). Fact or artifact: The impact of measurement errors on the farm size-productivity relationship. *Journal of Development Economics, 103*(1), 254–261. https://doi.org/10.1016/j.jdeveco.2013.03.004

Cerasoli, C. P., Nicklin, J. M., & Ford, M. T. (2014, February 3). Intrinsic Motivation and Extrinsic Incentives Jointly Predict Performance: A 40-Year Meta-Analysis. Psychological Bulletin. Advance online publication. http://dx.doi.org/10.1037/a0035661

Cornia, G. A. (1985). Farm size, land yields and the agricultural production function: An analysis for fifteen developing countries. *World Development, 13*(4), 513–534. https://doi.org/10.1016/0305-750X(85)90054-3

Costanza, R., D'Arge, R., Groot, R. d., Farber, S., Grasso, M., Hannon, B., … van den Belt, M. (1997). The value of the world's ecosystem services and natural capital. *Nature, 387*(15th May), 253–260. http://doi.org/10.15713/ins.mmj.3

Costanza, R., de Groot, R., Braat, L., Kubiszewski, I., Fioramonti, L., Sutton, P., … Grasso, M. (2017). Twenty years of ecosystem services: How far have we come and how far do we still need to go? *Ecosystem Services, 28,* 1–16. https://doi.org/10.1016/j.ecoser.2017.09.008

Crawley. (2007). The R Book. Chichester: John Wiley & Sons Ltd. http://doi.org/10.15713/ins.mmj.3

Crawley, M. J. (2013). The R Book (Second Edi). Chichester: John Wiley & Sons Ltd.

Cross, J. V, Fountain, M., Markó, V., & Nagy, C. (2015). Arthropod ecosystem services in apple orchards and their economic benefits Arthropod ecosystem services in apple orchards. *Ecological Entomology, 40*, 82–96. http://doi.org/10.1111/een.1223

Crowder, D. W., & Reganold, J. P. (2015). Financial competitiveness of organic agriculture on a global scale. *Proceedings of the National Academy of Sciences, 112*(24), 7611–7616. http://doi.org/10.1073/pnas.1423674112

DEFRA. (2012). Agriculture in the United Kingdom. *The Food Industry* (chapter 7). Retrieved from http://www.defra.gov.uk/statistics/files/defra-stats-foodfarm-crosscutting- auk-2011-chapter07-foodchain-120612.xls

Dorward, A. (2013). Agricultural labour productivity, food prices and sustainable development impacts and indicators. *Food Policy*, *39*, 40–50. https://doi.org/10.1016/j.foodpol.2012.12.003

Edwards, P. J., & Abivardi, C. (1998). The Value of Biodiversity: Where Ecology and Economy blend. *Biological Conservation, 83*(3), 239–246.

Fan, S., & Chan-Kang, C. (2005). Is small beautiful? Farm size, productivity, and poverty in Asian agriculture. In D. Colman & N. Vink (Eds.), *Reshaping Agriculture's Contributions to Society*. Blackwell Publishing Inc

Fountain, M. T., & Harris, A. L. (2015). Non-target consequences of insecticides used in apple and pear orchards on Forficula auricularia L . (Dermaptera: Forficulidae). *Biological Control*, *91*(2015), 27–33. http://doi.org/10.1016/j.biocontrol.2015.07.007

Gabriel, D., Sait, S. M., Kunin, W. E., & Benton, T. G. (2013). Food production vs. biodiversity: Comparing organic and conventional agriculture. *Journal of Applied Ecology*, *50*(2), 355–364. http://doi.org/10.1111/1365-2664.12035

Garibaldi, L. A., Carvalheiro, L. G., Vaissière, B. E., Gemmill-herren, B., Hipólito, J., Freitas, B. M., … Blochtein, B. (2016). Mutually beneficial pollinator diversity and crop yield outcomes in small and large farms. Science, 351(6271), 388–391.

García, D., Miñarro, M., & Martínez-sastre, R. (2018). Birds as suppliers of pest control in cider apple orchards: Avian biodiversity drivers and insectivory effect. *Agriculture, Ecosystems and Environment*, *254*(December), 233–243. http://doi.org/10.1016/j.agee.2017.11.034

García, D., Miñarro, M., & Martínez-Sastre, R. (2021). Enhancing ecosystem services in apple orchards: Nest boxes increase pest control by insectivorous birds. J*ournal of Applied Ecology, 58*(3), 465–475. https://doi.org/10.1111/1365-2664.13823

Garratt, M. P. D., Wright, D. J., & Leather, S. R. (2011). The effects of farming system and fertilisers on pests and natural enemies: A synthesis of current research. *Agriculture, Ecosystems and Environment*, *141*(3–4), 261–270. http://doi.org/10.1016/j.agee.2011.03.014

Google Earth, V 7.3.4.8573 (2015; 2016), Eye alt 426 metres. Maxar Technologies 2021. http://www.earth.google.com [July 15, 2021].

Greiner, R., Patterson, L., & Miller, O. (2009). Motivations, risk perceptions and adoption of conservation practices by farmers. Agricultural Systems, 99(2–3), 86–104. https://doi.org/10.1016/j.agsy.2008.10.003

Greiner, R., Patterson, L., & Miller, O. (2009). Motivations, risk perceptions and adoption of conservation practices by farmers. *Agricultural Systems, 99*(2–3), 86–104. https://doi.org/10.1016/j.agsy.2008.10.003

Hillocks, R. J. (2012). Farming with fewer pesticides: EU pesticide review and resulting challenges for UK agriculture. *Crop Protection, 31*. http://doi.org/10.1016/j.cropro.2011.08.008

HSE. (2016). Changes to authorisations for products containing chlorpyrifos. Retrieved November 16, 2017, from http://www.hse.gov.uk/pesticides/news/information-update-0316.htm

Hungate, A. B., & Cardinale, J. B. (2017). Biodiversity: what value should we use ? *Frontiers in Ecology and the Environment*, *15*(6), 283. http://doi.org/10.1002/fee.1511

Jose, S. (2009). Agroforestry for ecosystem services and environmental benefits: An overview. *Agroforestry Systems, 76*(1), 1–10. https://doi.org/10.1007/s10457-009-9229-7

Jouzi, Z., Azadi, H., Taheri, F., Zarafshani, K., Gebrehiwot, K., Van Passel, S., & Lebailly, P. (2017). Organic Farming and Small-Scale Farmers: Main Opportunities and Challenges. *Ecological Economics, 132*(2017), 144–154. https://doi.org/10.1016/j.ecolecon.2016.10.016

Kleijn, D., Bommarco, R., Fijen, T. P. M., Garibaldi, L. A., Potts, S. G., & van der Putten, W. H. (2019). Ecological Intensification: Bridging the Gap between Science and Practice. *Trends in Ecology and Evolution, 34*(2), 154–166. https://doi.org/10.1016/j.tree.2018.11.002

Kremen, C., & Miles, A. (2012). Ecosystem Services in Biologically Diversified versus Conventional Farming Systems: Benefits, Externalities, and Trade-Offs. *Ecology and Society*, *17*(4).

Kronenberg, J. (2014). What can the current debate on ecosystem services learn from the past? Lessons from economic ornithology. Geoforum, 55(2014), 164–177. http://doi.org/10.1016/j.geoforum.2014.06.011

Lamichhane, J. R. (2017). Pesticide use and risk reduction in European farming systems with IPM: An introduction to the special issue. *Crop Protection*, *97*(2017), 1–6. http://doi.org/10.1016/j.cropro.2017.01.017

LEAF. (2016). *LEAF in 2016: Delivering more sustainable food and farming*. Warwickshire.

LEAF. (2017). *LEAF Marque Standard v.14.1*. Warwickshire.

Lele, S., Springate-Baginski, O., Lakerveld, R., Deb, D., & Dash, P. (2013). Ecosystem services: Origins, contributions, pitfalls, and alternatives. *Conservation and Society, 11*(4), 343–358.

https://doi.org/10.4103/0972-4923.125752

Letourneau, D. K., Ando, A. W., Jedlicka, J. A., Narwani, A., & Barbier, E. (2015). Simple-but-sound methods for estimating the value of changes in biodiversity for biological pest control in agriculture. *Ecological Economics*, *120*, 215–225. http://doi.org/10.1016/j.ecolecon.2015.10.015

Letourneau, D. K., Jedlicka, J. A., Bothwell, S. G., & Moreno, C. R. (2009). Effects of Natural Enemy Biodiversity on the Suppression of Arthropod Herbivores in Terrestrial Ecosystems. *Annual Review of Ecology, Evolution, and Systematics*, *40*(1), 573–592. http://doi.org/10.1146/annurev.ecolsys.110308.120320

Lewis, W. J., Lenteren, J. C. Van, Phatak, S. C., & Tumlinson, J. H. (1997). A total system approach to sustainable pest management. *Proceedings of the National Academy of Sciences of the United States of America, 94*(November), 12243–12248

Lloyd-Smith, P. (2018). A Note on the Robustness of Aggregate Ecosystem Service Values. *Ecological Economics*, *146*(November), 778–780. http://doi.org/10.1016/j.ecolecon.2017.12.008

Luck, G. W., Harrington, R., Harrison, P. A., Kremen, C., Berry, P. M., Bugter, R., … Zobel, M. (2009). Quantifying the Contribution of Organisms to the Provision of Ecosystem Services. *American Institute of Biological Sciences, 59*(3), 223–235. https://doi.org/10.1025/bio.2009.59.3.

Martínez-Sastre, R., Miñarro, M., & García, D. (2020). Animal biodiversity in cider apple orchards: Simultaneous environmental drivers and effects on insectivory and pollination. *Agriculture, Ecosystems and Environment, 295*(November 2019), 106918. https://doi.org/10.1016/j.agee.2020.106918

Martínez-Sastre, R., García, D., Miñarro, M., & Martín-López, B. (2020). Farmers' perceptions and knowledge of natural enemies as providers of biological control in cider apple orchards. *Journal of Environmental Management, 266*(December 2019). https://doi.org/10.1016/j.jenvman.2020.110589

Mace, G.M. & Bateman, I. (2011) Conceptual framework and method- ology. *UKNEA: The UK National Ecosystem Assessment*, *pp. 11–25. UNEP-WCMC*, Cambridge, UK.

Mäder, P., Fliessbach, A., Dubois, D., Gunst, L., Fried, P., & Niggli, U. (2002). Soil fertility and biodiversity in organic farming. *Science*, *296*(5573), 1694–7. http://doi.org/10.1126/science.1071148

Marsden, J. (2018). *Personal Communication.*

Millennium Ecosystem Assessment. (2005). *MA Conceptual Framework*.

NACM. (2018). *NACM AGM and the state of the cider industry*. Gloucester.

NACM. (2021) *Harriet Baldwin MP shows support for cider makers.* Retreievd from https://ciderUK.com/harriett-baldwin-mp-shows-support-for-cider-makers/ [Accessed 22.08.21]

NFU. (2021). *Australia trade deal: Five questions the UK government must answer. National Farmers Union Online.* Retrieved from https://www.nfuonline.com/about-us/our-offices/external-affairs-westminster/westminster-news/australia-trade-deal-five-questions-the-uk-government-must-answer/ [Accessed 22.08.21]

Norton, B. G. (2017). A Situational Understanding of Environmental Values and Evaluation. *Ecological Economics*, *138*, 242–248. http://doi.org/10.1016/j.ecolecon.2017.03.024

Östman, Ö., Ekbom, B., & Bengtsson, J. (2001). Landscape heterogeneity and farming practice influence biological control. *Basic and Applied Ecology, 371*(2001), 365–371

Östman, Ö., Ekbom, B., & Bengtsson, J. (2003). Yield increase attributable to aphid predation by ground-living polyphagous natural enemies in spring barley in Sweden. *Ecological Economics, 45*(1), 149–158. https://doi.org/10.1016/S0921-8009(03)00007-7

Pandeya, B., Buytaert, W., Zulkafli, Z., Karpouzoglou, T., Mao, F., & Hannah, D. M. (2016). A comparative analysis of ecosystem services valuation approaches for application at the local scale and in data scarce regions. *Ecosystem Services*, *22*(November 2015), 250–259. http://doi.org/10.1016/j.ecoser.2016.10.015

Pantera, A., Burgess, P. J., Mosquera Losada, R., Moreno, G., López-Díaz, M. L., Corroyer, N., … Malignier, N. (2018). Agroforestry for high value tree systems in Europe. *Agroforestry Systems, 92*(4), 945–959. https://doi.org/10.1007/s10457-017-0181-7

Paul, C., Hanley, N., Meyer, S. T., Fürst, C., Weisser, W. W., & Knoke, T. (2020). On the functional relationship between biodiversity and economic value. *Science Advances, 6*(5). https://doi.org/10.1126/sciadv.aax7712

Penvern, S., Fernique, S., Cardona, A., Herz, A., Ahrenfeldt, E., Dufils, A., … Sigsgaard, L. (2019). Farmers' management of functional biodiversity goes beyond pest management in organic European apple orchards. *Agriculture, Ecosystems and Environment, 284* (May), 106555. https://doi.org/10.1016/j.agee.2019.05.014

Pimentel, D., & Peshin, R. (2014). *Integrated Pest Management*. Dordrecht: Springer Science + Business Media.

Pretty, J., Benton, T. G., Bharucha, Z. P., Dicks, L. V., Flora, C. B., Godfray, H. C. J., … Wratten, S. (2018). Global assessment of agricultural system redesign for sustainable intensification. *Nature Sustainability, 1*(8), 441–446. https://doi.org/10.1038/s41893-018-0114-0

Primdahl, J., Peco, B., Schramek, J., Andersen, E., & Oñate, J. J. (2003). Environmental effects of agri-environmental schemes in Western Europe. *Journal of Environmental Management, 67*(2), 129–138. https://doi.org/10.1016/S0301-4797(02)00192-5

Pywell, R. F., Heard, M. S., Woodcock, B. A., Hinsley, S., Ridding, L., Nowakowski, M., & Bullock, J. M. (2015). Wildlife-friendly farming increases crop yield: evidence for ecological intensification. *Proc R Soc B, 282*(October 2015). https://doi.org/10.1098/rspb.2015.1740

R Core Team. (2015). *R: A Language and Environment for Statistical Computing*. Vienna, Austria: R Foundation for Statistical Computing. Retrieved from https://www.r-project.org/

Reed, M., Lewis, N., and Dwyer, J. (2017) "The effect and impact of LEAF Marque in the delivery of more sustainable farming: a study to understand the added value to farmers." The CCRI, Gloucester, England

Reganold, J. P., & Wachter, J. M. (2016). organic agriculture in the twenty-first century. *Nature Plants*, *2*(February), 15221. http://doi.org/10.1038/nplants.2015.221

Ricciardi, V., Mehrabi, Z., Wittman, H., James, D., & Ramankutty, N. (2021). Higher yields and more biodiversity on smaller farms. *Nature Sustainability, 4*(7), 651–657. https://doi.org/10.1038/s41893-021-00699-2

Robinson, T. L. (n.d.). Managing High-Density Apple Trees for High Yield and Fruit Quality. Geneva. Retrieved from https://ag.umass.edu/sites/ag.umass.edu/files/pdf-doc-ppt/highdensityapple.pdf

Rosset, P. (2000). The multiple functions and benefits of small farm agriculture in the context of global trade negotiations. *Development (Basingstoke)*, *43*(2), 77–82. https://doi.org/10.1057/palgrave.development.1110149

Samnegård, U., Alins, G., Boreux, V., Bosch, J., García, D., Happe, A. K., … Hambäck, P. A. (2019). Management trade-offs on ecosystem services in apple orchards across Europe: Direct and indirect effects of organic production. *Journal of Applied Ecology, 56*(4), 802–811. https://doi.org/10.1111/1365-2664.13292

Segura, H. R., Barrera, J. F., Morales, H., & Nazar, A. (2004). Farmers' perceptions, knowledge, and management of coffee pests and diseases and their natural enemies in Chiapas, Mexico. *Journal of Economic Entomology*, *97*(5), 1491–1499. http://doi.org/10.1603/0022-0493-97.5.1491

Sekercioglu, C. H. (2006). Increasing awareness of avian ecological function. Trends in Ecology and Evolution, 21(8), 464–471. https://doi.org/10.1016/j.tree.2006.05.007

Seufert, V., 2012. Organic agriculture as an opportunity for sustainable agricultural development. Available at: http://www.mcgill.ca/isid/files/isid/pb_2012_13_seufert.pdf.

Seufert, V., Ramankutty, N., & Foley, J. A. (2012). Comparing the yields of organic and conventional agriculture. *Nature,* 485(7397), 229–232. https://doi.org/10.1038/nature11069

Stern, V. M., Smith, R. F., van den Bosch, R., & Hagen, K. S. (1959). The integration of chemical and biological control of the spotted alfalfa aphid: The integrated control concept.

Hilgardia, *29*(2).

TEEB. (2010). *The Economics of Ecosystems and Biodiversity: Mainstreaming the Economics of Nature: A synthesis of the approach, conclusions and recommendations of TEEB*

Tilman, D. (1999). Global environmental impacts of agricultural expansion: The need for sustainable and efficient practices. *Proceedings of the National Academy of Sciences*, *96*(11), 5995–6000. http://doi.org/10.1073/pnas.96.11.5995

Tilman, D., & Clark, M. (2014). Global diets link environmental sustainability and human health. *Nature*, *515*(7528), 518–522. http://doi.org/10.1038/nature13959

Tilman, D., Clark, M., Williams, D. R., Kimmel, K., Polasky, S., & Packer, C. (2017). Future threats to biodiversity and pathways to their prevention. *Nature*, *546*(7656), 73–81. http://doi.org/10.1038/nature22900

Tittonell, P. (2014). Ecological intensification of agriculture - sustainable by nature. C*urrent Opinion in Environmental Sustainability, 8,* 53–61.

Tschumi, M., Albrecht, M., Bärtschi, C., Collatz, J., Entling, M. H., & Jacot, K. (2016). Perennial, species-rich wildflower strips enhance pest control and crop yield. *Agriculture, Ecosystems and Environment, 2*20(2016), 97–103. https://doi.org/10.1016/j.agee.2016.01.001

Tuck, S. L., Winqvist, C., Mota, F., Ahnström, J., Turnbull, L. A., & Bengtsson, J. (2014). Land-use intensity and the effects of organic farming on biodiversity: a hierarchical meta-analysis. *Journal of Applied Ecology*, *51*(3), 746–755. http://doi.org/10.1111/1365-2664.12219

Ünal, Fatma Gül (2008) : Small is beautiful: evidence of an inverse relationship between farm size and yield in Turkey, *Working Paper, No. 55*1, Levy Economics Institute of Bard College, Annandale-on-Hudson, NY

Varah, A., Jones, H., Smith, J., & Potts, S. G. (2013). Enhanced biodiversity and pollination in UK agroforestry systems. *Journal of the Science of Food and Agriculture, 93*(9), 2073–2075. https://doi.org/10.1002/jsfa.6148

Wenny, D. G., DeVault, T. L., Johnson, M. D., Kelly, D., H. Sekercioglu, C., Tomback, D. F., & Whelan, C. J. (2011). The Need to Quantify Ecosystem Services Provided by Birds. *The Auk*, *128*(1), 1–14. http://doi.org/10.1525/auk.2011.10248

Wickham, H. (2016). ggplot2: Elegant Graphics for Data Analysis. New York: Springer-Verlag.

Wilson, C., & Tisdell, C. (2000). *Why farmers continue to use pesticides despite environmental, health and sustainability costs* (No. 53). *Economics, Ecology and the Environment*. Brisbane, Australia.

Zhang, H., Potts, S. G., Breeze, T., & Bailey, A. (2018). European farmers' incentives to promote natural pest control service in arable fields. *Land Use Policy*, *78*(July), 682–690. http://doi.org/10.1016/j.landusepol.2018.07.017

Zhang, W., & Swinton, S. M. (2009). Incorporating natural enemies in an economic threshold for dynamically optimal pest management. *Ecological Modelling*, *220*(9–10), 1315–1324. http://doi.org/10.1016/j.ecolmodel.2009.01.027

Zhang, W., & Swinton, S. M. (2012). Optimal control of soybean aphid in the presence of natural enemies and the implied value of their ecosystem services. *Journal of Environmental Management*, *96*(1), 7–16. http://doi.org/10.1016/j.jenvman.2011.10.008

Appendix O

Cider apple yield and yield value per tree ‘full’ LMER model summaries with insectivore community variables, excluding organic farms

Linear mixed effects full model (yield) or model average summary (yield value) examining apple yield and apple yield value probabilities as the response variable. Highlighted here are insectivore abundance, insectivore species richness and insectivore diversity given as separate models, all without organic farms in the data set. Each lmer included fixed effect variables: farm management, woody cover, orchard size and insecticide use. The intercept is conventional farm management. Abundance and species richness not included in model average for apple cider yield value. Values in bold are significant (P=<0.05). VIF is the variance inflation factor value derived from squared GVIF^(1/(2*Df)), where any variable above 5 was considered correlated and removed from the global model before dredging.

Cider Yield		**Insectivore Abundance**				**Insectivore Species Richness**				**Insectivore Diversity**			
Fixed Effects		Value	Standard Error	*P*-value	VIF	Value	Standard Error	*P*-value	VIF	Value	Standard Error	*P*-value	VIF
Farm Management	Conventional (Intercept)	5.874	1.917	**0.002**	1.123	0.005	0.016	**0.002**	1.123	0.051	0.016	**0.002**	1.123
	IPM	1.308	2.286	0.567	1.123	0.011	0.019	0.578	1.123	0.011	0.02	0.568	1.123
	LEAF	-3.513	2.510	0.162	1.123	0.029	0.021	0.171	1.123	-0.029	0.022	0.173	1.123
Pest control	Insecticide Volume	1.562	0.871	0.073	1.209	0.013	0.007	**0.008**	1.209	0.013	0.007	0.072	1.209
Environmental/ Landscape	Orchard size	-0.579	0.152	**<0.001**	1.026	0.005	0.001	**<0.001**	1.031	-0.005	0.002	**<0.001**	1.037
	Woody Cover	1.088	1.243	0.381	1.27	0.009	0.01	0.368	1.27	0.009	0.01	0.38	1.27
Insectivore community metrics	Abundance	<0.001	0.008	0.958	1.02	-	-	-	-	-	-	-	-
	Species Richness	-	-	-	-	<-0.001	<0.001	0.79	1.066	-	-	-	-
	Diversity	-	-	-	-	-	-	-	-	<-0.001	-	-	1.029

Cider Yield Value													
Farm Management	Conventional (Intercept)	5.797	1.92	**0.003**	1.123	5.797	1.92	**0.003**	1.123	5.8	1.92	**0.003**	1.123
	IPM	1.06	2.324	0.649	1.123	1.06	2.324	0.649	1.123	1.079	2.322	0.644	1.123
	LEAF	-3.237	2.512	0.199	1.123	-3.237	2.512	0.199	1.123	-3.253	2.512	0.196	1.123
Pest control	Insecticide Volume	2.544	1.893	0.18	1.209	2.544	1.893	0.18	1.209	2.583	1.889	0.172	1.209
Environmental/ Landscape	Orchard size	-1.159	0.298	**<0.001**	1.026	-1.159	0.298	**<0.001**	1.031	-1.153	0.299	**<0.001**	1.037
	Woody Cover	1.44	2.378	0.546	1.27	1.44	2.378	0.546	1.27	1.484	2.39	0.536	1.27
Insectivore community metrics	Abundance	-	-	-	1.02	-	-	-	-	-	-	-	-
	Species Richness	-	-	-	-	-	-	-	1.023	-	-	-	-
	Diversity	-	-	-	-	-	-	-	-	<-0.001	-0.004	0.836	1.029

Appendix P

Cider apple yield and cider yield value per tree 'full' LMER model summary/ average summary, without insectivore variables, excluding organic farms.

Linear mixed effects full model summary (yield) and model average summary (yield value) examining apple yield and apple yield value probabilities as the response variable. Investigation of both response variables was highlighted here looking at farm management options without the inclusion organic farms and excluding insectivore community variables. Each lmer included the same fixed effect variables: farm management, woody cover, orchard size and insecticide use. The intercept is conventional farm management. Values in bold are significant (P=<0.05). VIF is the variance inflation factor value derived from squared GVIF^(1/(2*Df)), where any variable above 5 was considered correlated and removed from the global model before dredging.

		Cider apple yield				**Cider apple yield value**			
Fixed Effects		Value	Standard Error	*P*-value	*VIF*	Value	Standard Error	*P*-value	*VIF*
Farm Management	Conventional (Intercept)	0.05	0.016	0.02	1.123	5.797	1.92	0.003	1.123
	IPM	0.011	0.019	0.581	1.123	1.06	2.323	0.649	1.123
	LEAF	-0.029	0.021	0.171	1.123	-3.237	2.512	0.199	1.123
Pest control	Insecticide use	0.013	0.007	0.08	1.209	2.543	1.893	0.18	1.209
Environmental/ Landscape impacts	Orchard size	-0.006	0.001	<0.001	1.007	-1.159	0.298	<0.001	1.007
	Woody cover	0.009	0.01	0.365	1.27	1.44	2.378	0.546	1.27

CHAPTER 6

Discussion

6.1 IN SUMMARY

Few studies, with the objective to understand the relationship between biodiversity and pest ecosystem services, compare the service availability on different types of farm management systems (Peisley et al., 2016; Howe et al., 2009; Viser & Mols, 2007). Furthermore there is a shortage of research that values ecosystem services in terms of a comparison to the man-made replacement and also in terms of three trophic levels: birds, their prey and then plants or crop (Wenny et al., 2011), using the final ecosystem of yield to give more meaning to farmers (Kleijn et al., 2019). In this thesis I addressed these knowledge gaps by using birds as the pest control ecosystem service to farmers, comparing the service provided on organic and non-organic alternatives over 30 orchards in Herefordshire and surrounding county borders or Gloucestershire and Worcestershire, in terms of predation rates, yield and yield value.

6.1.1 Farming practices and biodiversity

The first Chapter of my thesis adds to the plethora of knowledge that organic farms support higher levels of biodiversity than non-organic. Organic orchards had a higher diversity, species richness, abundance, and density of birds than conventionally managed orchards using chemical insecticides (Chapter 1). Although this is not novel (Tuck et al., 2014), it was an important first step in understanding the biodiversity levels on different farm management types, in order to start investigating the ecosystem services supported by biodiversity. However, this chapter does add to the literature regarding the effectiveness of alleged less intense methods of farming than conventional. Linking Environment and Farming (LEAF) guide farmers to follow landscape and nature conservation standards on LEAF Marque (EC, 2017; LEAF, 2017). LEAF is a growing community of farmers, yet scientific testing of its impacts on biodiversity does not exist. I have shown here that LEAF management holds the same biodiversity levels as conventional farming on apple orchards. This is an area that needs much further research, particularly as LEAF has started to expand practices to global farming communities without the scientific rigour to prove the effects of their farming strategy on biodiversity (LEAF, 2019).

Furthermore, not only did LEAF not provide benefits to biodiversity, they also did not bring yield benefits to the growers. Although this is not advertised specifically in their guiding principles, it is implied when describing the farming style as “prosperous” (Reed et al., 2017:9). With no increase in value, such as organic apples, there seems limited benefits of LEAF farming in terms of yield and benefits to biodiversity, over conventional or IPM, other than the increased access to market.

Although IPM and conventional farming practices in this study may be similar in terms of management – due to the self-diagnosis of the farm management used – the science behind IPM has not been developed even though widely accepted and incorporated into EU policies (Stenberg, 2017). IPM as a management practice that supports biodiversity is conflicted in the literature, with few studies agreeing on whether the impact is positive or negative (Todd et al., 2011; Genghini et al., 2006). This thesis adds to the literature to show that IPM was not as beneficial as organic in terms of biodiversity and biological control provision from birds, and that IPM methods are not widely known or understood by farmers. When describing farm management, farmers may class themselves as conventional when they are a low level IPM. This discovery shows that more research and exposure of IPM research is needed to improve the uptake of this farm management practice.

6.1.2 Ecosystem services in agricultural systems

Although studies explore the impacts of ecosystem services on yield and quality of crops (Mols & Viser, 2002; Classen et al., 2014; Garratt et al., 2014), most studies are based on a comparison between no ecosystem service present to the presence of one, rather than comparing a man-made ecosystem service to a natural one. As we see from Chapter 3, biodiversity is greater on organic orchards, but the functional consequences of this, is still poorly understood in the literature. Furthermore, there are still lack of studies demonstrating the role of birds in a temperate system (García et al., 2018). Extensive farm-based research, rather than plot level studies, on apples in temperate areas are not available in the literature. Chapter 4 is a new research area that links ecosystem service potential, with the ecosystem service provision, seen on working orchards in Herefordshire.

The comparison here of organic and non-organic systems in terms of a pest control service provision shows there are two quite different agro-ecosystems at play. Organic, or no-spray farms, reduce their pests with a natural pest control service, whereas non-organic farms harbour much reduced biodiversity and less natural predation services from birds. Pest moth levels show similarity across all farm types. As yield increases with the use of insecticides (Chapter 5), we can assume that pest damage is still greater on organic farms even though pest moth levels are similar: chemicals are used to reduce numbers on non-organic farms, and birds on organic.

6.1.3 Yield and yield value comparisons between natural and synthetic farming systems

In Chapter 5 I have compared current farm management options available in terms of providing yield and yield value between farms that use a synthetic system of chemical pest control and those than use the natural alternative from birds on organic farms. Per hectare, organic farms do not produce higher yields than farms which use the synthetic alternative, apart from LEAF farms that are not dissimilar to organic yields per hectare. Insecticide use itself has been shown to be less important than other farm management practices at increasing yields per hectare within non-organic farms. When reduced insecticides are use on non-organic farms, yield *value* per hectare is significantly greater than organic value (except LEAF). This result favours the non-organic farming system as most cost effective and profitable to apple growers in the study area, mirrored by Gabriel et al. (2013), Samnegård et al. (2019), Seufert et al. (2012) and Bengtsson (2015), but does not attribute it's success through increased insecticide use. For future decision making this is an important finding, when low-cost insecticide availability is decreased, alongside a static cider apple market and the inevitable resistance of pests to insecticides (Wilson & Tisdell, 2000) the use of insecticides becomes uneconomical compared to organic farming.

Although organic farming has the disadvantage of producing lower yields per hectare than IPM and conventional, there are significant benefits to farms through yield *value* when insecticide use is high and more volatile than the more stable organic farming: organic yield value becomes similar all non-organic yield value. Thereby, through enhancing environmental protection and resilience whilst reducing external input costs on organic orchards (Jouzi et al., 2017), increases organic farmer income to that of a non-organic farmer per hectare.

The impact of this finding for multinational companies such as PepsiCo, who wish to understand how their farmers can increase biodiversity but continue to provide ever increasing yields, is important. Without a change to the way companies value food production instead of valuing quantity (yield) as they do, there will be no change to the current system in place – increased profit for increased yield. The current food production and economic model needs re-focussing to support an organic farming model, where less yield is supported and financially rewarded by organic price premiums in order to support farms who enhance UK biodiversity on traditional apple orchards (Gobbi, 2000; Hole et al., 2005).

6.2 LIMITATIONS AND FUTURE WORK

Due to unfavourable weather in 2016, butterfly data collected was unable to be used towards a biodiversity analysis of farm management types in Chapter 1. Furthermore, butterfly abundances were only taken over one year, which did not account for annual fluctuations in butterflies due to variations in weather conditions from year to year (Brereton et al., 2011; Pollard, 1977). Not all farms were surveyed the same number of times during the surveying period in 2016, with some farms surveyed twice, and other up to five times. This was due to unforeseen fieldworker complications where survey data retrieval from three farms was unobtainable, leaving those farms with less repeat surveys. Re-analysing the butterfly data collected for two years, with fully repeated surveys on each farm, may show the poor availability of butterflies in one year compared to another, but also give enough difference to highlight if butterfly abundance and diversity did or did not share the same pattern across farm management types as birds.

The woody cover landscape product was a tool developed to overcome the pitfalls of just using Land Cover map (2007), and the National Forest Inventory dataset (2015), which both fail to take into account hedgerows (Tebbs & Rowland, 2014). The study area is known for the diverse, ancient hedgerows with some veteran wildlife-supporting trees (Natural England, 2014a, 2014b), however in the Herefordshire lowlands the hedgerows are much shorter (Natural England, 2013) which meant any hedgerow under 2m was not counted. As the main hypotheses were not in relation to landscape variables, rather the type of farm management, I decided these missing hedgerows were reasonable to omit because the woody cover dataset still included woodland areas. For future research in the area, hedgerows need to be verified in-field through ground truthing, and entered to the woody cover data used within GIS (Tebbs & Rowland, 2014).

IPM was used as one of the management options that farmers stated they had chosen as their farm management method during initial farm visits. There are three levels of IPM (Kogan, 1998), where level one is what the majority of IPM farmers in this study followed.

Level one is simple to measure pest thresholds and controls the timings of sprays so not to disrupt pollinators during pollination of apples. Level 2 includes natural enemy counts, use of selective pesticides only and crop rotations and level 3 includes habitat management, multi-crop interactions and takes an agro-ecosystem level approach to avoid disruption to the system (Barzman et al., 2015). To include only level 1 farmers in the study was a limitation as the difference between conventional and level 1 IPM was not shown to be significant, most likely due to the self-diagnosing impact. For future studies it would be more interesting to understand the level of IPM impacts and to which level shows a significant difference to biodiversity compared to conventional (Chapter 3), and to the ecosystem services they provide (Chapter 4).

The issue of self-diagnosing farm management practices is not a specific issue just in this thesis but can be an issue for many farm-scale studies, globally. It has been proven here to be complicated when deciphering the importance of results between farm management types. Without a full investigation into the practices used on farm at the beginning of the study, farmers are placed into a category in which they believe they should be placed, rather than evidentially placed. During the early stages of the project farmers have not gained enough confidence in the study nor built up a confident working relationship necessary to carry out the in-depth investigations needed – both on farm and through a separate, detailed interview process to place the farmers into the category they are matched to. This was only an issue with IPM farms, mainly because there is no formal certification process and information from the EU is hard to find and not widely known in the farming community. Most farmers who were conventional and IPM were advised by an agronomist. These agronomists are chosen based on recommendations within the community and the personal relationships between the farmer and agronomist allow the association to continue. If the agronomist is inclined to research new techniques that IPM methods use, they may or may not explain these to the farmer. In this study, and potentially this is an issue with other studies who rely on farmers self-diagnosing, the agronomist was not contacted. Thus, the details of the farm management, and where those management decisions were cultivated from, were not investigated. This may explain the results in this study where IPM and conventional have similar levels of biodiversity and natural pest control (Chapter 3 and 4) and this area is something that can be researched in more detail in future studies.

During the sentinel prey experiments in Chapter 4, I used both plasticine and dough models at different times of year. Firstly, a limitation here was the lack of a multi-year study. Although this study spanned across two years, the dough models were only used once during winter months on orchards. As birds predated on dough caterpillars more frequently than plasticine, also shown by Lövei & Ferrante, (2016), the use of dough during summer months would be something I would perform differently if repeated, as well as over multiple years to correspond with the bird abundance data collected for Chapter 3 and 4. Furthermore, although dough does not hold predator marks as well as plasticine (Howe et al., 2009) it consists of fatty carbohydrates and protein (Sam et al., 2015) so the energy returned through predation is rewarded with dough, and wasted with plasticine, optimising energy intake per unit of time (Muiruri et al., 2015). For this reason, future studies should use dough over plasticine sentinel prey, but to reduce damage through weather conditions, only leave out in the driest 24-hour period, and no longer than 24 hours to minimise natural weathering.

As an addition to sentinel prey experiments, bird nest boxes would be a good addition to enhance the use of the orchard by cavity nesting birds, such as great tits (*Parus major*) and blue tits (*Cyanistes caeruleus*), as shown by Mols and Visser (2007) and more recently García et al. (2021), who found positive impacts that reduced pest damage on apple trees from increased bird abundance and predation. Although García et al. (2021:1) describe the method of erecting nest boxes as "easy to implement, cheap and attractive measure of ecological intensification", this method was inaccessable to me as a lone worker and posed some health and safety risks that were prefereable to avoid. However, for future studies that have teams of students available, this method of measuring pest control capabilities from insectivorous birds would be important to replicate.

Pest moth abundances per farm management type were estimated as a measure of pest pressure on apple orchards, and cause major yield declines (Solomon et al., 1976). However, here I did not show that moth pest levels affected apple damage and yields directly. If further work were to be undertaken, pest moth as well as apple damage estimates through leaf assessments and apple visual inspections would link the pest levels to the amount of damage caused. Furthermore, pests were only measured using three moth pests, where as there are many other pests, included more moth pest species such as the Winter moth, the rosey apple aphid, and the apple blossom weevil that can cause yield reductions that can both be controlled by natural enemies (AHDB, 2018). For a full pest control service analysis given by birds, all pests should be assessed in future analysis.

During the comparison of pest control methods (natural and synthetic), the use of insecticide was used as the synthetic alternative to understand the impact on yield (Chapter

5). However, we also know that fertilisers (Garratt et al., 2011), fungicides and herbicides also increase yields (Barzman et al., 2015; Davis et al., 2012; Mäder et al., 2002). Future work should include all inputs that may increase yield in comparison to insecticides so to find the exact impact of just insecticides on yield in comparison to the natural pest control alternative. Furthermore, a total cost-benefit trade-off, using not just the extra inputs to non-organic farms, but also the negative impacts that birds may cause farmers in relation to yield would be a future avenue of this study (Peisley, Saunders, & Luck, 2016) over long time periods to understand a long-term, full agro-ecological system of the impact of birds on yields in comparison to the synthetic alternative.

Final thoughts on future work are to understand how yield gaps could be compensated, not just through reducing chemical inputs or being sold at an organic premium (Crowder & Reganold, 2015; Chapter 5), but to understand the public's willingness to pay. This research area was a major consideration of a 4th data chapter where apple juice made from organic farms would be sold to consumers at a slightly higher price to compensate for yield loss per hectare to compare sales of apple juice from non-organic orchards sold at the baseline price that make current profits. A choice experiment would evaluate consumers' responses to prevent losses to a pest control ecosystem service through purchasing the end product that protects them (Breeze et al., 2015) and gain an understanding as to whether consumers are willing to pay for a wildlife-premium product, thus incentivise producers to opt for the natural pest control farming option (Bateman et al., 2015). Time limitations and costly production of apple juice with correct labelling were the main barriers here as organic certification or another well-known wildlife-friendly logo that consumers would recognise would take time to certify our apple juice ready for experiments.

6.3 FINAL CONCLUSION

This thesis has demonstrated how farming practices that do not use chemical insecticides, such as organic farming, support higher levels of biodiversity in apple orchards of Herefordshire and bordering counties compared to three types of chemical-spray farming: Integrated Pest Management (IPM), conventional and Linking Environment and Farming (LEAF). These analyses have shown that biodiversity has the capacity to support the functional group necessary to provide a natural pest control ecosystem service from the diverse, abundant and species rich insectivorous bird community on organic (no-spray) orchards. The relationship of increased insectivorous birds on organic farms mirrors the relationship of daily predation rates from birds.

Due to increased market price of organic apples, organic yield value is in-different to non-organic apples when insecticide-use and cost is high. I have shown there is a shift in non-organic decision making to reduce chemical use in response to insecticide insecurity. Insecticide use was not significant to apple yields on non-organic farms and indicates that high-input high-yield intense farming system may not necessarily be the continued agricultural direction, making no-chemical input farming more desirable when insecticide availability is volatile. Other farm management practices on non-organic farming are more important for yields and yield value than insecticide use.

The results of this thesis are directly relevant to the apple growing community of Herefordshire, Gloucestershire, and Worcestershire. They can be used to inform LEAF farms of the pitfalls this management practice has, in comparison to other non-organic practices, in biodiversity and pest control ecosystem service provisions as well as reduced final ecosystem service - crop provision and yield value. Results here can provide insights from four different farm management practices commonly used in the UK to better understand impact to biodiversity, ecosystem service provision and overall crop production to assist future farm practice guidelines and regulations that aim to reduce the loss of biodiversity through farming.

6.4 REFERENCES

AHDB. (2018). *Apples Best Practice Guide. Pests and Disease Control*. Retrieved from http://apples.ahdb.org.uk/ipdm.asp#

Barzman, M., Bàrberi, P., Birch, A. N. E., Boonekamp, P., Dachbrodt-Saaydeh, S., Graf, B., … Sattin, M. (2015). Eight principles of integrated pest management. *Agronomy for Sustainable Development*, *35*(4), 1199–1215. http://doi.org/10.1007/s13593-015-0327-9

Bateman, I. J., Coombes, E., Fitzherbert, E., Binner, A., Bad'ura, T., Carbone, C., … Watkinson, A. R. (2015). Conserving tropical biodiversity via market forces and spatial targeting. *Proceedings of the National Academy of Sciences*, 201406484. http://doi.org/10.1073/pnas.1406484112

Breeze, T. D., Bailey, a. P., Potts, S. G., & Balcombe, K. G. (2015). A stated preference valuation of the non-market benefits of pollination services in the UK. *Ecological Economics*, *111*, 76–85. http://doi.org/10.1016/j.ecolecon.2014.12.022

Brereton, T., Roy, D. B., Middlebrook, I., Botham, M., & Warren, M. (2011). The development of butterfly indicators in the United Kingdom and assessments in 2010. *Journal of Insect Conservation*, *15*(1), 139–151. http://doi.org/10.1007/s10841-010-9333-z

Classen, A., Peters, M. K., Ferger, S. W., Helbig-bonitz, M., Schmack, J. M., Maassen, G., … Bo, K. (2014). Complementary ecosystem services provided by pest predators and pollinators increase quantity and quality of coffee yields. *Proc R Soc B*, *281*.

Crowder, D. W., & Reganold, J. P. (2015). Financial competitiveness of organic agriculture on a global scale. *Proceedings of the National Academy of Sciences*, *112*(24), 7611–7616. http://doi.org/10.1073/pnas.1423674112

Davis, A. S., Hill, J. D., Chase, C. a, Johanns, A. M., & Liebman, M. (2012). Increasing cropping system diversity balances productivity, profitability and environmental health. *PloS One*, *7*(10), e47149. http://doi.org/10.1371/journal.pone.0047149

Gabriel, D., Sait, S. M., Kunin, W. E., & Benton, T. G. (2013). Food production vs. biodiversity: Comparing organic and conventional agriculture. *Journal of Applied Ecology*, *50*(2), 355–364. http://doi.org/10.1111/1365-2664.12035

García, D., Miñarro, M., & Martínez-sastre, R. (2018). Birds as suppliers of pest control in cider apple orchards: Avian biodiversity drivers and insectivory effect. *Agriculture, Ecosystems and Environment*, *254*(December), 233–243. http://doi.org/10.1016/j.agee.2017.11.034

García, D., Miñarro, M., & Martínez-Sastre, R. (2021). Enhancing ecosystem services in apple orchards: Nest boxes increase pest control by insectivorous birds. *Journal of Applied Ecology, 58*(3), 465–475. https://doi.org/10.1111/1365-2664.13823

Garratt, M. P. D., Breeze, T. D., Jenner, N., Polce, C., Biesmeijer, J. C., & Potts, S. G. (2014). Avoiding a bad apple: Insect pollination enhances fruit quality and economic value. *Agriculture, Ecosystems & Environment*, *184*, 34–40. http://doi.org/10.1016/j.agee.2013.10.032

Garratt, M. P. D., Wright, D. J., & Leather, S. R. (2011). The effects of farming system and fertilisers on pests and natural enemies: A synthesis of current research. *Agriculture, Ecosystems and Environment*, *141*(3–4), 261–270. http://doi.org/10.1016/j.agee.2011.03.014

Gobbi, J. a. (2000). Is biodiversity-friendly coffee financially viable? An analysis of five different coffee production systems in western El Salvador. *Ecological Economics*, *33*(2), 267–281. http://doi.org/10.1016/S0921-8009(99)00147-0

Hole, D. G., Perkins, a. J., Wilson, J. D., Alexander, I. H., Grice, P. V., & Evans, a. D. (2005). Does organic farming benefit biodiversity? *Biological Conservation*, *122*(1), 113–130.

http://doi.org/10.1016/j.biocon.2004.07.018

Howe, A., Lövei, G. L., & Nachman, G. (2009). Dummy caterpillars as a simple method to assess predation rates on invertebrates in a tropical agroecosystem. *Entomologia Experimentalis et Applicata*, *131*, 325–329. http://doi.org/10.1111/j.1570-7458.2009.00860.x

Kogan, M. (1998). Integrated Pest Management: Historical Perspectives and Contemporary Developments. *Annual Review of Entomology*, *43*(1), 243–270. http://doi.org/10.1146/annurev.ento.43.1.243

LEAF. (2019). Linking Environment and Farming. *LEAF Marque Standards.* https://archive.leafuk.org/leaf/farmers/LEAFmarquecertification/standard.eb [*Accessed 7th September 2019*]

Lövei, G. L., & Ferrante, M. (2016). A review of the sentinel prey method as a way of quantifying invertebrate predation under field conditions. *Insect Science*, 1–40. http://doi.org/10.1111/1744-7917.12405

Mäder, P., Fliessbach, A., Dubois, D., Gunst, L., Fried, P., & Niggli, U. (2002). Soil fertility and biodiversity in organic farming. *Science*, *296*(5573), 1694–7. http://doi.org/10.1126/science.1071148

Muiruri, E. W., Rainio, K., & Koricheva, J. (2015). Do birds see the forest for the trees? Scale-dependent effects of tree diversity on avian predation of artificial larvae. *Oecologia*. http://doi.org/10.1007/s00442-015-3391-6

Natural England. (2013). *National Character Area profile: 100. Herefordshire Lowlands*.

Natural England. (2014a). *National Character Area profile: 101. Herefordshire Plateau*.

Natural England. (2014b). *National Character Area profile: 104. South Herefordshire and Over Severn*.

Pandeya, B., Buytaert, W., Zulkafli, Z., Karpouzoglou, T., Mao, F., & Hannah, D. M. (2016). A comparative analysis of ecosystem services valuation approaches for application at the local scale and in data scarce regions. *Ecosystem Services*, *22*(November 2015), 250–259. http://doi.org/10.1016/j.ecoser.2016.10.015

Peisley, R. K., Saunders, M. E., & Luck, G. W. (2016). Cost-benefit trade-offs of bird activity in apple orchards. *PeerJ*, *4*, e2179. http://doi.org/10.7717/peerj.2179

Pollard, E. (1977). A method for assessing changes in the abundance of butterflies. *Biological Conservation*, *12*.

Reed, M., Lewis, N., and Dwyer, J. (2017) "The effect and impact of LEAF Marque in the delivery of more sustainable farming: a study to understand the added value to farmers." The CCRI, Gloucester, England

Sam, K., Remmel, T., & Molleman, F. (2015). Material affects attack rates on dummy caterpillars in tropical forest where arthropod predators dominate: an experiment using clay and dough dummies with green colourants on various plant species. *Entomologia Experimentalis et Applicata*, 1–8. http://doi.org/10.1111/eea.12367

Solomon, M., Glen, D., Kendall, D., & Milsom, N. (1976). Predation of overwintering larvae of codling moth (Cydia pomonella (L.)) by birds. *Journal of Applied Ecology*, *13*(2), 341–352. Retrieved from http://www.jstor.org/stable/10.2307/2401784

Stenberg, J. A. (2017). A conceptual Framework for Integrated Pest Managment. *Trends in Plant Science*, *22*(9), 759–769.

Tebbs, E., & Rowland, C. (2014). *A high spatial resolution woody cover map for Great Britain: preliminary results.* Lancaster.

Tuck, S. L., Winqvist, C., Mota, F., Ahnström, J., Turnbull, L. a., & Bengtsson, J. (2014). Land-use intensity and the effects of organic farming on biodiversity: a hierarchical meta-analysis. *Journal of Applied Ecology*, *51*(3), 746–755. http://doi.org/10.1111/1365-2664.12219

Wenny, D. G., DeVault, T. L., Johnson, M. D., Kelly, D., H. Sekercioglu, C., Tomback, D. F., & Whelan, C. J. (2011). The Need to Quantify Ecosystem Services Provided by Birds. *The Auk*, *128*(1), 1–14. http://doi.org/10.1525/auk.2011.10248