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The influence of management and land use diversity at local and landscape scales on communities of generalist predators in commercial dessert apple orchards

Thesis submitted for the degree of Doctor of Philosophy

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Declaration

I confirm that this is my own work and the use of all material from other sources has been properly and fully acknowledged.

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Contents

Abstract

This study explored the combined influences of land use diversity, chemical pest management and grass alley management on the abundance of two key predatory taxa which act as natural enemies of pests in UK commercial dessert apple orchards, namely ground beetles (Coleoptera: Carabidae) and the European Earwig *Forficula auricularia* (Dermaptera: Forficulidae).

Carabid and earwig numbers were positively related to higher predation levels; however, each was affected differently by land use diversity. Land use factors were not related to carabid activity-density but were related to community composition and the representation of traits in the community. Higher proportions of larger, polyphagous species and fewer spring breeders were observed in more diverse land use settings, while adjacent land use was found to influence diurnal activity and habitat associations. The proportion of polyphagous carabids in the assemblage was in turn found to be related to increased predation. Earwigs were not affected by land use diversity in isolation at either scale, though an interaction between adjacent land use and distance into the orchard was observed, indicating a distance mediated effect of non-crop habitats on earwig numbers. Earwig abundance was positively related to increased predation.

Use of the neonicotinoids flonicamid and thiacloprid was associated with declines in the numbers of carabids and earwigs, with thiacloprid use also associated with a number of other carabid community metrics. Chlorpyrifos was associated with greater activity-density of carabids, but lower functional diversity. Earwig numbers did not appear to be affected by chlorpyrifos use under the field conditions employed here.

Both a reduction in the frequency of mowing and raising the height of mower blades led to increases in the abundance of epigeal predators in a study orchard. In contrast, arboreal predator communities were not affected by the mowing treatments applied.

The work presented here clearly demonstrates that generalist natural enemies found within commercial dessert apple orchards are influenced by both the adjacent land use and the diversity of the surrounding land use at a landscape scale. Farm management and chemical pest control have also been shown to affect orchard natural enemy communities. These findings show that both carabids and the European earwig have the potential to contribute to natural pest control in orchards. Further, this work may explain some of the variation observed in earwig populations between orchards. The provision of non-crop habitat, reductions in chemical use and vegetation management could be employed to enhance these natural enemy communities in commercial dessert apple orchards.

Chapter 1 Introduction

1.1. Context

By current estimations, global population is anticipated to reach 9.7 billion by 2050 (United Nations, 2015), with growing numbers and per capita consumption driving greater demand for food production (Tilman *et al*., 2002). As a consequence, around 40% of the global landmass is estimated to be in cultivation (Foley *et al*., 2011). Agricultural intensification is a consequence of this demand, and has been implicated in the decline of biodiversity across a wide range of taxa (Benton *et al*., 2003). Declines are largely attributed to habitat destruction, fragmentation and increased use of agrochemicals (Landis, 2017; Martinson and Fagan, 2014; Potts *et al*., 2010).

In 2016, the Utilised Agricultural Area (UAA) in the UK was measured at 17.4 million hectares, representing approximately 71% of the national land area (Defra, 2017). The majority of the cropping area within the UAA in the UK is utilised for arable production, with approximately 3.1 million hectares laid to cereal in 2016 (17.8% of UAA), with an estimated value of £3.2 billion. In contrast, only 25,000 hectares are laid to orchard crops, just 0.15% of the UAA. However, in 2016, the value of fruit production was estimated at £670 million, of which £205 million is attributed to orchard fruits (Defra, 2017). Of that, £99 million is attributed to dessert apple production with the remaining £106 million divided between culinary apples (£40 million), pears (£11 million) and the remainder comprising primarily cider and stone fruits (Defra, 2017). Dessert apples are therefore a high value component of the UK's agricultural output, with Gala and Braeburn accounting for 43% of the UK's dessert apple production (Garthwaite *et al*., 2016). Nonetheless, improvements in production have led to a global increase in the supply of apples over the past few decades, such that supply is outstripping demand, driving prices down, and increasing pressure on growers to produce top quality fruit (Defra, 2017).

In order to achieve Class I status, and therefore the best prices, dessert apple producers must meet stringent quality criteria (European Commission, 2011). This can be compromised by direct damage to fruit caused by certain pests, e.g. Codling moth *Cydia pomonella* (Lepidoptera: Tortricidae) or Apple Sawfly *Hoplocampa testudinea* (Hymenoptera: Tenthredinidae) (Solomon et al., 2000). In dessert apple varieties, demand for lower classed fruits is low and prices are poor (Cross *et al*., 2015) with consumers selecting apples with external visual defects between just 0.5% and 7% of cases at point-of-purchase (Jaeger *et al*., 2018). While aesthetic selectivity will have implications for producers, visual perceptions of poor quality will also increase food waste overall (de

Hooge *et al*., 2017). Beyond visual appeal, there is a growing trend towards consumption of organic fruit and vegetables, with consumption growing (Willer and Lernoud, 2016). In one study by Zepeda and Deal (2009) organic shoppers 'almost universally' cited health and environmental protection as motivations for organic consumption. Such findings are supported by other research indicating a belief that organic produce is healthier and safer for both the consumer and environment (Denver and Jensen, 2014; Jensen *et al*., 2011; Wier and Calverley, 2002). There is a conflict therefore between the need to produce more food to meet demand (United Nations, 2015), the desire for sustainable, low input produce (Willer and Lernoud, 2016) and the apparently innate preference of the consumer to reject all but visually flawless fruit (Jaeger *et al*., 2018).

Despite the conflicting requirements of the market, the pressures of pest and disease control in commercial dessert apple production means these orchards remain one of the most intensively sprayed crops grown in temperate regions today (Simon *et al*., 2010). However, emerging pesticide resistance, reductions in the range of chemicals authorised, the slow progress in development of new pesticides, and the costs of use, all contribute to growers looking to reduce their dependence on chemicals (Pennell, 2006; Regnault-Roger, 1997). A greater understanding of the complexities of natural enemy abundance within orchards may contribute to a reduced dependency on chemical pest control and more sustainable food production.

1.2. Pests and pest control in apple orchards

Dessert apples are susceptible to damage from a large number of pests, and in the UK these include codling moth and other tortrix moths, rosy apple aphid *Dysaphis plantaginea* and woolly apple aphid *Eriosoma lanigerum* (Hemiptera: Aphididae), apple leaf-curling midge *Dasineura mali* (Diptera: Cecidomyiidae), rhynchites weevil *Tatianaerhynchites aequatus* (Coleoptera: Curculionidae) and apple sawfly.

Under experimental conditions, where UK dessert apple orchards (cv. Cox & cv. Gala) were left entirely untreated over a five year period, the reduction in yield as a result of pest damage was estimated at between 43% and 52% (Cross *et al.* 2005, in Cross *et al.*, 2015).

In a recent review of insecticide usage in UK orchards, covering the period autumn 2015 to summer 2016, treatment against codling moth and rosy apple aphid accounted for 51% of insecticide sprays in non-Cox dessert apple orchards. A further 20% targeted generic ‘caterpillars’ and 15% generic ‘aphids’. The remaining 14% of spray activity was split between ‘tortrix’, apple blossom weevil and woolly aphid (Garthwaite *et al*., 2016).

The range of effective insecticide/acaricide products available to growers has diminished in recent years owing to the development of resistance amongst pests, harmful effects on beneficial species and human health, and consequent changes in legislation to restrict or ban the use of certain chemicals (Way and Van Emden, 2000). Increased consumer pressure for more sustainable produce is also driving a shift away from chemical dependence (Willer and Lernoud 2016).

Consequently a system of Integrated Pest (and Disease) Management (IPM) has developed, incorporating a range of biological and cultural control techniques together with organic and inorganic chemical application to protect host plants from pests (Easterbrook *et al*., 1985; MacHardy, 2000). Under IPM, a range of ‘tools’ are employed by growers to control pest outbreaks in the orchard, from semiochemicals used in mating disruption, to phenological forecasting models which accurately predict pest outbreaks (Damos *et al.*, 2015).

Insect natural enemies are also an important part of IPM strategies, some of which, for example parasitoid wasps (Hymenoptera: Parasitica), lacewings (Neuroptera: Chrysopidae) and ladybirds (Coleoptera: Coccinellidae), are commercially bred and are available to buy and deploy directly into crops (Koppert, 2018). Such specialist natural enemies can maintain pest populations below damaging thresholds in orchards (Cross *et al*., 1999).

Generalist predatory species such as spiders (Araneae), earwigs (Dermaptera: Forficulidae) and carabids (Coleoptera: Carabidae), can also contribute to IPM programs though they are not bred commercially (Way and Van Emden, 2000). As generalist predatory species are less specific in their feeding preferences than species with more specific dietary preferences, they are potentially more resilient to habitat disturbance (Symondson *et al*., 2002). For example, the loss of a food resource, i.e. a particular pest, would not necessarily be expected to lead to the loss of a generalist predator, as that predator may be capable of sourcing food elsewhere. Many generalist species are associated with highly disturbed, agricultural land (Benton *et al.*, 2003) indicating changes in management or crop type are less likely to result in total losses of species. This flexibility and resilience makes generalist species potentially valuable service providers with regard to biological pest control (Thies *et al.*, 2003).

1.3. Orchard vegetation management

The vegetation management practices within orchards can have an impact on the abundance of natural enemies found therein. The non-crop vegetation in commercial dessert orchards is typically intensively managed, regularly mown and treated with herbicides (J. Cross, pers. comm.). Commercial apple cultivars are shallow rooted trees which suffer ill effects from competition for water and nutrients when vegetation is allowed to remain in tree rows (Granatstein and Sánchez, 2009). As a result, growers typically spray herbicide in the strip beneath the tree row. In the alleys between the tree rows, vegetation is allowed to remain, but is typically mown frequently to a short height owing to the perception that too much vegetation will lead to competition for the trees and create opportunities for some pest species (J. Fitzgerald, pers. comm.). As a result of such management, the orchard floor forms a habitat mosaic of bare ground and short sward grassland.

Such management practices are detrimental to predatory invertebrate communities, as tall herbaceous vegetation generally supports a greater abundance and diversity of arthropods while regular management is associated with lower structural complexity in the sward and reduced forb presence (Morris, 2000). Cizek *et al.* (2012) and Gobbi *et al*. (2015) found mowing intensity to influence epigeal predator communities, with a shift towards more disturbance tolerant ground beetles (Coleoptera: Carabidae), hereafter referred to as carabids, and spider species.

In contrast, increases in vegetation cover in the alleys has been found to be associated with higher numbers of natural enemies in the orchard canopy (Campbell *et al*., 2017) directly impacting pest control. However, as many orchard pests spend part of their lifecycles on the ground, they are potentially vulnerable to predation by generalist epigeal predators too (Boreau de Roincé *et al*. 2012, Cross *et al*. 2015; Unruh *et al*. 2016). As such, there is an argument for enhancing this community regardless of readily identifiable effects within the tree canopy.

Much work to enhance the orchard floor habitat for IPM has focused on the addition of species-rich seed mixes, frequently including grassland forb species, to boost numbers of pollinators and predators (Campbell, 2013; Markó *et al*., 2013). Successes in the form of increased abundances of both groups have been observed to an extent, though the impact on pest suppression is not always guaranteed (Markó and Keresztes, 2014). Treatments such as this pose an additional cost and require specific management for establishment and maintenance in the long term.

A reduction in mowing of the pre-existing vegetation within orchards may provide a simple and cheaper alternative to the sowing of expensive seed mixes, boosting predator numbers without compromising tree health. This alternative approach has not been well studied in orchards; however, reduced mowing will allow a more structurally and floristically diverse sward to develop (Morris, 2000) which in turn has been associated with increased numbers of natural enemies (Wan *et al*., 2014) which may be of benefit to growers.

1.4. Generalist predators and land use diversity

Agrochemicals are known to impact negatively upon natural enemy populations, for example earwigs (Fountain and Harris, 2015; Shaw and Wallis, 2010), and may be responsible for a large proportion of inter-farm variation observed in natural enemy abundance (Malone *et al*., 2017a). In addition, vegetation management as discussed above, will influence natural enemy communities. Nonetheless, a range of generalist species are found in agricultural habitats typified by this type of disturbance (Benton *et al*., 2003).

Commercial dessert apple orchards are relatively stable agro-ecosystems in comparison to many other crops. Crops are planted on a 15 – 20 year rotation so disturbance is lower than that experienced in arable farms, and, when unsprayed, a large and diverse arthropod fauna can establish in orchards (Cross *et al*., 2015). Chemical pest management has been associated with reductions in the abundance of beneficial arthropods found in orchards; from pollinators (Pisa *et al.*, 2014) to pest regulators; e.g. predatory Heteroptera (anthocorids, mirids) and Neuroptera (lacewings) (Solomon *et al.*, 2000), predatory mites (Fitzgerald *et al.*, 2007), spiders (Markó *et al.*, 2009), coccinellids (Dib *et al*., 2016b), the European earwig *Forficula auricularia* (Fountain and Harris, 2015) and carabids (Hedde *et al.*, 2015).

Farm management practices typically associated with intensive systems undoubtedly have detrimental effects on arthropod communities in agricultural systems. Increasingly research is demonstrating an influence of the diversity of surrounding land use on arthropod abundance and diversity at the farm scale, and the effects of this on the ecosystem services and dis-services provided therein (Lichtenberg *et al.*, 2017). Numerous studies show this relationship with pollinators (e.g. Kennedy *et al*., 2013) and natural enemies (e.g. Rusch *et al*., 2016). The majority of this work has focussed on arable systems, with little work investigating these effects in orchard environments. Increases in the proportion of intensively managed agricultural land in a landscape is associated with reduced abundances of natural enemies and altered functionality within predatory communities. Given the weight of research showing land use management intensity and diversity effects within farm ecosystems, it is considered likely that the arthropod communities in commercial apple orchards will be influenced to a degree by surrounding land use. However, the semi-permanent nature of the crop might mitigate these effects. Regardless of this, numbers of generalist predators have been consistently shown to be associated with land use diversity at landscape and local scales, which has knock-on implications for pest suppression within farms (Chaplin-Kramer *et al*., 2011; Rusch *et al*., 2016).

Unlike in arable systems where carabids have been well studied, carabids found within orchards are poorly studied though they have been recognised as potential contributors to IPM (Way and Van Emden, 2000). In contrast earwigs have been studied more often in orchards though understanding of the causes of inter-orchard population fluctuations is still lacking (Cross *et al*., 2015). While land use is known to influence natural enemies, the relationship between land use and predatory carabids and earwigs in dessert apple orchards is poorly understood. These knowledge gaps make both groups attractive subjects for study.

1.5. Carabids and earwigs in apple orchards

The population dynamics of the European earwig in fruit crops are relatively well understood. These voracious predators (Unruh *et al*., 2016) were once considered pests themselves, as fruit damage caused by other insects was often wrongly attributed to earwigs (Fitzgerald *et al.*, 2010). Although omnivorous, when the earwig diet was assessed no evidence of apple tissue was found (Fitzgerald *et al*., 2010). However, evidence of predation on pest species has been demonstrated using gut contents analysis (Romeu-Dalmau *et al*., 2012).

Beyond chemical pest management, the drivers of inter-orchard variation in earwig densities are not fully understood and variously attributed to factors such as farm management, climatic conditions, food availability, predation and disease (Helsen *et al.* 1998, Gobin *et al*. 2008, Moerkens *et al*. 2012, Lordan 2014, Dib ***at al.*** 2016). A limited amount of work has been documented specifically exploring the influences of land use diversity at a large scale, or the influence of the neighbouring land use type on *F. auricularia* abundance in orchards (e.g. Stutz and Entling, 2011). The negative geotropism, aggregation tendencies and nesting habit of earwigs (Philips 1981, after Atwell 1927) would suggest a preference for arboreal living. As such, it is considered that this predator may be influenced by the diversity of land use surrounding orchards, as has been documented for a range of other predators in agricultural settings (Chaplin-Kramer and Kremen, 2012). However, the dispersal range of *F. auricularia* is relatively low, with a maximum value of 29 m recorded in one study (Moerkens *et al*., 2010). As such, the influence of land use diversity may not be apparent at spatial scales above 30 m. Nonetheless, understanding if land use diversity at a landscape scale and/or local scale influences orchard earwig populations would provide valuable information on the ecological preferences of this important orchard predator, which may be of benefit to growers.

Carabid beetles are less well studied compared to other predatory groups in orchards, overlooked as predators to a degree as they are ground dwellers. Although epigeal, this largely predatory group is likely to contribute to IPM, particularly for those pests that spend part of their life cycles on the ground (Cross *et al*., 2015). Certain carabid species have been assessed in terms of their pest control potential but results have been inconclusive; gut contents analysis has proved positive for certain pests (Boreau de Roincé *et al.* 2012), though suppression of populations was not observed in the field (Marliac *et al*., 2015b). However, an exploration of the influence of an orchard carabid assemblage as a whole has not been undertaken. The functional diversity of carabid assemblages is known to be influenced by land use diversity, adjacent habitats and farm management (i.e. Epstein *et al.*, 2001; Woodcock *et al.*, 2007, 2014). The functional traits present within communities has knock on implications for ecosystem services. As such, a broader understanding of the functional composition and structure of the orchard carabid community in relation to farm management practices, and land use factors, would provide a clearer indication of the value of carabids to orchard IPM and what strategies could be employed to exploit that.

1.6. Research aims and thesis structure

The overall purpose of the research reported in this thesis is to investigate the influences of farm management and land use diversity at landscape and local scales on generalist predators in commercial dessert apple orchards. The work focuses primarily on predatory carabids and the European earwig. Both are considered likely to be influenced by orchard management and surrounding land use, which may be a good predictor of abundance within farms.

The initial part of this work (Chapter 2) is concerned with understanding land use diversity and enabling site selection. This portion of the work aims to first establish the relationship between land use diversity and the habitats found in a given landscape. The second aim of this section is to establish the diversity of the land use surrounding potential study farms to allow stratification of a sample to cover the range of land use diversity present in the area, enabling selection of appropriate sites for more in-depth study.

The main body of the thesis relates to the relationships between land use diversity, farm management and the focal natural enemies; carabids and earwigs. Chapter 3 has three main aims; firstly, to quantify the influences of land use diversity on the community structure and function of orchard carabids; secondly, to understand the influences of chemical pest regulation on carabid community structure and function and thirdly, to assess the relationship between carabid communities and pest regulation within study orchards.

In Chapter 4, orchard earwig populations are studied. This section of the work has three main aims; firstly, to quantify the influences of land use diversity on the abundance and population dynamics of earwigs within study farms; secondly, to understand the influences of chemical pest regulation on orchard earwig populations and thirdly, to assess the relationship between earwig populations and pest regulation within study orchards. An additional, secondary aim of this section of the work compares the efficacy of alternative earwig sampling techniques.

The final part of the thesis concerns orchard vegetation management. The aim of this section is to compare the influence of different mowing regimes on communities of generalist predatory arthropods both in the tree canopies and on the orchard floor in a dessert apple orchard (Chapter 5).

The thesis closes with a discussion on the findings of this work, drawing together the results and implications of the data chapters. Limitations of this study are discussed, and potential avenues of future research are highlighted. Recommendations on the practical implications of this work to improve natural pest control within orchards are made.

Chapter 2 Defining land use diversity at landscape and local scales

2.1. Introduction

The abundance and diversity of species in agro-ecosystems varies spatially (Lüscher *et al*. 2014 & 2015). Increasingly, research is finding that the 'complexity' or 'diversity' of land use can influence arthropod abundance and diversity at the farm scale and the ecosystem services and dis-services provided (Lichtenberg *et al*. 2017).

Agricultural intensification tends to produce more simplified landscapes owing to the economic need for homogenisation within farms to minimise costs, maximise profitability and production and facilitate management, thus reducing the number and types of habitats present (Tscharntke *et al*., 2012). As such, landscape 'complexity' or 'diversity' is often related to the intensity of management associated with different land uses; agricultural land, being regularly subjected to physical and chemical management will be at the high end of the management intensity scale. Habitats such as woodland, which are rarely if ever chemically or physically managed or harvested, are at the low end of the management intensity scale.

Diversity in a landscape context is a simple metric of the number of different habitat types present in a given landscape and the relative proportions of each (Steffan-Dewenter, 2003), whilst in the studies of the agricultural landscapes of northern Germany from the Gottingen group, complexity is taken to mean a high proportion of non-crop habitat in a landscape (Rand *et al*., 2012). Complex landscapes in the agricultural areas of north western Europe are generally considered to support >20% non-crop habitat (Tscharntke *et al*., 2012).

The complexity of a landscape can be defined in a number of ways which can be broadly considered as either measures of structural complexity or functional complexity (Chisholm *et al.*, 2014). Using functional complexity, habitats in a landscape are classified with regards to their functionality for a species or group; acting as either sources of migrating populations, sinks drawing populations in, or as a neutral resource for the species of interest (Chisholm *et al.*, 2014). Habitats can be categorised in terms of the resources provided for the species under consideration; e.g. food, nesting, shelter or some other opportunity which will support the species of interest at various life stages. The use of functional complexity is applicable where the ecological requirements of the study species are well understood, when certain habitats/habitat features are of high importance to the species and where distinct functional land parcels can be identified (Chisholm *et al.*, 2014). Such an approach is common in pollination studies for example, where floral resources are demonstrably important and species requirements are generally well understood (Shackelford *et al.*, 2013).

Structural complexity is more commonly employed where the precise nature of the relationships between study species and landscapes are not fully understood (Chisholm *et al.*, 2014). Structural complexity focuses on characterising the landscape in terms of the number and type of habitats present and has been used by a number of authors investigating land use-invertebrate associations. In this context, structural complexity can be characterised in a number of ways, for example by establishing diversity indices for land use within a landscape (Gardiner and Neal, 2009), by simply measuring the proportion of semi-natural habitats versus the proportion of agricultural habitats in a landscape (e.g. Chaplin-Kramer and Kremen 2012; Rusch *et al.* 2012; Schüepp *et al*. 2014) or by defining discrete categories relevant to a given study, such as a specific crop type (Monteiro *et al.*, 2013).

There is some risk that this method of characterising landscape complexity can oversimplify a landscape, as within habitat variations are not always considered. For example, considerable variation in the type of agriculture present in a landscape might cause a mismatch between assumptions based on management intensity and the actual level of disturbance experienced in that habitat type. Orchards are a good example of this; as perennial crops, levels of disturbance to the habitat, in other words physical management intensity (e.g. tillage), will be considerably different to that experienced in arable land, though the frequency and rate of agrochemical applications may be the same or indeed greater. Nonetheless, the structural classification of landscapes has been found to correlate well with metrics of associated species such as the Shannon-Weiner index and predator-pest relationships within farms (Grab *et al.*, 2018).

There are numerous studies which have investigated the land use-invertebrate relationship for pollinator and natural enemy groups. Solitary bees, bumblebees and honey bees were found to be associated with the amount of semi-natural habitats in a landscape (Steffan-Dewenter *et al.*, 2002). Likewise, predation was greater in the crop in landscapes characterised by a larger proportion of non-crop habitats (Chaplin-Kramer and Kremen, 2012; Simons *et al.*, 2016). Simons *et al*. (2016) concluded that highly managed land uses 'select for smaller, more mobile, and less specialized species across taxa'. Gámez-Virués *et al*. (2015) agreed with these findings, suggesting that arthropod specialisation is reduced overall and traits associated with lower activity periods and increased dispersal ability are more common in highly managed landscapes. Such shifts lead to community homogenisation and a reduction in pest suppression (Woodcock *et al.*, 2014). A number of meta-analyses and reviews have drawn together the research and found an overarching trend; greater management intensity and landscape simplification leads to a reduction in ecosystem services in crop habitats (e.g. pollination services in Kennedy *et al*. 2013; pest control in Rusch *et al.* 2016). The effects of land use 'complexity' or 'diversity' in relation to focal species in this study is discussed further in chapters three and four.

Investigations of the land use-invertebrate relationship require a consideration of scale, as the appropriate scale to consider will vary according to species (Schellhorn *et al*., 2014; Thies *et al.*, 2003). Research has assessed land use diversity at distances from sampling sites ranging from 100 m to 6 km or more, with 1 km the most commonly studied range. In the study of Rusch *et al*. (2013a and b) land use diversity at the 0.5 km – 1 km scale was most useful in predicting natural pest control. Owing to these findings, land use diversity in this study was characterised at the 1 km scale.

In addition to landscape level 'complexity' or 'diversity', the composition of adjacent land uses can also be a determinant of invertebrate communities within crops. Ernoult *et al.* (2016) found natural crop protection benefitted from adjacent, non-crop woody vegetation. Similarly, Miliczky and Horton (2005) demonstrated this effect in orchards, where distance from neighbouring non-crop habitat determined the communities recorded within the orchard to a degree. The influence of neighbouring habitats on target species in this study is considered further in chapters three and four. As the invertebrate communities found within an orchard are likely to be affected by the composition of the land use immediately surrounding it, this was also considered when choosing farms for study.

2.2. Aims

This chapter aims to first establish the relationship between the habitats present in a given landscape and measures of land use 'diversity'. The second aim of this chapter is to establish the context of the land use surrounding potential study farms. This will enable the selection of farms in high and low complexity/diversity settings to allow an exploration of the influences of land use on carabid and earwig populations in study orchards.

2.3. Methodology

For the purposes of this exercise a large number of growers were approached for advice and to obtain farm planting plans and maps. The most commonly grown variety amongst respondents was cv. 'Gala', while 1 ha was considered to be the minimum commercially viable plantation size. As such, only orchards growing this variety and being larger than 1 ha in extent were selected for further investigation.

2.3.1. Land use data extraction

In order to identify study farms in either simple or complex landscapes, it was necessary to define the land use surrounding prospective study farms. In this study a 'farm' represents the land-holding of a single producer, comprising numerous individual plantations. The diversity of land use surrounding prospective study farms was defined by extracting data on each discrete habitat parcel from the 'Landcover 2007 Mastermap' (LCM) (Morton *et al.*, 2011). This was done using ArcGIS 10.2 (ESRI, 2013). The extracted data included information on the land use classes present and the area of each discrete parcel at each geographic location. That data was used to calculate the proportion of different land use and habitat classes present within a 1 km radius from the centre of each potential study farm. Each potential study farm contained three potential study orchards or 'plots', i.e. three distinct blocks of c.v. Gala ≥1 ha in area.

The LCM was developed by the Natural Environment Research Council (NERC) and comprises twenty-three land cover classes based on the UK's terrestrial Broad Habitats (Morton *et al.*, 2011). These Broad Habitats were defined as part of the development of the UK Biodiversity Action Plan described by the Joint Nature Conservation Committee, JNCC (Jackson, 2000). Where appropriate, these Broad Habitats classes are further divided into a number of 'Sub-Habitats'. A list of the Broad and Sub Habitat classes which were recorded during this study is provided in Table 2.1 in the results section of this chapter.

For the purposes of this study the LCM landcover class data were manipulated slightly. The LCM dataset has an accuracy of 83% (Morton *et al.* 2011), however due to the nature of the orchard habitat, i.e. scattered trees over grassland, this habitat is often wrongly included within other land cover classes, such as Parkland (Jackson *et al.*, 2000). Owing to this, and to maintain the focus of this study on orchards, the LCM data was supplemented with data obtained from growers, Ordnance Survey mapping and the most recent aerial photography available from Google Maps™ and Google Earth™ (ranging from 2006 – 2013). Using this additional information, any land use parcels known to comprise orchard, and misclassified under the LCM, were removed from the LCM habitat classes in which they previously fell and were re-classified to form a new 'Orchard' Habitat class.

2.3.2. Land use classification at the landscape scale

The land use surrounding potential study orchards was characterised at two scales in this study. At the landscape scale, land use data were summarised for a circular area of radius 1 km stretching from the central point of each potential study farm. The classification of land use at the landscape scale was established in two ways. In the first instance, the Shannon–Wiener habitat diversity index (Steffan-Dewenter, 2003) was used as a means of defining the diversity of land use surrounding potential study farms, where H_s is the diversity of all habitat types and Pi is the proportion of each land use type:

$$H_{Landscape} = -\sum Pi.\ln Pi$$

Shannon-Weiner habitat diversity (SDI*h*) is based on the number of habitats recorded (richness) and the relative abundance (evenness) of these habitats. SDI*h* calculations were performed on each potential 'study farm' using data for both the broad habitat classes and sub-habitat classes.

Secondly, each habitat in a given study area was considered in terms of the intensity of the agricultural management typical for such a habitat. The areas of habitats associated with intensive management (i.e. arable, improved grassland, orchards) in a study area were combined to give an overall proportion of productive agricultural land in the surrounding landscape. The same was done for the proportion of 'unmanaged' (or less managed) habitats in a landscape such as woodlands or unmanaged grasslands. See Table 2.1 for a full list of all habitat classes which were recorded in this study.

2.3.3. Land use classification at the local scale

At the local scale, land use identity was incorporated into the study by selecting study orchards within 25 m or less of:

i. Other orchards
ii. Arable or improved grassland
iii. Unmanaged or less managed habitats

All statistical analysis was undertaken using R statistical software (version 3.3, R Development Core Team 2016). The ‘lme4’ package (Bates *et al*. 2016) was used to undertake general linear models comparing SDI*h* values with land use classes.

The data were analysed in relation to the SDI*h* values to ascertain if any relationships were evident between either the combined, management based land use classes or individual habitat types as defined in the LCM, and the overall diversity of land use in a landscape at the 1 km scale.

2.4. Results

Twenty-six potential ‘study farms’ were assessed following the methodology discussed above. Following the reorganisation of the habitat data, 14 Broad Habitat classes and 17 Sub-Habitat classes occurred within the 1 km radius landscapes assessed surrounding potential ‘study farms’, see Table 2.1.

Table 2.1: The Broad Habitat classes and Sub-Habitat classes recorded within the landscapes within 1 km of the centre of the 26 potential study farms assessed.

Broad Habitat Classes	**Sub Habitat Classes**
Arable, horticulture, improved grassland *(i)*	Arable unknown *(i)* Improved grassland *(i)*
Orchard *(i)*	Orchard *(i)*
Acid grassland	Acid grassland
Broad leaved, mixed and yew woodland	Deciduous woodland Mixed Felled woodland
Built up areas and gardens	Suburban
Coniferous woodland	Conifer woodland
Freshwater	Lake
Inland rock	Inland rock
Littoral sediment	Saltmarsh
Neutral grassland	Neutral grassland
Rough low-productivity grassland	Rough low-productivity grassland
Salt water	Sea water
Urban	Urban
Urban industrial	Urban industrial Despoiled land

(i) refers to a land use classed as 'intensively managed', otherwise land use is considered 'unmanaged'

Using the SDI*h* values, land use diversity at the landscape scale was characterised for all twenty-six potential 'study farms' with farms SDI*h* values ranging from between 0.56 to 1.49 for Broad Habitats and 1.09 to 1.88 for Sub-Habitats.

Analysis showed significantly lower SDI*h* scores for Broad Habitat classes where the proportion of 'managed' habitats, (arable, improved grassland, orchard) was high in a landscape (r = -.696, $p \leq 0.001$). Conversely, SDI*h* scores were found to be significantly higher where cover of 'unmanaged' habitats was greater (r = .795, $p \leq 0.001$). These results suggest a strong correlation between land use diversity and the proportion of intensively managed agricultural land in the landscape, Fig. 2.1.

The potential study farms were ranked using the values for each of the three assessments, providing a classification of land use according to SDI*h*, proportions of 'managed' habitats and the proportions of 'unmanaged habitats' in the surrounding landscape. Farms with low SDI*h* scores generally exhibited high proportions of arable/improved grassland (i.e. 'managed' habitats) in the surrounding landscape, while farms with higher SDI*h* scores were generally associated with a greater proportion of semi-natural (i.e. 'unmanaged') habitat, see Table 2.2.

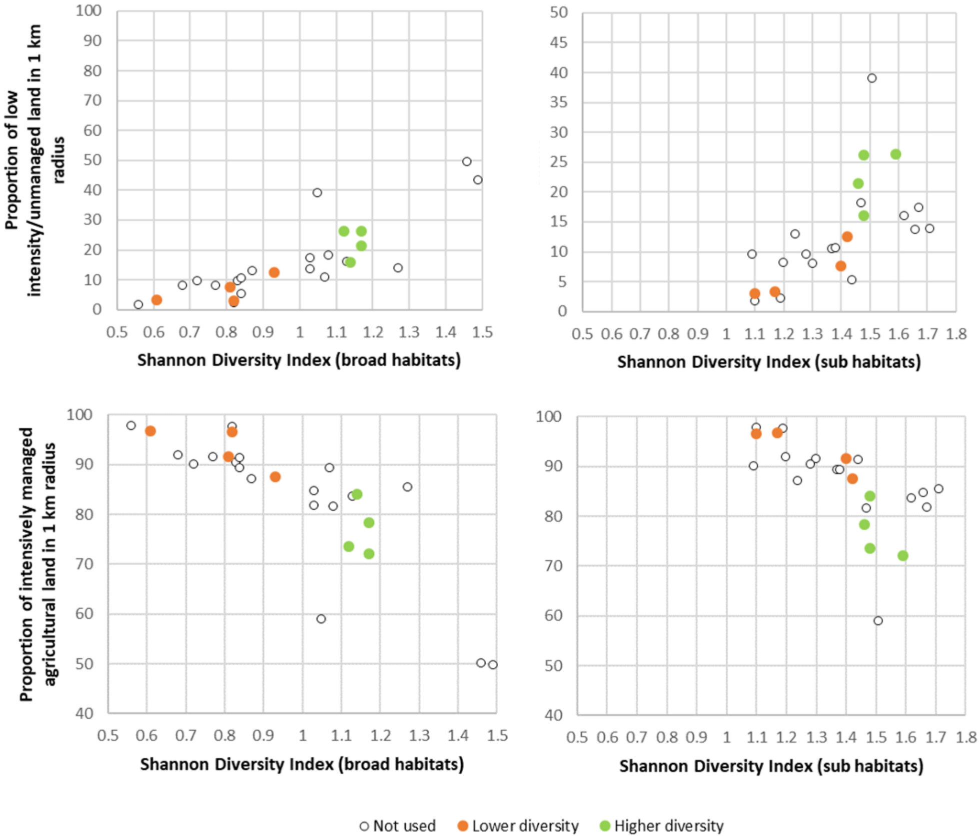


Figure 2.1: The location of study farms, showing land use diversity classification and location within the Natural Character Areas that have significant areas of commercial apple orchards in Kent (North Kent Plain, Wealden Greensand and Low Weald).

Table 2.2: The 26 potential study farms ranked according to SDIh values and showing the proportion of managed and unmanaged habitats present within surrounding landscape to a radius of 1 km. Those farms listed in bold were selected for further study.

Potential Study Farm	Shannon Diversity Index		Management Intensity *(i.e. proportion of land use type in a radius of 1 km)*	
	Broad habitats	Sub-habitats	**High**: arable, improved grassland, orchard	**Low**: wood/scrubland, unmanaged grassland, suburban
Owens	0.56	1.10	97.76	1.74
West_Pike	0.61	1.17	96.71	3.29
Bockfold	0.68	1.20	91.81	8.19
Gibbens	0.72	1.09	90.01	9.58
Stallance	0.77	1.30	91.62	8.07
Batteries	0.81	1.40	91.64	7.68
Paramour	0.82	1.19	97.53	2.21
Santon	0.82	1.10	96.62	2.99
Combourne	0.83	1.28	90.41	9.59
Howt_Grn	0.84	1.44	91.26	5.31
Gore	0.84	1.37	89.39	10.42
Marshalls	0.87	1.24	87.05	12.95
Griffins	0.93	1.42	87.53	12.47
FeldFarm	1.03	1.66	84.80	13.69
Ratcliffe	1.03	1.67	81.73	17.38
Ufton	1.05	1.51	59.03	38.99
Broadwater	1.07	1.38	89.33	10.67
Rob_Mitch	1.08	1.47	81.64	18.06
Loddington	1.12	1.48	73.53	26.25
Sheerland	1.13	1.62	83.63	16.04
Orch_Lodge	1.14	1.48	83.99	16.01
HMounts	1.17	1.59	72.19	26.34
Denstead	1.17	1.46	78.29	21.48
High_Crt	1.27	1.71	85.40	13.85
Bounds	1.46	1.88	50.18	49.32
Pump_Farm	1.49	1.88	49.73	43.41

2.5. Site selection

As SDI*h* scores and the intensity of agricultural management, are correlated, study farms were ultimately selected on the basis of their SDI*h* scores. Every potential study farm with a Broad Habitat SDI*h* value greater than 1 is considered to be on the higher end of the land use diversity scale, while those below 1 are considered to be on the lower end, Fig. 2.1. Henceforth, the term 'land use diversity' will refer to the land use classification of a study farm or plot.

Site selection was constrained by the need to ensure that study plots had a range of neighbouring land use types, as well as the availability of appropriately sized c.v. Gala plantations (i.e. 1 ha minimum). Consequently, a large number of potential study farms were considered unsuitable for inclusion in a comparative study and excluded from further work.

Of the 26 farms assessed, 4 study farms were selected from the lower end of the SDI*h* values recorded, and 4 from the higher end of the SDI*h* scores recorded, see Table 2.2. Farms were paired in order to reduce geographic variability, with each pair consisting of an orchard in a highly diverse land use setting and one in a lower diversity setting. Study farms were selected at eight locations in the Natural Character Areas that have significant areas of commercial apple orchards in Kent (North Kent Plain, Wealden Greensand and Low Weald), Fig. 2.2.

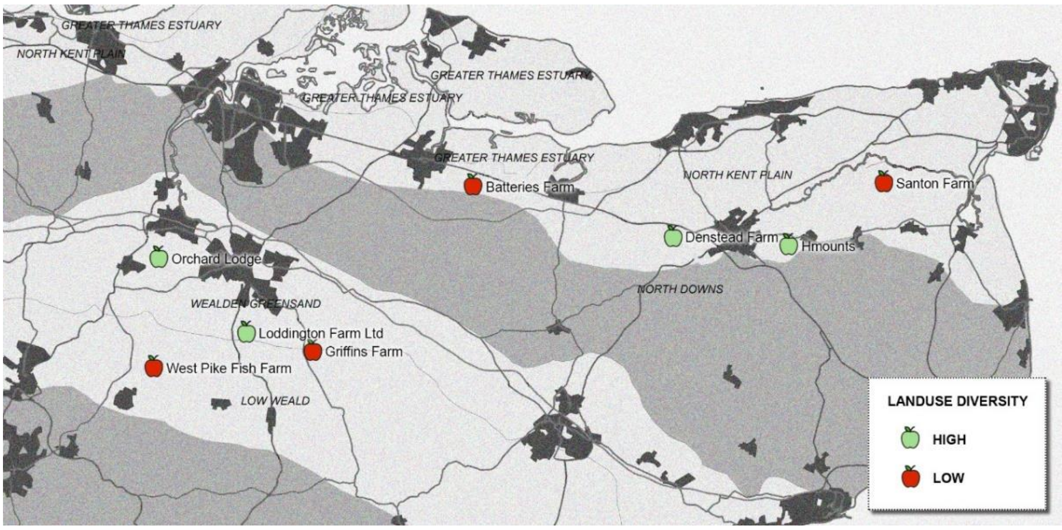


Figure 2.2: *The location of study farms, showing land use diversity classification and location within the Natural Character Areas that have significant areas of commercial apple orchards in Kent (North Kent Plain, Wealden Greensand and Low Weald).*

2.6. Site layout

Analysis indicated correlations between SDI*h* scores and the proportion of ‘managed’ and ‘unmanaged’ land surrounding farms. The proportion of intensively managed agricultural land (arable, improved grassland and orchard) and infrequently or un-managed habitats such as woodland were particularly correlated with low and high SDI*h* values respectively. Consequently, representative habitats from each of these management classifications have been selected as focal habitats to investigate the influence of land use at the local scale (i.e. the neighbouring land use adjacent, i.e. within 25 m, to each study plot).

Each of the selected farms contained three distinct 1 ha study plots situated within 25 m of one of the adjacent focal habitats. Each study plot began on the first row of orchard planting, such that they were located between 4 m and 25 m from the adjacent habitat. The boundary vegetation typically comprised a grass access strip subject to the standard orchard management regime, beyond which was variably a fence, windbreak, hedgerow or, in the case of study plots adjacent to orchard, additional orchard planting. All three study plots occurred within the same farm but were separated by a minimum distance of 250 m. Henceforth these adjacent land use types will be referred to as ‘agricultural’ (i.e. arable land or improved grassland), ‘orchard’ and ‘unmanaged’ (typically woodland), Fig 2.3.

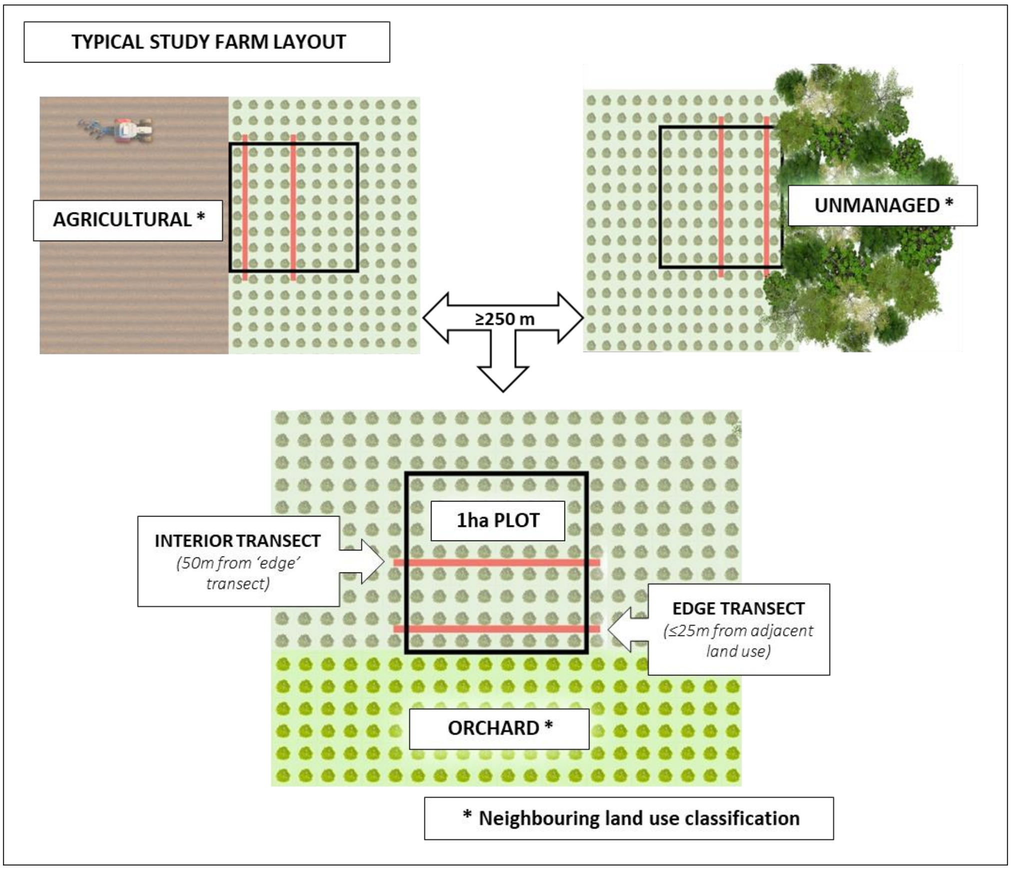


Figure 2.3: *An illustration showing the typical layout of study farms and transects in relation to local land use. Each farm contained three distinct 1 ha study plots separated by ≥ 250 m; one 'adjacent to' (i.e. 1*st *tree row within ≤25m of) agricultural land, one 'adjacent to' unmanaged land and one 'adjacent to' orchard planting.*

The methodology outlined here has allowed study plots to be selected on the basis of the diversity of land use in a 1 km radius surrounding each, and according to the classification of the neighbouring land use within < 25 m of the plot. Two study transects were set-up within each plot to allow a comparison of arthropod communities at two distances from the adjacent focal habitats (Fig 2.3).

The focus of this study was to investigate the influence of land use diversity at a 1 km scale and at a local scale on the abundance and diversity of natural enemies in orchards. As such, smaller habitat parcels such as windbreaks and mown grass strips, which are commonplace on the margins of UK apple orchard plantations, have been deliberately excluded from investigation.

The planting date of each study plot and the neighbouring land use are listed in Table 2.3. All study plots are planted with cv. 'Gala'. It became apparent part way through study that a small number of cv. ‘Discovery’ and cv. ‘Windsor’ were present in the ‘unmanaged’ plot of one of the farms, Orchard Lodge.

Table 2.3: The planting dates and neighbouring land use type and classification of each study plot

Farm	**Study plot planting year and neighbouring land use**				
	Orchard	Agricultural		Unmanaged	
Batteries	1998	1997	Arable	1985	Woodland
Denstead	1995	2013	Improved grassland	2001	Woodland
Griffins	2012	2012	Arable	2013	Woodland
HMounts	1992	1994	Arable	1995	Mature wooded garden
Loddington	2008	2012	Improved grassland	1998	Woodland
Orchard Lodge	1998	1991	Improved grassland	1995	Woodland
Santon	1994	2002	Improved grassland	1992	Mature wooded garden
West Pikefish	1988	1988	Improved grassland	1993	Woodland

Chapter 3 The community composition and pest regulation potential of Carabid beetles (Coleoptera: Carabidae) in commercial apple orchards: the influences of land use diversity and farm management practices on community structure and function.

3.1. Abstract

Arthropod communities in agricultural habitats are known to be influenced by a range of biotic and abiotic factors, from geographic location to farm management and food availability. Many researchers have demonstrated a link between the heterogeneity of land use and the abundance and structure of carabid communities in agricultural systems, though few studies have investigated this effect in orchards. This study explores the influences of land use diversity and farm management on carabid community structure and function, and the related pest control potential in commercial dessert apple orchards.

The findings indicate that land use diversity at a landscape scale or adjacent land use type are not important determinants of overall carabid activity-density in commercial dessert apple orchards. However, land use factors were found to influence community composition and the representation of traits in the community. Land use diversity at the landscape scale was found to affect the community weighted means for size, diet and breeding with higher proportions of larger, polyphagous species and fewer spring breeders in more diverse land use settings. Diurnal activity and habitat associations were found to be influenced by adjacent land use type.

The effects of chemical pest management on the carabid community were mixed; greater chlorpyrifos use was associated with higher activity-density as well as an increase in the proportion of polyphagous species in the community, though functional diversity was found to decline. Thiacloprid use was associated with increased functional diversity of response traits and a decrease in the proportion of larger carabids in the community.

A greater proportion of polyphagous species in the community was associated with increased predation in this study. The communities observed appear to be typical of agricultural carabid communities observed elsewhere; dominated by disturbance tolerant species. Nonetheless, community structure and function such as that recorded here is associated with opportunistic predation by generalist predators, and thus the potential provision of pest control by carabids in orchards. Maintaining areas of unmanaged habitat such as tussocky grass and woody vegetation within or around the orchard will increase opportunities for carabids and other beneficial species and likely support natural pest regulation.

3.2. Introduction

The carabid beetles (Coleoptera: Carabidae), hereafter referred to as carabids, are a highly diverse family of beetles, common and widespread across a broad range of managed and unmanaged habitats around the world (Holland, 2002). Generally omnivorous, carabids are largely represented by polyphagous, opportunistic species, with fewer phytophagous or carnivorous specialists (Larochelle, 1990; Lövei and Sunderland, 1996). Movement is dictated by breeding, foraging and climatic/habitat requirements and is usually undertaken in adulthood, as this holometabolous family have little or no dispersal ability as larvae (Lövei and Sunderland, 1996). The lifespan is usually between 1 and 2 years, though larger species, and species which breed in autumn tend to live longer (Lövei and Sunderland 1996). In agricultural habitats, overwintering generally occurs in non-crop habitats (Collins *et al*., 2003) where conditions are more favourable (Lövei and Sunderland 1996).

Carabids are known to contribute to natural pest control in agricultural systems, consuming a variety of pest arthropods, though evidence suggests this vital ecosystem service is mediated by landscape factors and affected by farm management (Kromp, 1999; Symondson *et al.*, 2002).

Studies of the epigeal communities within orchards are underrepresented in the literature and although orchards could be expected to support carabid assemblages contributing to natural pest control, there has been little work in this area.

3.2.1. Carabids in orchards

The orchard understory is typically a highly managed environment owing to the practices of applying herbicide under tree rows to reduce competition between trees and ground vegetation, and tightly mowing (sward height 9 cm on average, pers. obs.) the inter-row alleys to reduce opportunities for pest species (Granatstein and Sánchez, 2009). Orchard floor management is likely to influence carabid communities within orchards as mowing has been found to affect predator communities in other systems (e.g. Cizek *et al*. 2012; Gobbi *et al.* 2015). Taller grassland generally supports a greater abundance and diversity of epigeal arthropods (Morris, 2000). Further, where a structurally and botanically diverse sward has been allowed to develop in orchards, carabid activity-density and diversity increased (Marko and Kadar 2005). Beyond the initial impacts of injury or death, mowing is a non-selective form of vegetation management that will reduce vegetative structure and refugia for epigeal arthropods in the sward (Morris, 2000).

The discordant nature of orchard habitats; i.e. an intermittent canopy of trees over a herbicide-treated understorey interspersed with mown turf, will drive considerable spatial variation in arthropod communities. The epigeal carabid assemblage typically found in orchards in Europe is characterised by a small number of species relatively common in European agro-ecosystems (Kutasi *et al.*, 2004; Letardi *et al*., 2015). In contrast, very few carabids are found in orchard trees, particularly in the UK (Luff, 2007). In an assessment of the Coleoptera occurring in apple and pear canopies in Hungary, Marko *et al.* (2010) identified just four species of carabids via canopy sampling, all of which were small and largely saproxylic.

Predatory carabids are not typically found in fruit tree canopies and as such are often overlooked as potential pest control agents in orchards where the pests are arboreal. However, interspecific effects of predation may force pests to drop from the canopy (Losey and Denno, 1998), while many key orchard pests spend part of their lifecycles on the ground, making them potentially vulnerable to epigeal carabid predation, for example; woolly apple aphid *Eriosoma lanigerum* (Hemiptera: Aphididae), rhynchites weevil *Tatianaerhynchites aequatus* (Coleoptera: Curculionidae), apple leaf midge *Dasineura mali* (Diptera: Cecidomyiidae), apple sawfly, codling moth *Cydia pomonella* (Lepidoptera: Tortricidae) and some other tortrix moths (Boreau de Roincé *et al.* 2012, Cross *et al.* 2015; Unruh *et al*. 2016).

3.2.2. Carabids and insecticide usage

Orchards are one of the most intensively sprayed cropping systems in temperate regions (Simon *et al.,* 2010). Despite this, there is evidence of a trend in agro-ecological research indicating that crops most heavily subjected to insecticide use (i.e. fruits and vegetables) are underrepresented in the research into the effects of these products on natural pest control (Mall *et al*., 2018).

Epstein *et al.* (2000 & 2001) and Hedde *et al.* (2015) have looked at the impacts of crop protection strategies on carabids in orchards and found lower activity-density and altered community composition with increased pesticide usage. The results of the Hedde *et al*. (2015) study suggested that summer breeders were more at risk from the effects of pesticides as their breeding period, a vulnerable life stage, coincided with the greatest amount of pesticide use in the orchard. These kinds of impacts would suggest insecticide usage may select for certain species/traits in a community, for example those which breed outside of the main spraying season in a given crop. Hedde *et al*. (2015) found carabid-activity density to be largely unaffected by chemical pest management, suggesting that reductions in the numbers of some species may be counterbalanced by increases of more resistant ones. Using a traits-based approach however, this study found species' morphology, habitat and feeding preferences were influenced by chemical use. Hanson *et al.* (2016) found reduced size and greater dispersal abilities under increased management suggesting these attributes compensated for the intensity of local management. Indeed, Letardi *et al.* (2015) speculated that the assemblage found in Italian orchards was typical of those found in arable farms, comprising 'disturbance – tolerant' species.

As well as anticipated effects of insecticides on target and non-target invertebrates, Malone *et al.* (2017) identified links between plant growth regulators and fungicides and the inter-orchard variation in arthropod communities. The need to regulate tree development and combat diseases like mildew mean both are employed regularly in conventional apple orchards. The results were largely attributed to knock-on effects to the orchard ecosystem brought about by decreases in overall food supply or temporal mis-matches in prey availability.

3.2.3. The influence of land use diversity on invertebrates

When looking at invertebrates in agricultural habitats in relation to characteristics of the surrounding landscape, consideration of scale is important (Tscharntke *et al.*, 2012). The appropriate scale will vary according to the species or group under consideration. For example, Thies *et al*. (2003) found that the scale at which the surrounding landscape was characterised was not related to trophic level as anticipated. However, the rates of herbivory and parasitism were affected differently according to the scale at which land use was categorised. In turn, Wamser *et al.*, (2011) found carabids with greater dispersal ability responded to variations in land use at a coarser scale than those less vagile species recorded in the study.

Lüscher *et al.* (2014 and 2015) suggest that biological abundance and diversity across a range of taxa in agricultural settings varies within geographical region and according to chemical input. Indeed, geographic regions can be expected to support differing species pools owing to variations in management, climate and soil conditions and as such, large scale geographic variation is a major determinant of species distribution (Holzschuh *et al.*, 2007). Despite this, there is an increasing body of evidence suggesting that the diversity of land use at a sub-regional, landscape scale does in fact influence arthropod abundance and diversity at the farm scale, and the ecosystem services and dis-services provided therein (Lichtenberg *et al.*, 2017). Numerous studies show this relationship with pollinators and natural enemies; Kennedy *et al*. (2013) provide a meta-analysis in relation to pollination services and Rusch *et al.* (2016) do likewise for pest control.

With reference to land use diversity in agricultural landscapes, ‘homogeneity’ is generally associated with a higher proportion of managed agricultural habitat and a lower proportion of natural/semi-natural habitats in a landscape (Rusch *et al.,* 2016). Increased landscape homogeneity is associated with lower natural pest control within crops (Chaplin-Kramer and Kremen, 2012; Rusch *et al.*, 2016). This idea has been expanded somewhat to reflect the diversity and size of managed habitats in a landscape rather than simply the degree of management. Again, increased heterogeneity of farmed land has been found to promote predator abundance and pest control at the field scale (Bosem-Baillod *et al.,* 2017; Redlich *et al.*, 2018).

The scales employed to assess land use diversity in relation to ecosystem services in other studies range from 100 m (Schüepp *et al.*, 2014) through to 6 km (Thies *et al*., 2003), though Rusch *et al*. (2013a and 2013b) found land use diversity at the 0.5 km – 1 km scale was most useful in predicting natural pest control.

Adjacent land use is also an important factor driving arthropod movement into and out of the crop (Duflot *et al.*, 2018; Ernoult *et al*., 2016). Spill-over effects, i.e. “the dispersal or foraging of organisms across habitat borders” (Schneider *et al*., 2013) would suggest that non-crop habitats act as a source or sink for both pests and predators, and consequently the non-orchard habitats surrounding an orchard plantation are likely to influence the arthropod communities within it. The composition of the adjacent land use has been shown to be a potential determinant of pest/predator movement (Miliczky and Horton 2005; Ricci *et al.* 2009; Woodcock *et al.* 2010; O’Rourke *et al*. 2011; Rusch *et al*. 2012; Monteiro *et al.* 2013). Detecting an influence of adjacent land use will be dependent on the dispersal tendencies of the species in question; while some naturally migrate to and from non-crop to crop habitats during their lives (‘ecotone’ species or ‘dispersers’), others will persist in the crop more permanently (‘cultural’ species). For the former, improved resources may boost numbers or movement into the crop, but for the latter the effects may be neutral (Duelli and Obrist, 2003)

In orchard systems the same influences have been observed, with greater abundance of natural enemies recorded as proximity to native, non-orchard land use increases (Miliczky and Horton, 2005). In a cherry orchard, Schüepp *et al*. (2014) found that the level of isolation from non-crop habitats at a local scale (≤ 100 m) was a determinant of herbivory and natural enemy influx to a cherry orchard, while landscape level heterogeneity was not an influencing factor.

The efficacy of carabid-mediated pest suppression can be enhanced through local habitat manipulation at the field scale. Unmanaged or sown ecological field margins within cereal crops have been shown to increase numbers of predatory species (Woodcock *et al.* 2005; Woodcock *et al.* 2010; Mansion-Vaquié *et al.* 2017) while beetle banks are known to boost numbers of predators and contribute to pest suppression within crops (Collins *et al*., 2002; MacLeod *et al.*, 2004; Pywell *et al.*, 2015). The impact of higher numbers of predacious species on pest control is restricted by distance, with pest suppression decreasing with distance from non-crop habitats (Collins, 2002).

3.2.4. Functional diversity

Functional traits are ‘*morphological, biochemical, physiological, structural, phenological, or behavioural characteristics that are expressed in phenotypes of individual organisms’* (Violle *et al.* 2007; Díaz *et al.* 2013). The use of functional diversity as a means of classifying a community has been around for some years. Historically used to assess plant communities, the use of functional traits to classify faunal communities is increasingly common (Laureto *et al.,* 2015). Traits can be divided into ‘response’ and ‘effects’ traits. Response traits reflect an organisms adaptation to environmental stimuli, while effects traits are indicative of an organisms potential influence on, or ability to, affect ecosystem processes (Días *et al.* 2013). There will be some overlap between the classifications as all characteristics will have been selected to reflect available resources (de Bello *et al*., 2010).

Variations in community structure and function will reflect the adaptations necessary for success in a given habitat, region or under a particular management regime (Vandewalle *et al.*, 2010). The varying morphological and phenological attributes of an individual will determine its capacity to deliver pest control services. In turn, the relative proportions of different individuals within a community will define the community's functional identity (Moretti *et al*. 2009; Dias *et al*. 2013).

Functional redundancy describes the "*tendency for species to perform similar functions, such that they can compensate for changes in each other's contribution to ecosystem processes"* (Oliver *et al*., 2015). Where species in a community share effects traits, but differ in their response traits functional redundancy can arise (Oliver *et al.,* 2015), i.e. where there is a similar range of traits present in a community, despite the number or richness of species present (Straub *et al.*, 2008). Where redundancy is high, loss of individual species is likely to have a low, or no impact on the overall provision of an ecosystem service, conversely where redundancy is low, loss of a species could cause considerable disruption to ecosystem service provision (Flynn *et al.,* 2009). While high redundancy could lead to a more functionally stable ecosystem, it could have the opposite effect should there be a mis-match between dominant traits and ecosystem service provision. As Straub *et al.* (2008) discussed, boosting the diversity of natural enemy communities may not always result in increased pest suppression, as factors such as intraguild predation and functional redundancy may undermine top down control.

The associations between land use heterogeneity and natural pest control are likely to reflect changes in arthropod community structure and function, as well as simple abundance. Carabid activity-density and species richness appear less driven by land use heterogeneity, whereas variation in functional diversity at the community level related to land use heterogeneity has been more frequently observed (Woodcock *et al.,* 2014). The activity-density and diversity of carabid assemblages has been shown to respond in different ways to land use variables (Ernoult *et al.,* 2016*;* Gardiner *et al.,* 2010). Woodcock *et al.* (2014) showed carabid functional diversity increased with greater proportions of semi-natural habitats in the surrounding landscape, while species richness declined. Functional diversity can impart more information about a community's stability, resilience or ability to provide ecosystem services (Flynn *et al.*, 2009; Oliver *et al.,* 2015), and as such Kotze *et al.* (2011) suggest that functional diversity is a more useful approach to studying carabid communities, rather than richness or activity-density alone owing to these shortcomings.

In carabids, traits frequently studied include a range of morphological measurements relating to size and mobility, flight ability, breeding season and feeding preferences which are considered to be indicative of potential ecosystem service provision and tolerance of disturbance (Letardi *et al.*, 2015; Ribera *et al*., 1999a). Further, activity preferences may influence vulnerability to management. For example, diurnally-active carabids are more likely to come into direct contact with pesticides during spraying, while breeding season will determine activity levels and food requirements, and may also influence pesticide exposure (Hedde *et al.,* 2015).

It is thought likely that the ecosystem services provided by an orchard carabid assemblage will be affected by adjacent land uses and the broader diversity of land use at a landscape scale (Fusser *et al.*, 2018; Mansion-Vaquié *et al.,* 2017; Woodcock *et al.,* 2010). The intensive mowing and spraying regimes employed in orchards are also likely to alter the functional diversity of an assemblage (Epstein *et al.* 2000 and 2001; Hedde *et al*. 2015). The influences of these factors on carabid communities, and the knock-on effects for carabid mediated pest control in orchards is not yet fully understood.

3.3. Study aims

There are three main aims to this study: i) to assess orchard carabid community structure and function in relation to land use diversity at the landscape scale and in relation to adjacent land use, ii) to assess the influence of insecticide usage and mowing regime on orchard carabid communities and iii) to assess the potential contribution orchard carabid communities to pest regulation.

3.4. Methodology

3.4.1. Experimental design

The study was carried out within eight commercially managed farms in Kent, southern England situated in two different landscape types. Four were situated within the North Kent Plain and four in the Weald (Wealden Greensand and Low Weald). Sites were paired, with each pair consisting of an orchard in a highly diverse land use setting and one in a lower diversity setting. Study farms were selected based on specific criteria to allow an assessment of the influences of the surrounding land use at different spatial scales on orchard carabid communities.

At the landscape scale, study farms were classified in terms of the diversity of the surrounding land use within a 1 km radius. Broadly speaking, land use diversity was defined based on the number of different habitat types and the management intensity associated with each, found within the study area. For a full methodology on land use characterisation, along with details of site locations and study plot layouts, refer to Chapter 2.

When characterising land use in Chapter 2, the proportions of certain land use types were found to be correlated with overall land use diversity. For example, SDIh scores were lower where the proportion of managed habitats, (arable, improved grassland, orchard) was high in a landscape (see section 2.4). Consequently, representative habitats from each of the land use types found to correlate with land use diversity were chosen as a focus to investigate the influences of adjacent land use on orchard carabid communities. Within each farm three distinct plots, separated by a distance of at least 250m, were selected for further study. Each of these study plots was situated adjacent (≤ 25m) to a representative of one of the focal land-use types; 'agricultural' land (i.e. arable or improved grassland), additional apple orchard planting or an 'unmanaged', natural/semi-natural habitat (i.e. woody or scrub vegetation).

Within these plots two transects, running parallel to the boundary with the adjacent land use type, were established to sample carabids. The first transect was sited at the orchard edge, between the first and second row of orchard planting, adjacent to (≤ 25m) the land use of interest (henceforth 'edge' transects). The second transect is sited 50 m further into the orchard planting, as measured from the 'edge' transect (henceforth 'interior' transects). This allowed for qualification of differences between communities present at the ecotone between orchard and non-orchard habitats and those farther into the orchard. See Chapter 2 for more details.

3.4.2. Assessing carabid communities

Carabid communities were quantified using pitfall traps. Pitfall trapping is the most widely used collection method for carabids, though it provides a measure of activity-density rather than true abundance which can weaken estimates of community composition (Kotze *et al.*, 2011; Lövei and Sunderland, 1996). Nonetheless, the results from this approach are considered meaningful, particularly in predation studies, providing an estimation of species/trait diversity and activity relevant to predator-prey interactions (Woodcock, 2004). Suction sampling has also been shown to be an effective tool for sampling grassland invertebrates (Brook *et al.*, 2008), though the catch is somewhat restricted to smaller, diurnally active invertebrates, unless refugia such as vegetative tussocks are present. The method was not considered appropriate for this study owing to the focus on predacious carabids, the majority of which are large and nocturnal. These species generally retreat to micro-habitat features such as tussocks or refugia during daylight hours (Luff, 2007). The lack of structural diversity and refugia observed within the sward in study orchards was therefore anticipated to limit areas suitable for suction sampling of target carabids. Nonetheless, to confirm which technique would produce the most meaningful results for this study, suction sampling was trialled using a Vortis sampler. As anticipated, the methodology produced a community dominated by small herbivorous and detritivorous species, with few predacious species recorded, of which all were Collembola specialists. The use of pitfall sampling was therefore considered the most appropriate technique for this study.

Six pitfall traps were installed along each survey transect, set 20 m apart. Traps consisted of a removable plastic inner cup set within a harder outer cup (diameter of 80 mm, depth 105 mm). Traps were set flush with the ground and filled with a 70% dilution of propylene glycol as preservative and a small amount of detergent to break surface tension.

Gala apple does not typically begin flowering until April with fruit set ongoing until harvest in late August or early September. Sampling in this study was undertaken to reflect this seasonality. Although studies in arable systems have found that the bulk of carabid dispersal into the crop environment occurs early in the year from March through to May, when the crops have completed their main growth (Holland, 2002), sampling outside these dates was considered appropriate for this study to reflect the apple growing season.

Pitfall traps were opened for ten days at a time at approximately 20 day intervals during the apple growing season, between April to August 2013 and May to September 2014.

Trap catch was collected with samples drained and stored in 70% ethanol. Numbers caught were totalled across the six traps from each transect to represent a single sample. With the exception of the generally phytophagous genus *Amara,* all carabids were identified to species following the nomenclature of Luff (2007).

3.4.3. Measuring community diversity

This study aimed to explore general carabid community composition, rather than seasonal variations, and so for the purposes of analysis, the catches from each study transect were combined for each survey year. A number of diversity metrics were developed using the activity-density data to explore the community variance across treatments. Shannon-Weiner diversity and species richness were calculated for each transect using the 'vegan' package in R (Oksanen *et al.*, 2016; R Development Core Team, 2016).

This study focused on pest control in a farmed environment and so the traits employed reflect morphological adaptations to the ecosystem service provision of pest control (Ribera *et al.* 1999; Woodcock *et al.* 2010; Woodcock *et al.* 2014) and degree of adaptation to chemical and physical disturbance (Gobbi *et al.*, 2015; Letardi *et al.*, 2015). To calculate functional diversity, morphological and behavioural traits were recorded for each carabid species (Table 3.1; see Appendix 1 for full list of traits per species). The degree to which the adult carabid diet is specialised will restrict that species ability to affect pest control (Woodcock *et al.*, 2014) it will also be an indicator of the adaptability of that species to changes in food availability. Diurnal activity will influence what prey is likely to be encountered (Luff, 1978). As spraying is undertaken diurnally in orchards, diel activity will also influence a species' likely exposure to chemicals. Size is considered to be an indicator of the functional role of a species (Saint-germain *et al.*, 2007) and will reflect the ability of a carabid to consume prey of different sizes and types. Specific morphological traits such as wing morphology, metatibia length, eye size etc. are key indicators of visual and sensory acuity and a species ability to move in response to pest outbreaks thus describing the ability of a species' to locate, capture and consume prey (Ribera *et al.*, 1999; Woodcock *et al.*, 2010; Woodcock *et al.*, 2014). Size and wing morphology are also considered indicators of the degree to which a species is tolerant of disturbance (Gobbi and Fontaneto, 2008).

A measure of functional diversity was established for both response (RTFD) and effects (ETFD) traits; 'response traits', being those characteristics thought to reflect an organisms' response to environmental variables, and 'effects traits' comprising those characteristics affecting an organisms' ability to provide an ecosystem service (Díaz *et al.*, 2013).

Table 3.1: Morphological and behavioural traits associated with functional diversity after: [1] Ribera *et al.*, 1999, [2] Bauer, 1985, [3] Luff, 1978, [4] Wamser *et al.*, 2011, [5] Kotze and O'Hara, 2003, [6] Larochelle, 1990.

Effects Functional Diversity	Traits measured
Diet	*Food of adult* (Collembola specialist, generalist predator, mixed diet, plant based) [1, 6] *Diel activity* (diurnal, nocturnal) [2] *Size* (length) [1]
Responsiveness to pest outbreaks	*Wing morphology* (apterous/brachypterous, macropterous) [5]
Prey location, capture, handling	*Eye size* [2] *Antenna length* [2] *Femora length* [1] *Metatibia length* [1] *Metatarsi length* [1] *Metafemur width* [1]
Response Functional Diversity	
Adaptability to changing food resources	*Food of adult* [1, 6] *Size* [1]
Habitat flexibility	*Habitat associations*
Vulnerability to farm management	*Diel activity* [3] *Breeding season* (spring, autumn) [4] *Wing morphology* [5]

Functional diversity was established using the Rao quadratic entropy index of diversity (Rao Q) using the distance-based functional diversity indices function 'dbFD' in the FD package in R (Laliberté *et al*. 2014).

Rao Q uses information on functional dissimilarity between the species in a community and the relative abundances of species in that community to establish a diversity index (Rao, 1982). Rao Q was established using:

$$\mathrm{FD} = \sum_{i=1}^{S} \sum_{j=1}^{S} d_{ij} p_i p_j$$

where dij expresses the dissimilarity between each pair of species i and j according to their trait values (Rao, 1982; Ricotta, 2005). It was considered a suitable choice for measuring functional diversity as it allows consideration of abundances and multiple traits (Botta-Dukat, 2005).

The effect of a given species on ecosystem processes is proportional to its relative contribution to the total community (Dias *et al.,* 2013; Grime, 1998). As such, the mean values of a trait weighted by abundance of that trait in a community, i.e. the Community Weighted Mean (CWM), will reflect the properties of a community with regards to a given ecosystem function or service (Dias *et al.,* 2013). For this reason, appropriately chosen CWMs are a useful tool in identifying the potential of a community to tolerate disturbance, for example, or deliver pest suppression services.

Community weighted means were calculated, using the 'dbFD' package in R, for individual traits of particular relevance to this study, namely; predation potential, land use associations and vulnerability to pest management. Predation potential is represented by three CWMs; one based on size and two based on dietary response variables; i.e. a CWM based on the relative abundance of strictly predacious carabids, and a second CWM based on the relative abundance of polyphagous carabids. Land use associations are represented by a CWM based on known association with agricultural land. This also provides an indication of disturbance tolerance, as agricultural communities are thought to be more resilient to disturbance (Letardi *et al.* 2015). CWMs for daily activity and breeding seasons were calculated as proxies for vulnerability to farm management.

3.4.4. Vegetation composition

Vegetation composition was recorded from a 1 m^2 quadrat placed around each pitfall. The percentage cover of graminoids, forbs and bare ground was estimated using vertical projection. The mean vegetation height was also estimated from each quadrat based on the maximum vegetation height (cm) recorded from four randomly placed measuring sticks. Quadrat data was collected at the beginning of each 10-day pitfall survey round.

3.4.5. Assessing predation by carabids

Live sentinel aphids *Acyrthosiphon pisum* (Hemiptera Aphididae) were used to measure variations in predation according to land use diversity and carabid communities.

Aphids were maintained in the laboratory on a diet of young pea plants. Colonies were kept at approximately 16^{o}C and subject to a 12hr +/- light dark cycle. Three bait cards were deployed along each of the study transects for each survey round. The survey was repeated once a month from April through to August of 2015 on dry nights only.

Figure 3.1. *An aphid bait card: practicing the gluing technique in the laboratory. Cards were pinned to the ground overnight using gardening wire.*

Each card consisted of a plastic plant tab, (1 cm x 5 cm) roughened with sandpaper to ensure grip. Ten live pea aphids were glued to each bait card using PVA glue (Fig. 3.1). Other similar studies using sentinel prey have employed adhesives of the Cyanoacrylates family (i.e. Superglue™) to affix prey (Geiger *et al.*, 2010; Ximenez-Embun *et al.*, 2014). However, these adhesives are expensive, difficult to work with and cause aphids to desiccate quickly. Laboratory trials were undertaken as part of this study into alternatives and PVA glue was shown to be adequately adhesive, holding aphids in place for 24 hours or more, as well as sufficiently waterproof, withstanding simulated dew and light rain. In Chapter 4, feeding trials using adult earwigs found that aphids attached with PVA were consumed without prejudice. As such, PVA was considered unlikely to deter carabids and was utilised in this study. Glue was applied to the tabs and allowed to dry slightly. Live aphids, of 3rd instar or older, were gently affixed by the hind legs/posterior using a paintbrush.

As the majority of carabids are nocturnally active, bait cards were deployed at dusk and collected 12 hours later. Three cards were set on the ground at 30 m intervals along the existing survey transects. Using garden wire, cards were laid lengthways perpendicular to the soil surface to prevent excessive moisture build-up. The proportion of aphids consumed was recorded when cards were collected.

3.4.6. Measuring Insecticide Usage in Orchards

The Treatment Frequency Index (TFI) is a commonly used metric to characterise pesticide usage based on the mean number of treatments weighted by the ratio of the dose applied to the recommended dose (after Jørgensen, 1999 in Malagnoux *et al.* 2015). It is calculated using the following formula:

$$\mathrm{TFI} = \sum_{i=1}^{n} \frac{\mathrm{AD}_i}{\mathrm{HD}}$$

where n is the total number of insecticides applied in one year in an orchard, ADi is the amount of each insecticide applied and HD is the recommended amount per hectare.

The TFI was calculated using records of each pesticide applied in each study plot in each survey year. Pesticide data for this calculation was extracted from growers' spray records, though only insecticides are considered in this study. No *a priori* estimations of toxicity were employed, rather calculations were made per plot based on the TFI values of annual insecticide pressure.

3.4.7. Analysis

To explore the influences of land use diversity on carabid communities, a series of linear mixed effects models were run in R (version 3.3, R Development Core Team, 2016) using the 'lme4' package (Bates *et al.*, 2016). The response variables tested were: activity-density, species richness, Shannon-Weiner diversity, effects traits and response traits functional diversity, and the community weighted means for diet (strictly predacious and polyphagous), diel activity, breeding season, size and habitat associations. The explanatory variables tested were: land use diversity at the landscape scale (high or low), plot (adjacent land use: agricultural, unmanaged or orchard), transect location (edge or interior of plot), survey year, % cover of forbs, average vegetation height and annual insecticide pressure (TFI). Interactions between variables were included initially and then excluded if found not to be significant. Interactions tested were; land use diversity*plot, land use diversity*transect location, plot*transect location and land use diversity*year, transect location*year, plot*year. Three-way interactions were tested but found to be non-significant so are not considered further. *Post hoc* testing was undertaken using Tukey's HSD tests.

The effects of applications of individual insecticides on carabid communities was also tested using a series of linear mixed effects models using the 'lme4' package. Insecticides which were only applied once were excluded from analysis. Likewise, where insecticide applications were highly intercorrelated (i.e. commonly applied simultaneously), one was excluded to prevent model over-parameterisation. The response variables tested were those detailed above. The explanatory variables tested were usage of methoxyfenozide, thiacloprid, flonicamid and chlorpyrifos and survey year.

To account for the hierarchical structure of the experimental design, land use, plot and transect location were included as random effects. Total counts and species richness were log transformed prior to analysis.

Predation was assessed as a binomial response using glmer models from the 'lme4' package in R. Land use data is fixed across all study years; however, community data in this experiment is based on the 2014 carabid assemblage data as it represents the greatest contributor to the overall dataset.

The variation in aphid predation was tested against the following explanatory variables: land use diversity at the landscape scale (high or low), plot (adjacent land use: agricultural, unmanaged or orchard), transect location (edge or interior of plot), and a number of diversity metrics, namely the CWMs for strictly predacious carabids, polyphagous carabids and size, as well as species richness and Shannon Diversity. The remaining diversity metrics (i.e. response and effect functional diversity, diurnal, seasonal and habitat CWMs) were excluded to prevent overparameterization of the model and avoid intercorrelation with other metrics.

3.5. Results

During sampling 12,938 individual adult carabids were caught, representing 36 different species. Catch varied significantly between the two sampling years with 4938 specimens recorded overall in 2013 (mean 102.88, s.e.m 11.98), compared with 8000 in 2014 (mean 166.67, s.e.m 16.48).

The assemblage was dominated by a small number of species, with just two accounting for 71% of the total catch; *Harpalus rufipes* (De Geer) at 47%, and *Nebria brevicollis* (Fabricius) at 24%. Of the remaining species *Calathus fuscipes* (Goeze) represented 8%, *Harpalus affinis* (Schrank) 7%, *Pterostichus madidus* (Fabricius) 6% and *Pterostichus melanarius* (Illiger) 3%. The remaining 30 species represented 1% or less of the total catch. See Table 3.2 for a full species list.

Vegetation characteristics varied across farms, however there was no significant difference in average height, the percentage cover of graminoids or of forbs according to land use diversity or adjacent land use. Across all transects the mean cover of forbs was found to be as 36%, the mean cover of graminoids was 50% and the mean height was 8.95 cm.

3.5.1. The effects of adjacent land use and land use diversity at a landscape scale on orchard carabid communities

3.5.1.1. Activity-density and diversity

Analysis confirmed survey 'year' was the only factor to influence carabid activity-density; higher counts were recorded in 2014 compared to 2013 (Wald χ^2 = 15.919, d.f = 1, p <0.001). There were no significant differences in carabid activity-density according to the diversity of land use at the landscape scale or in relation to adjacent land use, or of any interactions between these factors. Neither vegetation height nor the cover of forbs were related to activity density.

Shannon diversity was significantly higher in 2013 (Wald χ^2 = 7.969, d.f = 1, p = 0.005), as was ETFD (Wald χ^2 = 22.019, d.f = 1, p <0.001) and to a lesser degree, RTFD (Wald χ^2 = 6.293, d.f = 1, p = 0.01). Species richness was not significantly affected by any of the measured variables.

3.5.1.2. Community weighted means

The effects of land use diversity on the community weighted means (i.e. the proportional representation of a given trait in an assemblage) and other community diversity metrics were analysed. Table 3.2 provides the total counts of individuals recorded for each community weighed mean trait. Table 3.3 lists the species recorded and their CWM traits.

Table 3.2: *The numbers of carabids recorded per Community Weighted Mean trait.*

Trait	No. of Individuals	Trait	No. of Individuals
Nocturnal	11946	Diurnal	992
Autumn breeding	11457	Spring breeding	1481
Insectivorous	5096	Phytophagous	7482
Polyphagous	11893	Restricted diet	1045
Associated with agricultural habitats	11866	Not associated with agricultural habitats	1072
Size class			
≤ 5 mm	206	*9 – 15 mm*	11525
5 – 9 mm	1112	*4 ≥ 15 mm*	95

Table 3.3: *Carabid species list, total counts and CWM traits categories.* (Sources: [1] Ribera *et al.*, 1999, [2] Bauer, 1985, [3] Luff, 1978, [4] Wamser *et al.*, 2011, [5] Kotze and O'Hara, 2003, [6] Larochelle, 1990).

Species	**Total Count**	CWM Diurnal Activity[3]	CWM Seasonal Activity[4]	CWM Size[1, 2]	CWM Predacious[1, 6]	CWM Polyphagous[1, 6]	CWM Habitat[1]
Abax parallelepipedus	**22**	*n*	*a*	*l*	*i*	*m*	*na*
Acupalpus meridianus	**5**	*n*	*s*	*vs*	*i*	*m*	*a*
Amara sp.	**195**	*d*	*s*	*s*	*p*	*s*	*na*
Anchomenus dorsalis	**2**	*n*	*s*	*s*	*i*	*m*	*a*
Anisodactylus binotatus	**1**	*n*	*s*	*m*	*p*	*s*	*a*
Asaphidion flavipes	**7**	*d*	*s*	*vs*	*i*	*m*	*na*
Bembidion gilvipes	**1**	*n*	*s*	*vs*	*i*	*m*	*na*
Bembidion lampros	**83**	*d*	*s*	*vs*	*i*	*m*	*a*
Bembidion lunulatum	**1**	*d*	*s*	*vs*	*i*	*m*	*na*
Bembidion obtusum	**1**	*d*	*a*	*vs*	*i*	*m*	*a*
Bembidion properans	**13**	*d*	*s*	*vs*	*i*	*m*	*na*
Calathus ambiguus	**2**	*n*	*a*	*m*	*i*	*m*	*na*
Calathus erratus	**1**	*n*	*a*	*m*	*i*	*m*	*na*
Calathus fuscipes	**1069**	*n*	*a*	*m*	*i*	*m*	*a*
Calathus melanocephalus	**2**	*n*	*a*	*s*	*i*	*m*	*a*
Carabus violaceus	**72**	*n*	*a*	*l*	*i*	*m*	*na*
Clivina fossor	**6**	*d*	*s*	*s*	*i*	*m*	*a*
Curtonotus aulicus	**1**	*n*	*a*	*m*	*p*	*s*	*na*
Harpalus affinis	**843**	*n*	*s*	*s*	*p*	*s*	*a*
Harpalus latus	**4**	*n*	*s*	*s*	*p*	*s*	*na*
Harpalus rufipes	**6055**	*n*	*a*	*m*	*p*	*m*	*a*
Laemostenus terricola	**1**	*n*	*a*	*l*	*i*	*m*	*na*
Leistus spinibarbus	**1**	*n*	*a*	*m*	*i*	*m*	*na*
Loricera pilicornis	**56**	*d*	*s*	*s*	*i*	*m*	*a*
Nebria brevicollis	**3107**	*n*	*a*	*m*	*i*	*m*	*a*
Notiophilus biguttatus	**88**	*d*	*s*	*vs*	*i*	*m*	*a*
Ophonus ardosiacus	**1**	*n*	*a*	*s*	*p*	*s*	*na*
Poecilus cupreus	**173**	*d*	*s*	*m*	*i*	*m*	*a*
Pterostichus cristatus	**4**	*n*	*a*	*m*	*i*	*m*	*na*
Pterostichus madidus	**742**	*n*	*a*	*m*	*p*	*m*	*na*
Pterostichus melanarius	**369**	*d*	*a*	*m*	*i*	*m*	*a*
Pterostichus vernalis	**2**	*n*	*s*	*s*	*i*	*m*	*na*
Stomis pumicatus	**1**	*n*	*s*	*s*	*i*	*m*	*na*
Trechus quadristriatus	**2**	*n*	*a*	*vs*	*i*	*m*	*a*
Trechus rivularis	**1**	*n*	*a*	*vs*	*i*	*m*	*na*
Trechus secalis	**4**	*n*	*a*	*vs*	*i*	*m*	*a*

CWM Diurnal Activity: d = diurnal, n = nocturnal,
CWM Seasonal Activity: s = spring breeding, a = autumn breeding,
CWM Size: vs: ≤ 5 mm, s: 5 – 9 mm, m: 9 – 15 mm, l ≥ 15 mm,
CWM Predacious: i = insectivorous, p = phytophagous,
CWM Polyphagous: m = mixed diet, s = strict insectivore or phytophage,
CWM Habitat: a = associated with agricultural habitats, na = not associated with agricultural habitats

3.5.1.3. Diet

'Predacious CWM', i.e. the proportion of strictly predacious species in a given assemblage, was unaffected by the diversity of land use at landscape or local scales, nor by survey year.

'Polyphagous CWM', i.e. the proportion of species in the assemblage with a mixed diet, was significantly affected by land use diversity at the landscape scale, with a greater proportion of polyphagous individuals recorded in more heterogenous land use settings at the landscape scale (Wald χ^2 4.620, d.f = 1, p = 0.031; Fig. 3.2). The most parsimonious model included an interaction effect for landscape*plot (Wald χ^2 = 5.782, d.f = 2, p = 0.055) although *post hoc* analysis using a Tukey's HSD test found no significant variation in polyphagous CWM according to landscape*plot.

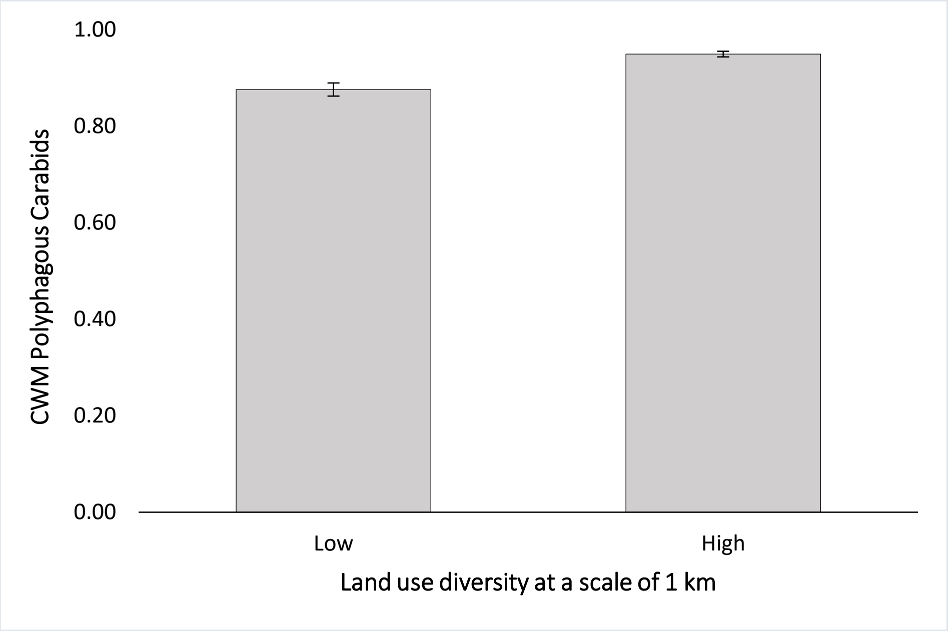


Figure 3.2. *A bar plot showing the influence of the diversity of land use at the 1 km scale on the CWM for polyphagous carabids. The proportion of species in the assemblage with a mixed diet was higher in orchards in more heterogenous land use settings (p = 0.031). Error bars denote the standard error of the mean.*

In addition, polyphagous CWM was significantly affected by 'year' (Wald χ^2 = 6.591, d.f = 1, p = 0.010), with a higher proportion of polyphagous species being recorded in the assemblage in 2014 (mean 0.922, s.e.m 0.01) compared to that recorded in 2013 (mean 0.903, s.e.m 0.01; Fig. 3.3).

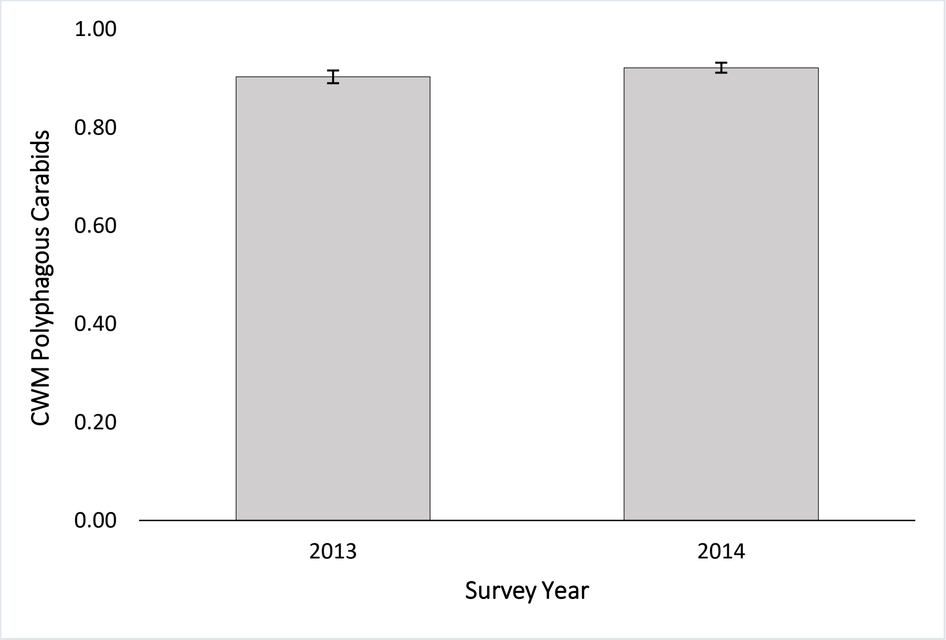
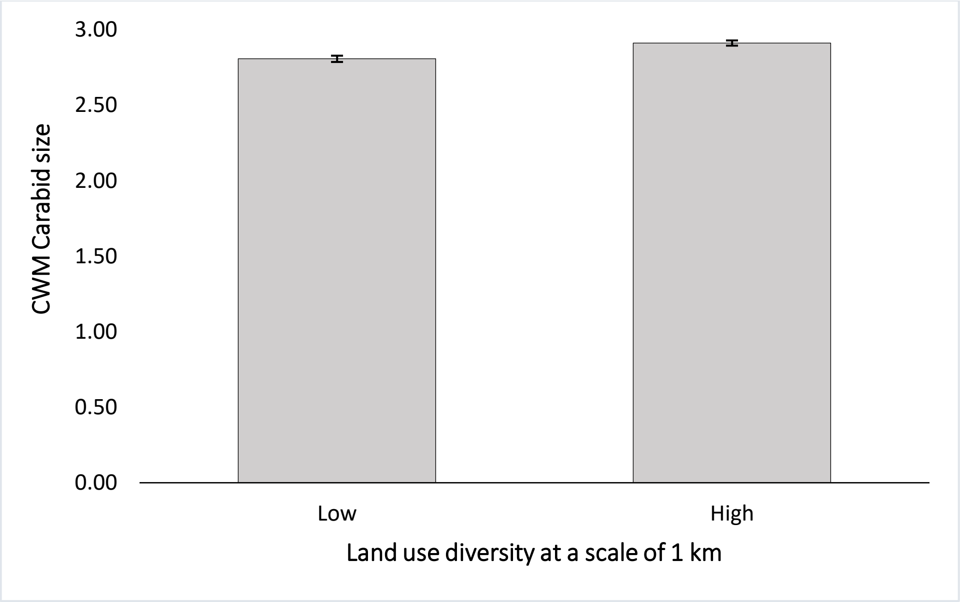


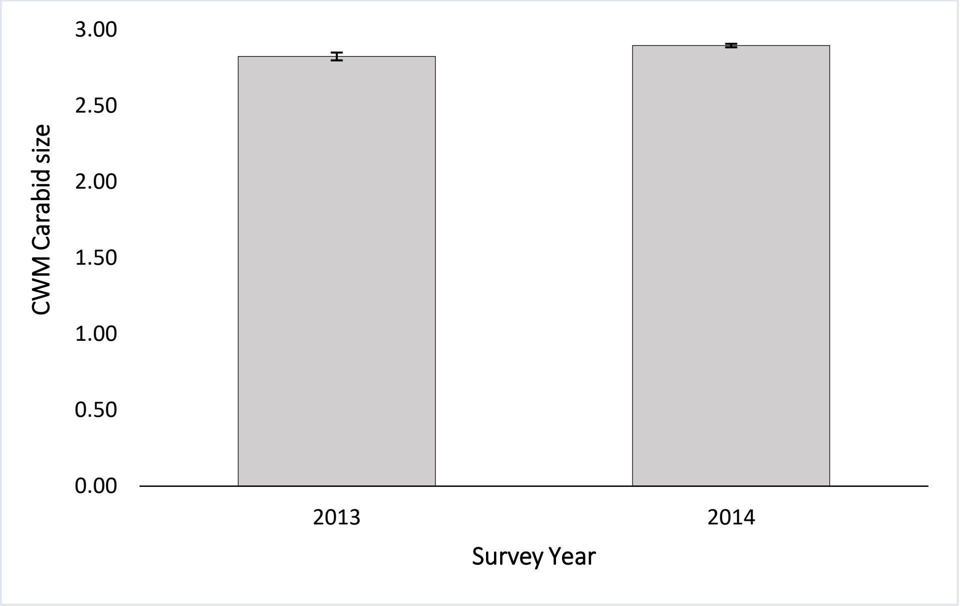
Figure 3.3. *A bar plot showing the influence of survey year on the CWM for polyphagous carabids. The proportion of species in the assemblage with a mixed diet was greater in 2014 (p = 0.010). Error bars denote the standard error of the mean.*

The percentage cover of forbs in vegetation quadrats remained in the most parsimonious models for polyphagous CWM (Wald χ^2 = 3.748, d.f = 7, p = 0.053), however *post hoc* analysis using a Tukey's HSD test found forb cover was not a significant influence.

3.5.1.4. Size

Size CWM was found to vary significantly according to land use diversity at a 1 km scale, with the assemblage containing a greater proportion of species with large body size in more heterogenous landscapes (Wald χ^2 = 6.670, d.f = 1, p = 0.009), Fig. 3.4. Year was also identified as being a significant influence on size CWM, with a greater proportion of large species being recorded in 2014 (Wald χ^2 = 10.279, d.f = 1, p = 0.001), Fig. 3.5.





Figures 3.4 & 3.5: *Bar plots showing the variation in the CWM for size according to a) the diversity of land use at the 1 km scale, where a greater proportion of large species was recorded in more heterogenous landscapes (p = 0.009) and b) survey year, where a greater proportion of large species was recorded in the assemblage in 2014 (p = 0.001). Error bars denote the standard error of the mean.*

3.5.1.5. Daily and seasonal activity

Diel CWM, i.e. the proportion of diurnally active carabids in the assemblage, was shown to be significantly affected by an interaction between year and the adjacent land use type (Wald χ^2 = 11.726, d.f = 2, p = 0.003). *Post hoc* analysis using a Tukey's HSD test found a higher proportion of diurnally active species in the assemblage in plots adjacent to agricultural land in 2013 (mean 0.161, s.e.m 0.032) compared to 2014 (mean 0.082, s.e.m 0.016), Fig. 3.6.

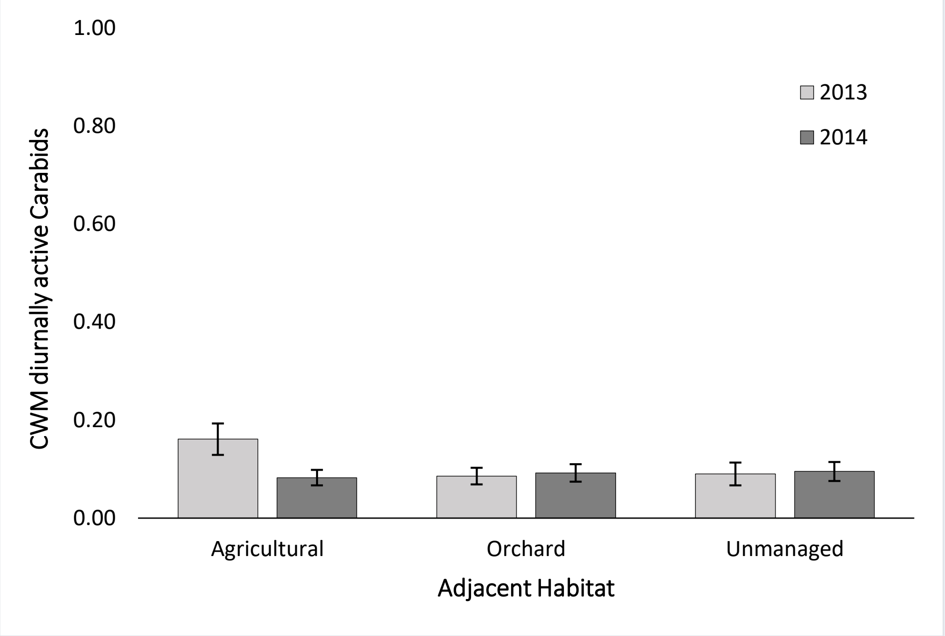


Figure 3.6: *A bar plot showing the influence of the interaction between survey year and adjacent land use on the CWM for diurnal activity. A higher proportion of diurnally active species were recorded in the assemblage in plots next to agricultural land in 2013 (p = 0.003). Error bars denote the standard error of the mean.*

Breeding CWM, i.e. the proportion of spring breeding species in the assemblage, was found to be influenced by land use diversity at the landscape scale with a higher proportion of spring breeding recorded in the assemblage in orchards in more homogenous landscapes (Wald $\chi^2 = 5.794$, d.f = 1, p = 0.016), Fig. 3.7.

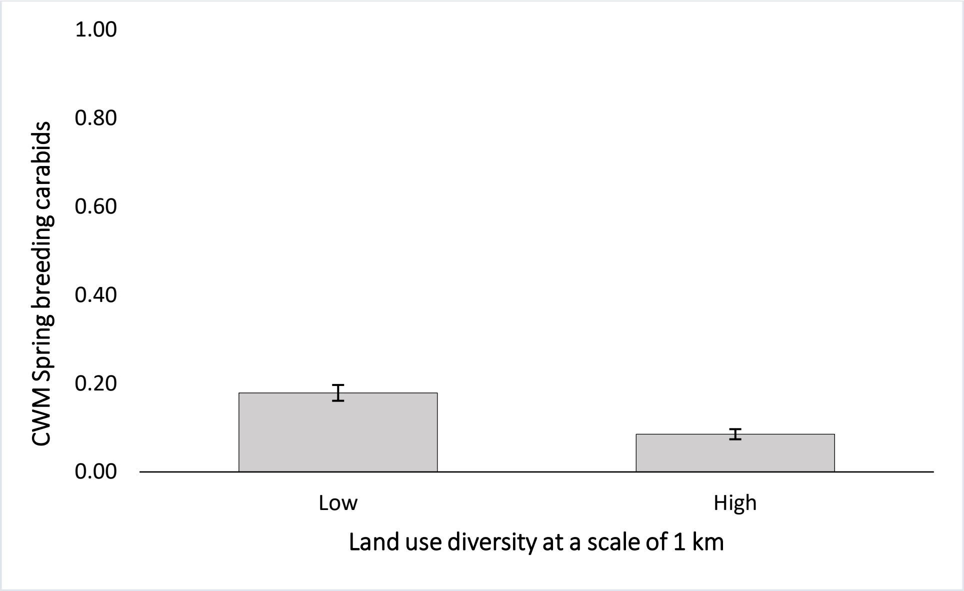


Figure 3.7: *A bar plot showing the influence of survey year on the CWM for breeding season. Carabid assemblages in orchards in more homogenous land use settings had a higher proportion of spring breeding species (p = 0.016). Error bars denote the standard error of the mean.*

3.5.1.6. Habitat

Habitat CWM, i.e. the proportion of species associated with agricultural habitats, was shown to be significantly influenced by an interaction between adjacent land use type and the distance from that land use (Wald $\chi^2 = 12.862$, d.f = 2, p = 0.002). *Post hoc* analysis using a Tukey's HSD test found a higher proportion of species associated with agricultural land in central transects of plots next to unmanaged land, compared to those in the edge transects, Fig. 3.8.

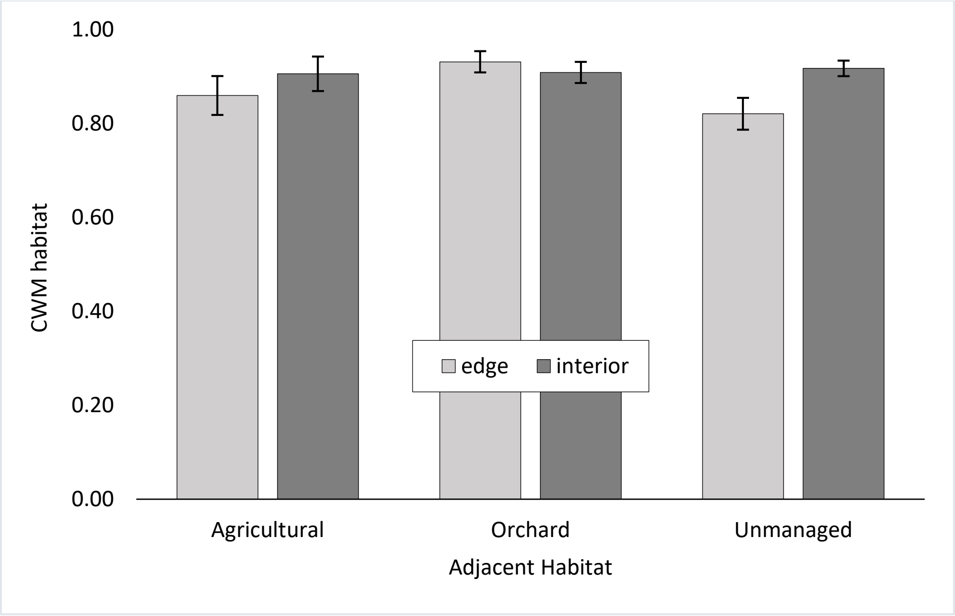


Figure 3.8: *A bar plot showing the influence of the interaction between adjacent land use and transect location on the CWM for habitat. The proportion of species associated with agricultural habitats in the assemblage was higher in the central transects of plots next to unmanaged land compared to those in the edge transects in those plots (Wald χ2 = 12.862, p = 0.002). Error bars denote the standard error of the mean.*

3.5.2. The effects of carabid communities, adjacent land use and land use diversity at landscape scale on pest control in orchards

Predation (i.e. the number of aphids consumed) was found to be significantly influenced by the proportion of polyphagous species in the assemblage (polyphagous CWM) (Wald χ^2 = 6.748, d.f = 1, p = 0.009), with greater predation recorded where the proportion of polyphagous species was higher, Fig. 3.9.

No other community metrics, nor the diversity of land use at any scale, were found to influence observed aphid predation rates.

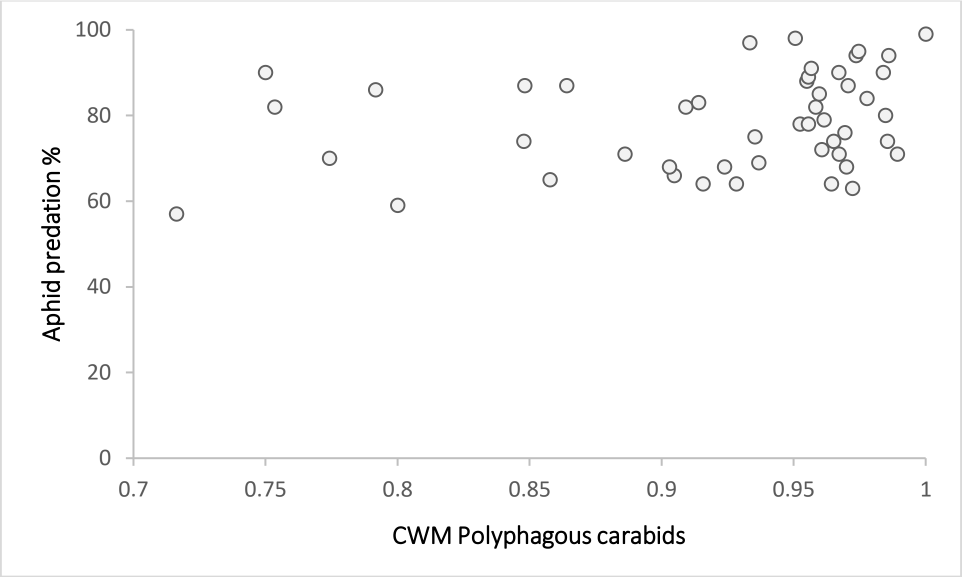


Figure 3.9: *Aphid predation in relation to the proportion of polyphagous individuals in the assemblage. The percentage of aphids consumed increased where the proportion of polyphagous individuals in the assemblage was higher ($\chi 2 = 6.748$, $p = 0.009$).*

3.5.3. The effects of insecticide usage on the carabid community

The TFI, i.e. the total number of insecticide treatments per study plot per year, was analysed in relation to orchard carabid assemblages. The cumulative TFI for insecticide usage during the study was found to be associated with an increase in carabid activity-density (t = 3.84, df = 94, p <0.001). As a result of this finding, the TFI values for individual chemicals applied were analysed to determine which insecticides were driving this trend and understand the effects of this on the community. Appendix 2 provides the overall TFI values, as well as TFI values for each individual chemical applied between 2013 and 2015.

3.5.3.1. The effects of insecticides on activity-density

Chlorpyrifos featured in the majority of significant models, being associated with greater numbers of carabids but lower diversity overall. Methoxyfenozide did not feature in any of the models.

Activity-density was found to be greater with increased use of chlorpyrifos (Wald $\chi^2 = 11.77$, d.f = 1, p <0.001). The most parsimonious model included flonicamid and year, though chlorpyrifos was the only insecticide found to significantly influence overall activity-density,

Fig. 3.10. It should be noted that application data for flonicamid and thiacloprid are somewhat intercorrelated. Activity-density was higher in 2014 than in 2013.

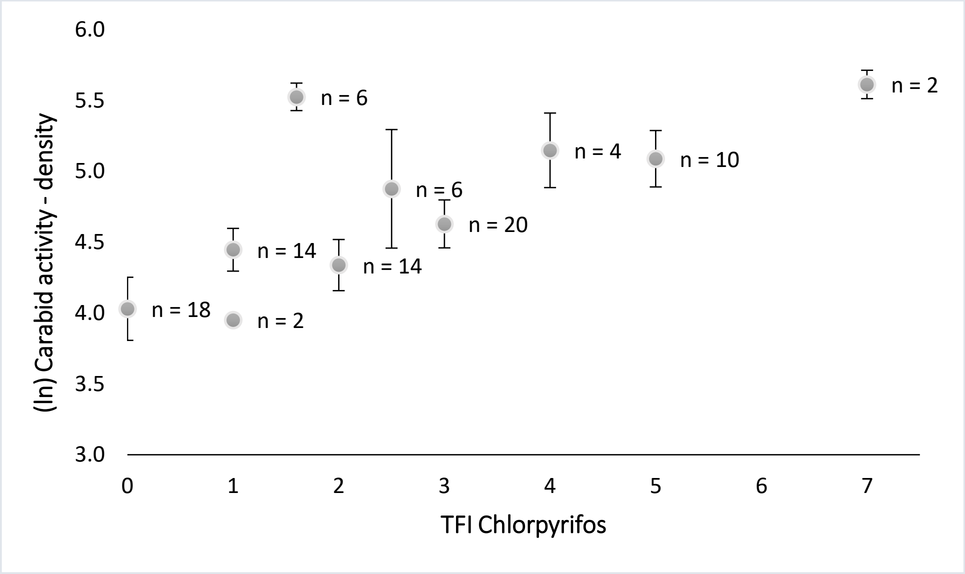
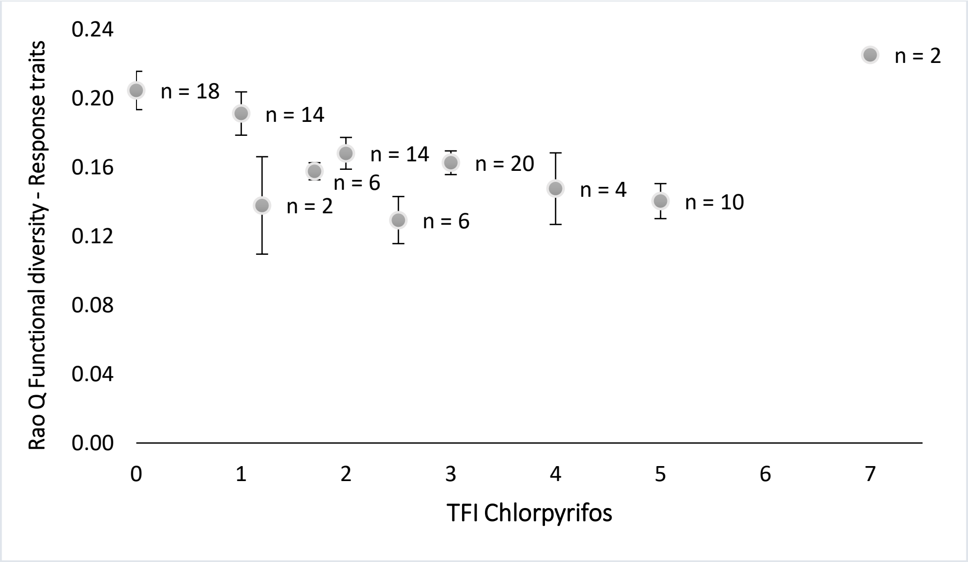


Figure 3.10: *A scatter plot showing the response of carabid activity-density to chlorpyrifos use. Activity-density increased where the TFI for chlorpyrifos was higher (Wald χ2 = 11.77, p <0.001). Error bars denote the standard error of the mean.*

3.5.3.2. The effects of insecticides on diversity and richness

In terms of functional diversity (Rao Q), both RTFD and ETFD were found to significantly decrease with greater chlorpyrifos use (Wald χ^2 = 5.97, d.f = 1, p = 0.014 and Wald χ^2 = 10.58, d.f = 1, p <0.001, respectively; Figs. 3.11 and 3.12). ETFD was also found to decrease in the 2014 survey year (Wald χ^2 = 23.31, d.f = 1, p <0.001), as overall numbers increased. Thiacloprid was associated with a significant increase in RTFD (Wald χ^2 = 7.905, d.f = 1, p = 0.005; Fig 3.13).

No effects of insecticide usage on Shannon diversity were observed, though Shannon diversity was significantly different between survey years, (Wald χ^2 = 7.97, d.f = 1, p = 0.005), with higher diversity recorded in 2013. Species richness was unaffected by insecticide usage or survey year.



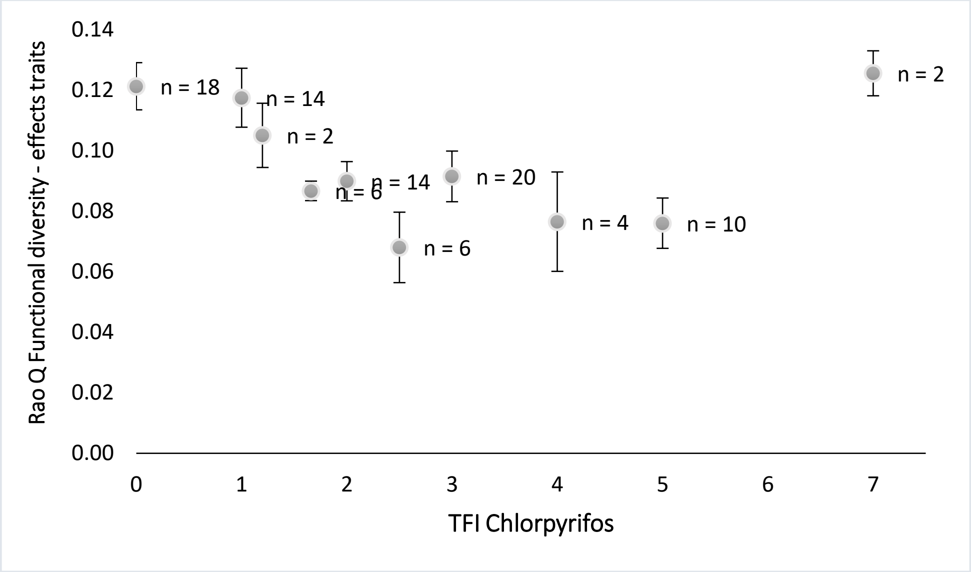


Figure 3.11 & 3.12: *Scatter plots showing the relationship between chlorpyrifos use and Rao Q Functional diversity. Both RTFD and ETFD were found to decrease where the TFI for chlorpyrifos was higher (Wald χ2 = 5.97, p = 0.014, Wald χ2 = 10.58, p <0.001 respectively). Error bars denote the standard error of the mean.*

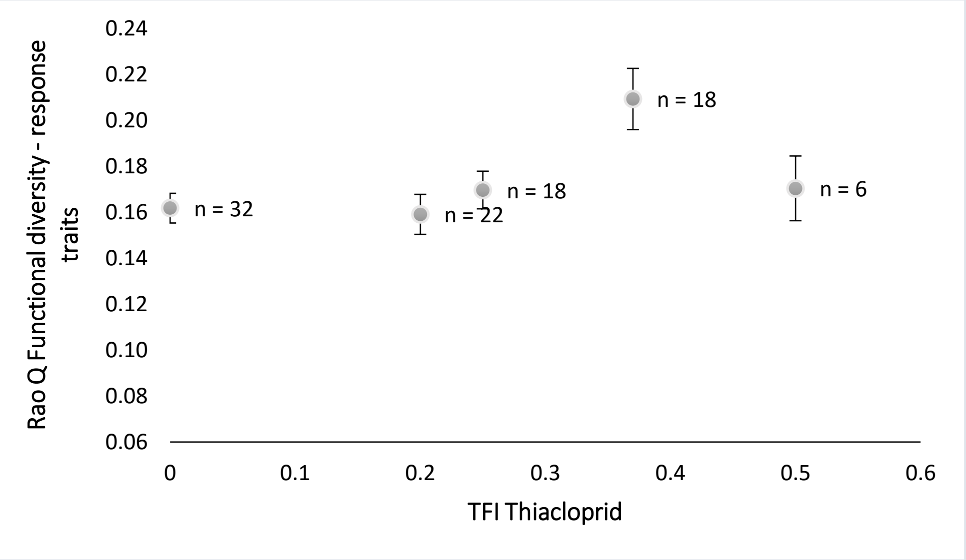


Figure 3.13: *A scatter plot showing the relationship between thiacloprid use and response traits functional diversity. RTFD was found to increase where the TFI for thiacloprid was higher (Wald χ2 = 7.905, p = 0.00). Error bars denote the standard error of the mean.*

3.5.3.3. The effects of insecticides on community weighted means

Size CWM was found to be significantly affected by thiacloprid usage (Wald χ^2 = 11.07, d.f = 1, p <0.001) with assemblages comprising a greater proportion of small species where thiacloprid usage was higher, see Fig. 3.14.

The use of chlorpyrifos was found to significantly influence polyphagous CWM with a greater proportion of polyphagous species recorded in the assemblage where chlorpyrifos usage was higher (Wald χ^2 = 4.48, d.f = 1, p 0.03), Fig. 3.15. No other CWMs were influenced by insecticide usage.

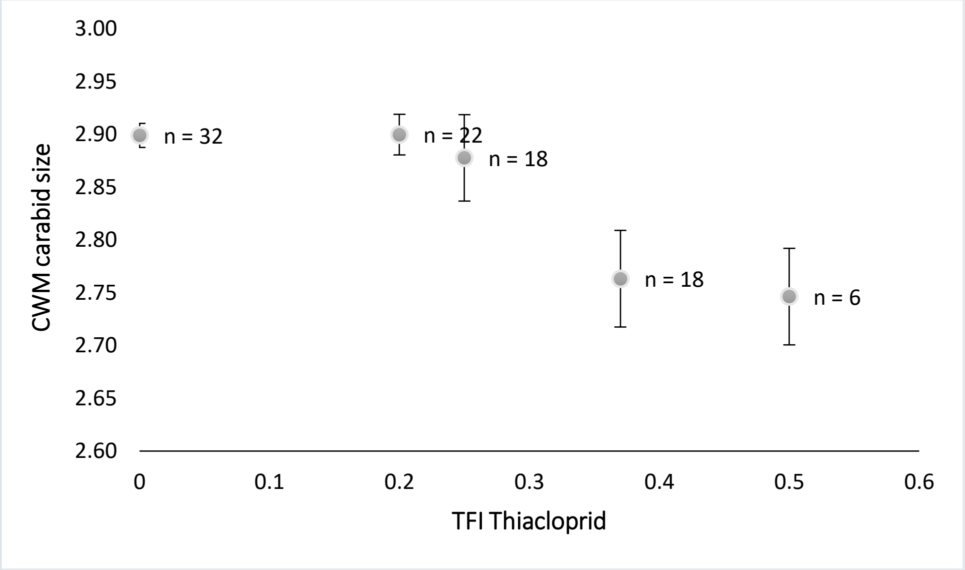


Figure 3.14: *A scatter plot showing the relationship between thiacloprid use and the CWM for carabid size. The proportion of smaller carabids in the assemblage increased with greater use of thiacloprid (Wald χ2 = 11.07, p <0.001). Error bars denote the standard error of the mean.*

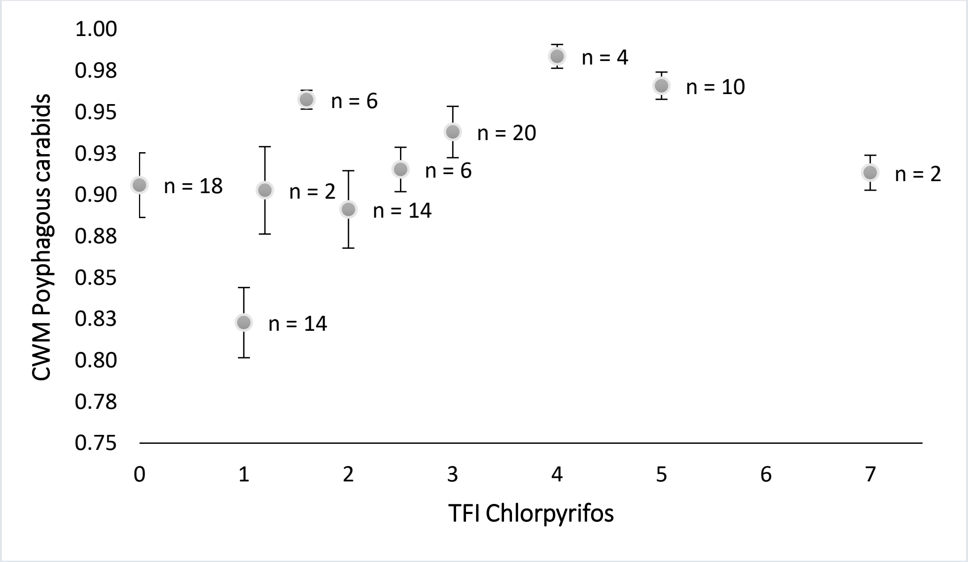


Figure 3.15: *A scatter plot showing the relationship between chlorpyrifos use and the CWM of polyphagous carabids. The proportion of polyphagous species was found to increase where the TFI for chlorpyrifos was higher (Wald χ2 = 4.48, p = 0.03). Error bars denote the standard error of the mean.*

3.6. Discussion

3.6.1. The effects of land use at landscape and local scales on orchard carabid communities

Neither carabid activity-density, nor diversity (Shannon-Weiner & Rao Q functional) were influenced by the diversity of land use at a 1 km scale, or adjacent land use type. However, all of the individual community weighted mean traits, with the exception of strictly predacious diet, were influenced by land use diversity.

A comprehensive study demonstrated that functional diversity metrics were better predictors of ecosystem function than abundance, and in turn that community weighted means based on individual traits were better predictors again (Gagic *et al.* 2015). Given that standard measures of the community were not found to respond to land use diversity, the findings presented here may indicate that the additional information associated with CWMs provides a more meaningful measure of the impact of environmental stimuli on carabid communities.

Carabid diet can be considered both a response, and an effect trait, as it will reflect a species' evolutionary adaptation to food availability as well as its potential ecosystem service/disservice provision. The orchard carabid assemblage was found to support a greater proportion of polyphagous carabids where the diversity of land use at 1 km was greater. This finding suggests that, contrary to other studies (Cizek *et al.*, 2012; Gobbi *et al.*, 2015), the level of diet specialisation in the carabid assemblage decreases in more diverse, less intensively managed, landscapes. Less managed habitats should be more stable and therefore can be expected to support higher levels of specialisation, and both specialist and generalist predatory species have been shown to increase with greater land use heterogeneity (Chaplin-Kramer *et al.*, 2011).

This study was undertaken in Kent where the agricultural landscape is dominated by orchards. To a degree, orchards could be considered more akin to semi-natural habitats than other forms of agricultural crop. Despite the high levels of management required, the crop itself is relatively permanent with traditional plantings remaining in-situ for up to 30 years (Cross *et al.,* 2015). Even with newer cropping systems, apple trees planted today are expected to produce for 15-20 years (pers. obs.). It may be that the orchard carabid assemblage is less responsive to surrounding land use than communities found in other crops as the species present have been selected to withstand a certain degree of management in a relatively stable habitat.

The results regarding habitat associations would seem to support this hypothesis as a greater proportion of species associated with agricultural land were recorded in the assemblage in interior transects, i.e. at a distance of ≥ 50 m, from the adjacent semi-natural/natural land use. The community variation between edge and interior transects may indicate the assemblage adjacent to non-crop habitat results from spill over and represents an ecotone community, while the more typical orchard assemblage, rich in agricultural species, can be seen farther into the orchard plantations. Carabid permeation into crops from non-crop habitats is known to be somewhat restricted by distance with a decline expected beyond 50 m (Woodcock *et al.*, 2016).

In contrast, there was no significant variation in agricultural associated CWM for plots next to additional orchard planting or agricultural land, suggesting these communities are already relatively stable, and experience little or no spill-over.

Diurnal and seasonal activity was also found to vary according to land use diversity. A higher proportion of spring breeding species were present in assemblages in landscapes with a lower land use diversity at a 1 km scale. This agrees with previous findings (Ribera *et al.* 2001) where more spring breeding species were associated with more disturbed, agricultural land. Breeding period will have implications for activity times, emergence and appearance in the crop and encounters with prey (Wamser *et al.,* 2011). Hedde *et al.* (2015) have suggested that early season breeders may be more at risk from pesticides in orchards owing to the overlap of spray intensity and increased activity by spring breeding carabids. However, the greater prevalence of spring breeding species in agriculturally dominated landscapes and habitats would suggest that the species present were selected based on a level of resistance to such management. It may be that those species with greater vulnerability to chemical application are no longer present in agricultural settings. Regardless, the presence of spring breeders in the study orchards implies the orchard carabid assemblage could provide a pest suppression service as main activity will coincide with the apple growing season, thus carabids could be expected to respond during pest outbreaks. Taking the woolly apple aphid as an example; Lordan *et al* (2014) demonstrated that woolly apple aphid will recolonize from the root from spring throughout the apple growing season. This tendency could expose colonies to predation by opportunistic epigeal generalists already present in the orchard, such as the polyphagous carabids found in this study.

An interactive effect between plots adjacent to agricultural land use and year was found for the CWM for diurnal activity. This finding suggests a slight shift in community composition from more diurnally active species in year one, when overall activity-density was lower, to more nocturnally active species in year two when activity-density increased. This trend was only observed in plots adjacent to agricultural land. It is not clear what may have been driving this shift.

The proportion of larger beetles in the assemblage was also found to vary according to land use diversity at a 1 km scale, with a smaller proportion of large species in the assemblage recorded in plots in more homogenous landscapes. This agrees with the prevailing research, that larger species are more prone to extinctions when compared to smaller ones, with such declines being more common in more managed landscapes (Kotze and O'Hara, 2003; Ribera *et al.*, 2001).

The majority of traits investigated in this study are considered indicative of a carabid assemblage's ability to affect pest suppression in orchards. From the results, it would appear that orchards surrounded by a more diverse land use at a 1 km scale are more likely to support assemblages with a greater proportion of polyphagous species, and large species, both traits which would directly influence prey choice. However, the influence of immediately adjacent land use on orchard the carabid assemblage appears negligible. This may be reflective of the relative stability of the orchard ecosystem.

It should be noted that all of the orchards in this study were frequently mown to a low height, 9 cm on average. Chapter 5 of this work has shown that such intense sward management has detrimental effects on carabid activity-density. As the treatment was applied universally across all study farms it is impossible to quantify to what extent, if any, this management is having on the assemblage. However, the results in Chapter 5, and those of other researches (Cizek *et al*., 2012; Lafage and Pétillon, 2014) would suggest it is highly likely that such intense vegetation management has an impact on the carabid assemblage .

3.6.2. The effects of land use diversity and carabid communities on pest control

In contrast to other work in the field (Bianchi *et al*., 2006; Chaplin-Kramer *et al.*, 2011; Rusch *et al*., 2016), it would appear that pest control in this study was not influenced by the diversity of the surrounding land use at a 1 km scale. Further, there was no evidence that the land use adjacent to an orchard plantation had an influence on predation. The lack of variation in predation according to adjacent land use may support the hypothesis that orchards are relatively stable habitats when compared to other crops, experiencing comparatively little ground disturbance and maintaining refugia suitable for overwintering carabids. Consequently we do not see the same early season influx of predators from overwintering habitats observed in arable settings (Holland, 2002).

It should be noted, that this study did not exclude other predators; it is possible therefore that other nocturnally active predators found and consumed bait aphids. Nonetheless, a greater proportion of polyphagous species in the assemblage was found to be positively correlated with the predation of aphid bait. This finding suggests that those species which possess feeding plasticity will take live prey on the ground when available. Interestingly, the proportion of strictly predacious species in the assemblage was not related to predation. This may be a result of lower numbers of strictly predatory species overall, or reflective of diet specialism among the strictly predacious species that were recorded in these assemblages.

The findings presented here suggest that greater numbers of generalist carabids in an assemblage can contribute to the ecosystem service provision of pest control in orchards. Though orchard pests generally inhabit trees, a considerable number spend at least part of their life cycle on the ground, e.g. rhynchites weevil, apple leaf midge and apple sawfly (Boreau de Roincé *et al*. 2012, Cross *et al.* 2015), leaving them vulnerable to predation by opportunistic carabids. Though this study only used aphid bait, the flexible diets of a carabid assemblage dominated by polyphagous species could be beneficial in reducing pests which are not otherwise vulnerable to predation while in the ground stages of their lifecycles. Further, the relative size of many of the carabids observed together with their active hunting styles would make them more capable of handling a range of prey (Ball *et al.*, 2015) than, for example, some of the spiders which may be present in the sward.

Further, interspecific predation with canopy dwelling predators may cause non-epigeal pests to drop to the orchard floor increasing the likelihood of opportunistic predation by carabids on other non-epigeal pests (Losey and Denno, 1998). Though land use may not directly influence carabid activity-density in orchards, the relationship between polyphagous species and predation may indicate a broader influence of land use diversity on pest predation, as the proportion of polyphagous species in the assemblage was found to increase with increased land use diversity at a 1 km scale.

3.6.3. The effects of effects of insecticide usage on carabid communities

A small number of studies of the effects of insecticide use on orchard carabid communities have been identified. In one study, neural-active insecticides, similar in mode of action to the thiacloprid tested here, have been shown to have a highly negative effect on carabid activity-density in commercial apple orchards (Epstein *et al.* 2001). The use of such products affected some species more than others which was partially attributed to the breeding season of those species. It was suggested that summer breeders were more at risk from the effects of insecticides as their breeding period, a vulnerable life stage, coincided with the greatest amount of insecticide use in the orchard. These kinds of impacts would suggest pesticide usage may select for certain species/traits in a community, for example those which breed outside of the main spraying season in a given crop. More recently Hedde *et al.* (2015) found carabid-activity density to be largely unaffected by chemical pest management, suggesting reductions in the numbers of some species may be counterbalanced by increases of more resistant ones. Using a traits based approach however, Hedde *et al.* (2015) found species' morphology, habitat and feeding preferences were influenced by chemical use. In a recent arable study, higher rate applications of a broad spectrum insecticide (chlorpyrifos) were associated with a reduction in overall carabid activity-density (Hill *et al.* 2017).

Insecticide application was shown to have a marked effect on the orchard carabid assemblage in this study, affecting activity-density and function. Somewhat counterintuitively, the TFI score for overall insecticide use was positively correlated with carabid activity-density. However, sub-lethal doses of insecticides have been associated with higher levels of carabid activity-density (Lövei and Sunderland, 1996). This could be partially attributed to a short-term increase in locomotive activity following exposure to insecticides (Tooming *et al.*, 2014). Further, it is thought that hungrier carabids are more active, and food shortages as a result of insecticide application can drive a boost in activity (Chiverton, 1984). As such, the increased activity-density observed may be a result of such factors.

Functional diversity, of both response and effects traits, decreased with increased use of chlorpyrifos in this study. The lower functional diversity observed in the carabid community may be indicative of a degree of functional redundancy, whereby numerous species in the community share the same traits (Oliver *et al.*, 2015). In this scenario, increased disturbance selects against less tolerant species, such that more tolerant species will compensate for reductions in numbers. This may be the case here, indeed higher numbers and lower ETFD were recorded in 2014. However, the shift does not appear to be having a marked detrimental effect on potential carabid mediated pest control in orchards. As the assemblage does not appear to be influenced by land use, the communities present may be relatively stable and well adapted to the levels of disturbance found in the study orchards. The high proportion of polyphagous species present and the relationship with polyphagous species and predation would suggest the assemblage is capable of pest suppression.

While the findings with regards to chlorpyrifos are interesting, they are somewhat irrelevant as chlorpyrifos was banned for use in most crops (save for the treatment of brassica seedlings) in 2016.

Neonicotinoids have been widely shown to have detrimental impacts on a range of beneficial invertebrates (Brandt *et al.*, 2017; Pisa *et al.,* 2014; Woodcock *et al*., 2017) and consequently several have been banned for use in the EU (European Commission, 2018). Thiacloprid is a neonicotinoid compound which has not yet been banned for use in the EU, as its toxicity is still under investigation (European Commission, 2018). At the time of writing, the compound was approved until April 2018, its use has now been extended until April 2019. Negative effects of thiacloprid on orchard natural enemies have been demonstrated in earwigs (i.e. Logan *et al.* 2011 and Chapter 4 of this study) and spiders (Funayama, 2011). There has been little specific work looking at carabids to date. Funayama (2011) found indications of a negative relationship between pest control programs, including thiacloprid, and *Amara chalcites.* The effects on other carabids were less conclusive, though that study did not examine pest control specifically. Low doses of thiacloprid has also been found to reduce numbers of soil arthropods including Collembola (Renaud *et al.,* 2018), which may reduce food availability for natural enemies.

In this study RTFD was shown to increase with thiacloprid use, while the proportion of larger carabids decreased. These findings suggest that thiacloprid is having an impact on the carabid assemblage function. The finding that the orchard assemblage is characterised by species more tolerant of disturbances, i.e. with more 'response' type traits, agrees with the conclusions of Letardi (2015), who likened the orchard carabid community to those usually found in more disturbed, arable habitats. Hanson *et al.* (2016) have shown increased management to be associated with reduced size and greater dispersal abilities, both responses of the assemblage to disturbance. Further, the loss of larger species is anticipated before smaller ones in more managed landscapes (Ribera *et al.* 2001; Kotze and O'Hara 2003) and, as such, these findings support previous research. Where carabid mediated pest control in orchards has been investigated, larger species (*Pterostichus* sp.) were more commonly used for study. However, the findings presented here suggest that communities responding to pest management regimes, and in this case, dominated by smaller species can still contribute to pest control services in orchards. Nonetheless, decreased predator size will ultimately restrict prey choice (Ball *et al.,* 2015) so thiacloprid use may reduce the pest control capacity of the assemblage in the longer term.

3.7. Summary and recommendations

The findings of this study indicate that the structure and function of the carabid assemblage present within the study orchards is influenced by the identity of the adjacent land use and the diversity of the surrounding land use, though overall activity-density is not. Chemical pest management impacted upon the assemblage, though was not associated with lower activity-density overall, indicating a degree of tolerance in the assemblage to this type of disturbance. The influence of the surrounding land use is less marked than that which has been observed by researchers in other agricultural systems. The semi-permanent nature of orchards may support relatively stable arthropod communities when compared to other crops, and as such we do not see the same levels of spill-over seen in arable settings, for example. Importantly, this study highlights the potential role of an assemblage dominated by opportunistic, polyphagous carabids in pest suppression in orchards. Diet flexibility and disturbance tolerance, both traits identified in the orchard assemblage observed, would allow carabids to predate upon certain pests on the ground that would otherwise be missed by other predatory species. Further research investigating communities of pests and carabids in combination is recommended to build understanding of the role of orchard carabid communities. Nonetheless, ecosystem enhancements to benefit these important predators are recommended.

Though land use diversity at a landscape scale cannot be modified by individual growers, simple modifications to the orchard habitat will increase opportunities for carabids. Beetle banks have been shown to boost predator numbers, diversity and predation in arable systems (MacLeod *et al.,* 2004). Unmanaged areas of scrubby, woody vegetation with tussocky grasses could be brought into the orchard environment to improve resources for arthropods. Such features will offer a range of feeding, sheltering and nesting opportunities for carabids and other beneficial species, such as earwigs, shown to be associated with natural/semi-natural land uses in Chapter 4.

All of the orchards in the study were conventionally managed, such that the sward height in the grass alleys was uniformly low. It is recommended that grass cutting height be raised to ≥ 15 cm to reduce the negative impacts of mowing on epigeal communities. Further study into the effects of insecticide use, particularly thiacloprid, is recommended. The use of thiacloprid is still under investigation, though the findings presented here, and in the following chapter, indicate a negative impact of this chemical on beneficial species.

Chapter 4 The European Earwig *Forficula auricularia* (Dermaptera: Forficulidae) in apple orchards: The influences of land use diversity at local and landscape scales on abundance and pest regulation potential.

4.1. Abstract

The European earwig *Forficula auricularia* Linnaeus (Dermaptera: Forficulidae) provides important pest regulation services within apple orchards, predating several pests of economic importance. However, drivers of earwig abundance within apple orchards are not fully understood. This study explored the influences of adjacent land use and land use diversity at landscape scale, as well as chemical pest regulation as potential determinants of earwig populations. A comparison of survey methodologies was undertaken and the role of earwigs in orchard pest control was investigated through field trials.

Earwig populations varied according to adjacent land use, with greater populations associated with orchards adjacent to natural/semi-natural land uses. The influence of adjacent land use was reduced with distance into the orchard, in line with the known dispersal range of the species. Insecticide use was higher in orchards in more homogenous landscapes. Applications of thiacloprid and flonicamid led to reductions in earwig abundance in the field, though chlorpyrifos had no effect on numbers. Pest predation was greater where earwig abundance was greater. Pest control was also influenced by surrounding land use, with increased pest control recorded in orchards in more heterogenous landscapes. Refugia were found to be the most effective method of assessing earwig populations in the field.

The findings presented here reaffirm the role earwigs can play in natural pest regulation in apple orchards. The study shows the potential value of incorporating natural/semi-natural habitats in orchard plantings to support and enhance earwig populations. New data has been presented which shows flonicamid to be detrimental to earwig populations. Sensitive spray timings and the use of refugia could be utilised by growers to protect earwig populations in their orchards and benefit in turn from increased natural pest control services and reduced chemical dependency.

4.2. Introduction

The European earwig *Forficula auricularia* Linnaeus (Dermaptera: Forficulidae) is the most commonly occurring of the four earwig species found in the UK (Phillips, 1981). Frequently found in apple orchards, the omnivorous *F. auricularia* feeds on vegetative matter, lichens, fungi and other arthropods (Phillips, 1981; Solomon *et al.,* 2000). Whilst *Forficula auricularia* is associated with some damage in soft-fruit orchards (Dib *et al.,* 2016a), the pest control benefits provided in apple orchards is considered to outweigh occasional damage of fruit in these systems (Solomon *et al.,* 2000). The determinants of earwig distribution in commercial apple orchards are not fully understood (Cross *et al.,* 2015), and as such, this important predator will be the focus of this study.

4.2.1. Earwig phenology

It was thought that different populations of *F. auricularia* may have different breeding strategies (Lamb and Wellington, 1975). However, DNA analysis has shown that *F. auricularia* represents two sibling species. One produces a single brood per season (single brood populations - SBP), the other producing two broods (double brood populations - DBP) (Wirth *et al.,* 1998). In both cases, earwigs pair up and mate in late autumn in a relatively complicated courtship (Walker and Fell, 2001). The female will excavate a subterranean nest, and the pair over-winter together in the nest. Sperm can be stored for a period of months, but eggs are eventually fertilised (Phillips, 1981) and the male leaves or is ejected from the nest by the female in spring when the first (or only) batch of eggs are laid (Moerkens *et al.,* 2009).

Helsen *et al*. (1998) developed a phenological model to enable prediction of earwig appearance and numbers in orchards. Numbers observed in the trees were found to be correlated with the temperature sum in day-degrees at 6°C or above since January 1st. Based on this model, peak population of the first brood (at 4th instar stage) is expected in trees between 600 – 750 day-degrees. Both SBP and DBP followed a similar pattern of emergence. In SBP oviposition occurs before winter when soil temperature drops below 5°C. Egg development is postponed, however until soil temperature increase above 5.3°C such that there is overlap with the DBP which oviposit in spring, and egg development is triggered when soil temperature reaches 6°C (Moerkens *et al.,* 2011).

Forficula auricularia displays a high degree of maternal care, with the female regurgitating food for the nymphs until mid-summer, when she vacates the nest and either dies or mates for a second time (Gomez and Kolliker, 2013; Kolliker and Vancassel, 2007; Lamb and Wellington, 1975; Staerkle and Kölliker, 2008).

Whilst there is undoubtedly some competition for resources amongst siblings; research has shown these associations can also aid development through allo-coprophagy (Falk *et al.,* 2014), though maternally derived food is preferred. Maternal care has been found to increase time spent in the nest before dispersal, to such an extent that prematurely abandoned nymphs will seek out alternative family groups (Kolliker and Vancassel, 2007). Nymph mortality increases in the absence of maternal care, though it is not clear to what extent this is a result of starvation, reduced defence against predation or disease, or through the prevention of premature dispersal before individuals are capable of self-maintenance (Dib *et al.,* 2016c; Kolliker and Vancassel, 2007). Nymphal development is dependent on ambient temperatures experienced in the soil and vegetation while nesting and foraging, as well as the composition of the diet with a mixed animal/vegetation diet leading to faster growth (Helsen *et al.*, 1998; Phillips, 1981).

The young enter the 'free-foraging' stage from late spring onwards, migrating to the trees to feed (Gobin *et al.*, 2008; Staerkle and Kölliker, 2008). Free foraging begins at the second instar stage in DBP, and at the third instar stage in SPB with increased maternal care causing SBP to spend more time in the nest (Moerkens *et al.*, 2009). During the free foraging stage nymphs begin to display both the tendency to aggregate and negative geotropism, gregariously occupying shelters and often living exclusively in trees (Philips 1981, after Atwell 1927).

The aggregation behaviour is facilitated by pheromonal signals given off at all life stages, though a precise compound triggering such behaviour has yet to be identified (Hehar *et al.*, 2008; Quarrell *et al.,* 2016; Sauphanor and Sureau, 1993; Walker *et al.*, 1993). There is little evidence of negative competition effects, rather, such aggregation is thought to provide multiple benefits to the species including higher frequency of mating interactions, enhanced protection from predators (Sauphanor and Sureau, 1993) and maintenance of familial interactions (Gomez and Kolliker, 2013).

F. auricularia will seek shelter during the day in any feature that provides narrow, dark spaces with tolerable ranges of heat and humidity. They can be found in sites such as dead plant matter, cracked or damaged trees and bark, flower heads, fruit clusters or rolled leaves (Phillips, 1981). Moerkens *et al.* (2009) speculated that pear orchards offered more sheltering opportunities than apple owing to the nature of the trees and fruit development. Several researchers have shown that rolls of corrugated cardboard, either housed or unhoused, can provide useful alternative resting structures to enabling more accurate population monitoring of *F. auricularia* (Cuthbertson and Murchie, 2013). It is likely that numbers sampled from canopies will be positively correlated with numbers found in shelters and provide a useful indicator of population size.

4.2.2. Earwigs in apple orchards

F. auricularia will emerge from shelters after dark and begin foraging in the trees. Various authors have documented the diversity of the earwig diet (Madge and Buxton 1976; Phillips 1981; Fitzgerald *et al.* 2010; Dib *et al.* 2011; Romeu-Dalmau *et al.* 2012). In some systems earwigs are considered pest species, potentially foraging or resting on crops, thus causing damage through injury or defecation (Saladini *et al.,* 2016). In apple orchards, vegetative matter forms a significant part of the earwig diet, and though some research has attributed apple damage to *F. auricularia* (e.g. Paternotte 1993 in Solomon *et al.,* 2000), it is generally accepted that such occurrences represent secondary damage caused to already injured fruit (Helsen *et al.,* 1998). Indeed, Fitzgerald and her co-workers explored the diet of *F. auricularia* using molecular screening techniques and found that apple tissue did not form a part of the diet of the earwigs sampled (Fitzgerald *et al.*, 2010).

Numerous studies have shown *F. auricularia* to be a voracious predator of many orchard pest species, capable of regulating populations; e.g. codling moth *Cydia pomella* (Unruh *et al.,* 2016), rosy apple aphid *Dysaphis plantaginea* (Dib *et al.,* 2011); woolly apple aphid *Eriosoma lanigerum* (Lordan *et al.,* 2014, Nicholas *et al.*, 2005), mussel scales (McLeod and Chant, 1952), spider mite *Panonychus ulmi* (Phillips, 1981) and apple leaf-curling midge *Dasineura mali* (He *et al.,* 2008). Many have concluded that *F. auricularia* provides a beneficial pest control service outweighing any potential damage e.g. (Romeu-Dalmau *et al.*, 2012).

4.2.3. Earwig distribution and mortality

Several authors have investigated the factors limiting earwig distribution and survivorship. Indigenous to Europe, western Asia and North Africa, temperature and humidity restrict *F. auricularia* to relatively temperate regions. Though found in the Mediterranean, Australia and North America today, *F. auricularia* is restricted to areas with an annual rainfall of at least 50 cm and average July temperature of around 24ºC (Crumb *et al.,* 1941). In a recent study Moerkens *et al.* (2011) highlighted the importance of soil temperatures for over-wintering; showing a negative relationship between the number of Cooling Day Degrees CDD and earwig numbers. CDD is similar to the day-degree model, using a cumulative value of days under a crucial temperature threshold (Moerkens *et al.,* 2012).

Moerkens *et al.* (2012) conclude that earwig mortality is largely determined by just three or four key factors. However, their work showed that cold temperatures over winter (accumulated CDD) had the most significant negative impacts on overwintering females, potentially causing a reduction in numbers of between 60% – 90%. The team also attributed notable dips in population during the later nymphal stages to predation and/or cannibalism during the last moult. They noted that, although predatory activity is possible given the vulnerability presented during moulting, there is little evidence of any predator capable of causing the sudden dips observed and thus it is more likely that cannibalism has a significant impact on population growth. Small effects of farm management (tillage) and parasitism were also noted. These conclusions are based on the results of work looking at natural and human causes of mortality in *F. auricularia* and the findings of several previous studies undertaken by the team (Moerkens *et al.,* 2011, 2010, 2009).

Inter-orchard variability in invertebrate communities, including earwigs, has been attributed to the use of a arrange of different agro-chemicals, i.e. insecticide/acaricides and plant and fungal regulators (Malone *et al.,* 2017a). The results of that study were not specific to natural enemies. However, Hill *et al.* (2017) showed that applications of broad spectrum insecticides to be associated with a reduction in natural enemy communities. This in turn, was thought to lead to secondary pest outbreaks. Other authors too have shown detrimental effects of insecticide application on non-target species, e.g. Pisa *et al.* (2014), who reviewed the literature regarding the impacts of neonicotinoids and fipronil.

As earwigs are nocturnal, seeking shelter during daylight hours (Phillips, 1981), they are less likely to come into direct contact with pesticides during application. Nonetheless, residues may be encountered during foraging and chemicals could be consumed via contaminated prey or plant material.

Looking at the impacts of conventional pest management on earwigs in kiwifruit orchards, Logan *et al* (2011) found a reduction in numbers of approximately 60% in the field for each spray application of an insecticide deemed to be ‘toxic’ to earwigs, among them chlorpyrifos and thiacloprid. In a similar study, Vogt *et al.* (2009 cited in Biondi *et al.,* 2012) found the number of earwigs was significantly lower (52%) in orchards treated with spinosad compared with the untreated control.

Whether chemical applications cause direct mortality or mortality through other indirect effects (e.g. chlorantraniliprole induced reduction in mobility (Freitas *et al.* 2017)) is not known. However, lethal, non-lethal and cumulative negative effects of insecticide applications have been demonstrated in the laboratory. Chlorpyrifos and thiacloprid were tested by Maher and Connolly (2009), and indoxacarb, thiacloprid, spinosad and diazinon by Shaw and Wallis (2010), with both studies demonstrating reduced earwig survivorship with applications of these chemicals. In a study combining laboratory assays and a small field survey, Fountain and Harris (2015) observed reduced nymphal development in the presence of spinosad, spirodiclofen, thiacloprid and methoxyfenozide, while chlorpyrifos, thiacloprid and flonicamid were found to reduce numbers of earwigs foraging in trees.

Consequently, while natural processes evidently account for a proportion of earwig mortality throughout the lifecycle, farm management, in particular chemical pest control, is considered likely to be an important determinant of earwig populations in commercial orchards.

4.2.4. Earwigs and habitat

Beyond that related to climatic conditions, there is limited information in the literature regarding the specific habitat requirements, or preferences, of *F. auricularia*. It is documented that the species is found across a variety of habitats ranging from urban to agricultural landscapes. However, the negative geotropism displayed by the species appears to produce a preference for arboreal living. This, combined with dietary preferences, means that *F. auricularia* can be particularly common in fruit crops. This is discussed at length by Crumb *et al.* (1941) and Phillips (1981), for example, together with many other authors already referred to in this text (e.g. Madge and Buxton 1976; He *et al*. 2008; Fitzgerald *et al*. 2010; Dib *et al*. 2011; Romeu-Dalmau *et al*. 2012, Saladini *et al*., 2016)

The dispersal range of *F. auricularia* is low compared to other arthropods (Moerkens *et al.,* 2010) and is largely thought to be influenced by either food availability or environmental stimuli. Crumb *et al.* (1941) found increased dispersal when *F. auricularia* was faced with desiccation while Lamb and Wellington (1975) found *F. auricularia* will not return to a refuge once local food resources are exhausted. These effects can be mediated to a degree in the presence of suitable shelter. Research has shown *F. auricularia* can survive for weeks without food, and consequently dispersal behaviour is altered where more sheltering opportunities exist (Moerkens *et al.,* 2009). In a mark-recapture experiment, Moerkens *et al.* (2010) showed that dispersal of SBP and DBP of *F. auricularia* differed markedly, though distance travelled by males and females of both cohorts did not. Results suggested a mean dispersal range (for 95% of released individuals) of 28.6m for SBP, and just 7.5m for DBP populations. This study was undertaken in an orchard system, however Crumb *et al.* (1941) recorded a similar range of movement in studies undertaken in a grass turf habitat (race track), with average distances travelled ranging from 9 m – 22 m. Lordan *et al* (2014) placed live earwigs in shelters and found the densities of 0.2 individuals/cm^2 will lead to the release of sufficient quantities of aggregation pheromone to attract earwigs for up to 5 weeks in a 0.5 m radius of the shelter, supporting the hypothesis that dispersal range is limited and largely dictated by food availability.

A growing body of research is highlighting the role of land use diversity in influencing invertebrate communities at the field scale, with knock-on impacts for natural pest suppression in agro-ecosystems (Bianchi *et al.*, 2006; Chaplin-Kramer *et al.*, 2011; Rusch *et al.,* 2016). Much of the work published to date has focused on arable systems, with few studies exploring the relationships between land use and invertebrate communities within perennial crops. Helsen *et al.* (1998), Gobin *et al.* (2008), Moerkens *et al.* (2012), Lordan (2014), Dib *et al.* (2016) and others recognise the incomplete understanding of the drivers of *F. auricularia* distribution and discuss various influencing factors such as farm management, climatic conditions, food availability, predation and disease. However, while some have looked at the relationship between hedgerow composition and beneficial orchard arthropods (Rieux *et al.* 1999; Debras *et al.* 2008; Senoussi *et al.* 2011; Dib *et al.* 2016), very few studies have specifically explored the influences of land use diversity at a large scale, or the neighbouring land use type on *F. auricularia* abundance in orchards, one exception being the study of Stutz and Entling (2011) in cherry orchards.

Although *F. auricularia* is frequently found within orchards, numbers are often unpredictable (Cross *et al.,* 2015). Based on the literature discussed above, it is considered likely that *F. auricularia* displays a preference for natural/semi-natural habitats (e.g. woodlands) which provide opportunities for food and shelter suited to the earwigs' phenology. Therefore, more diverse landscapes rich in such habitats may support greater numbers of earwigs overall. As such, the diversity of the land use at a large scale surrounding an orchard may influence the populations of *F. auricularia* found within it. As earwig dispersal ranges are relatively low, the influence of neighbouring land uses may be more evident on populations of earwigs within orchards.

4.3. Study aims

This study has four main aims. Firstly, to compare the efficacy of alternative sampling techniques in monitoring earwig populations in orchards. Secondly, to investigate the influence of land use diversity at a landscape scale and the influence of adjacent land use type, on earwig populations within orchards. Thirdly, to assess the influence of chemical pest management on orchard earwig populations. And lastly, to assess the potential contribution of earwig populations to pest regulation within orchards.

4.4. Methodology

4.4.1. Experimental design

The study was carried out within eight commercially managed farms in Kent, southern England. Four are situated on the North Kent Plain and four in the Weald (Wealden Greensand and Low Weald). Study farms were selected based on specific criteria to allow an assessment of the influences of the surrounding land use at different spatial scales on orchard earwig populations.

At the landscape scale, study farms were classified in terms of the diversity of the surrounding land use within a 1 km radius. For a full methodology on land use characterisation, along with details of site locations and study plot layouts, refer to Chapter 2.

The overall diversity of land use at the landscape scale was found to be highly positively correlated with the proportion of agricultural land and negatively with the area of unmanaged, (i.e. natural/semi-natural) habitats within the 1 km study area. In order to explore the influence of land use at a more local level, study plots within farms were selected based on the adjacent (i.e. within < 25 m) land use. Within each farm three distinct plots were chosen separated by a minimum of 250 m; one adjacent to an area of intensive agricultural land, one adjacent to an area of less intensively managed land and one adjacent to further apple orchard planting.

Within these plots two transects 100 m in length were established to monitor earwigs; one parallel with the orchard boundary, between the first and second row of orchard planting bordering the adjacent (≤ 25m) land use (henceforth 'edge') and a second transect parallel to this but a further 50 m into the orchard planting (henceforth 'interior'). This allowed for an estimation of the effects of distance from differing land use to be considered. See Figure 2.2 in Chapter 2 for an illustration.

Survey work was carried out over three years (2013 – 2015) at approximately 20-day intervals, although all methodologies were not employed in all years. Earwig populations were monitored in three ways: diurnal canopy sampling, nocturnal canopy sampling and using artificial refugia.

4.4.2. Measuring earwig populations

4.4.2.1. Canopy sampling

Canopy sampling was undertaken by vigorously striking a single tree branch twice with a padded stick (tree 'beating' or 'tapping'). Each study transect (8 farms x 3 plots x 2 transects) was walked in a zig-zag pattern and 30 randomly selected trees were sampled along the route, avoiding those with refugia already present. Specimens were collected using a large handheld net (1 m x 0.5 m) with a collecting bag attached. Diurnal tap sampling was undertaken on dry days between May and August 2013 & 2014. Canopy sampling such as this is a sampling technique widely used in apple orchards for pest and natural enemy assessments. However, the largely nocturnal habit of earwigs is likely to limit the numbers of earwigs observed using diurnal sampling. Alternative methods were also employed to account for this and to provide an opportunity to compare survey methodologies.

Nocturnal tap sampling surveys were undertaken in 2014 to allow comparison of diurnal and nocturnal canopy populations and those recorded in refugia. Following the same methodology as that used for diurnal tap sampling, surveys were undertaken at least half an hour after sunset on dry nights with a minimum temperature of 12^{o}C. Logistical constraints meant only a partial survey was achieved; 6 study orchards were surveyed in May; all eight study orchards were surveyed in July and August.

4.4.2.2. Refugia sampling

To gain an improved understanding of earwig populations within the study orchards, three artificial refugia were placed along each transect in April 2014. Refugia consisted of a roll of corrugated cardboard cut to size and placed inside a 0.5l plastic bottle with the lid sealed and the bottom removed, (see Fig 4.1). A single refuge was attached to the trunk of a central tree at three positions (between 0 m – 20 m, 40 m – 60 m and 80 m – 100 m) along each transect. Refugia were set at the base of the canopy, at a height of approximately 0.5 m, facing towards the grass alleys to allow surveyor access.

Refugia were checked once a month by carefully removing the corrugated cardboard and gently blowing or tapping earwigs out over a white tray and counting/sexing the individuals present. Earwigs were returned to the same refugia where possible or returned to the base of the same tree. Refugia were returned to the trees after survey. Surveys were undertaken between May and September 2014 and April and August 2015.

4.4.3. Measuring the effects of land use and earwig abundance on pest control

Following the methodology employed in Chapter 3, live sentinel aphids *Acyrthosiphon pisum* (Hemiptera: Aphididae) were used to measure variations in arboreal predation in relation to land use at local and landscape scales. Chapter 3, section 3.4.5 provides a full methodology.

Bait cards were deployed at dusk in a layout which mirrored the existing experimental design; a single bait card was attached to the trunk of a randomly selected tree at 30 m intervals along the existing transect, avoiding those which had refugia attached. Cards were set flush against the branch at a height of approximately 130 cm – 150 cm (surveyors shoulder height) and secured with garden wire, Fig 4.1. They were collected 12 hours later, and the proportion of aphids consumed was recorded.

The survey was repeated once a month from April through to August of 2015 on dry nights only.

Figure 4.1. *A pair of images showing (left) a trial aphid bait card deployed in an orchard, and (right) an artificial earwig refugia in situ on an apple tree.*

4.4.4. Measuring insecticide usage in orchards

The full methodology employed to establish insecticide usage in orchards is given in Chapter 3, section 3.4.6.

4.4.5. Analysis

Statistical analyses were undertaken in R (R Development Core Team, 2016). Non-parametric Wilcoxon Signed-rank tests were used to compare survey methodology. The 'lme4' package (Bates *et al.*, 2016) was used to design a range of linear mixed-effects models to explore the influences of land use and chemical management on earwig populations. Predation was assessed as a binomial response using glmer models from the 'lme4' package. Models were validated graphically using residual plots to ensure assumptions of normality and homogeneity of variance were met. The response variables tested were peak earwig abundance, peak count of females, peak count of males and the peak count of juveniles. The response data were log-transformed in order to ensure model assumptions were met.

To assess the impacts of land use, the explanatory variables tested were survey year, orchard age, land use diversity at the landscape scale, adjacent land use type and location of survey transect (i.e. distance from orchard boundary). Interactions between land use factors were included initially and then excluded if found not to be significant. Three-way interactions were tested but found to be non-significant so are not considered further.

Owing to either single, small dose applications or inter-correlations between some chemicals a reduced dataset was used to assess the influence of insecticide usage on earwig abundance. The acaricides/insecticides tested were thiacloprid, chlorpyrifos, flonicamid, spirodiclofen and carbamates (i.e. pirimicarb or fenoxycarb). TFI values (i.e. total number of insecticide treatments, weighted by the ratio of the dose used to the recommended dose) have been calculated per study plot per year (see Chapter 3, section 3.4.6). Fungicides were not considered beyond preliminary analysis. To account for the nested structure of the experimental design, land use factors were included as random effects in the models.

The 'Anova' function in package 'car' produces an analysis of deviance with Wald chi-square type II tests and was used to make inferences about the model parameters. *Post hoc* testing was undertaken using Tukey's HSD tests.

4.5. Results

Over the three-year sampling period, across all farms, the only earwig species recorded was *F. auricularia*.

Using diurnal tap sampling, the total catch across all study plots in 2013 was 294 earwigs and in 2014, total catch was 480 earwigs. A Wilcoxon Signed-rank test did not find this difference to be significant (W = 12595, p-value = 0.1459).

The numbers of earwigs recorded using refugia sampling also varied between survey years. In 2014 a total of 5863 earwigs were recorded across all plots, in 2015 a total of 2909 were recorded. A Wilcoxon Signed-rank test found this variation to be significant, with greater numbers of earwigs recorded in 2014 compared to 2015 (W = 14268, $p < 0.001$).

Nocturnal sampling was carried out in 2014 only and yielded a total catch of 771 individuals across all study plots.

The three sampling techniques employed in this study were compared using the 2014 data as all three techniques were employed concurrently in this year.

A comparison between nocturnal and diurnal sampling found approximately 40% more earwigs were observed at night, for the same survey effort (i.e. following the same technique as diurnal tap sampling). A Wilcoxon Signed-rank test showed a significant effect of survey technique (W = 16744, $p < 0.001$).

Pearson's Product-moment tests found correlations between all three survey methods i.e. positive correlations between total counts recorded in diurnal sampling, nocturnal sampling and via refugia ($p < 0.001$ in all cases). As refugia sampling contributed the greatest number of individuals caught, these data form the focus of the results presented. However, all three datasets were investigated using mixed-effects models to ensure no factors influencing earwig distribution in orchards were being overlooked by using the refugia data alone. No variables were found to significantly influence earwig numbers using the diurnal and nocturnal datasets that had not been identified using the refugia dataset. As such, earwig abundance data in this chapter is based upon the counts recorded from refugia sampling alone.

4.5.1. The effects of land use diversity at a landscape scale and the influence of adjacent land use type, on earwig populations within orchards.

Neither adjacent land use nor land use diversity at the landscape scale were found to be significant drivers of earwig numbers. More earwigs were recorded in more diverse settings at the landscape scale. Study plots adjacent to unmanaged, (i.e. semi-natural/natural) land uses supported higher numbers overall (mean 152.68, s.e.m 34.18), followed by plots adjacent to additional orchard planting (mean 97.06, s.e.m 26.98), with plots adjacent to agricultural land producing the lowest counts (mean 54.94, s.e.m 19.50).

A significant interaction between adjacent land use type and transect location (i.e. the proximity to that land use) was observed (Wald χ^2 = 7.61, d.f = 2, p = 0.022). Post-hoc analysis using a Tukey's HSD test showed the main significant variance (p = 0.034) to be between populations found in the interior of plots next to agricultural land uses, and those at the edge of plots next to unmanaged land uses (see Figure 4.2). Further, a difference which may be of biological relevance, though not statistically significant (p = 0.086) was found between populations in the interior of plots next to additional orchard planting and those in the interior of plots next to agricultural land uses, Fig. 4.2. The remaining interactions were not significant (p > 0.05).

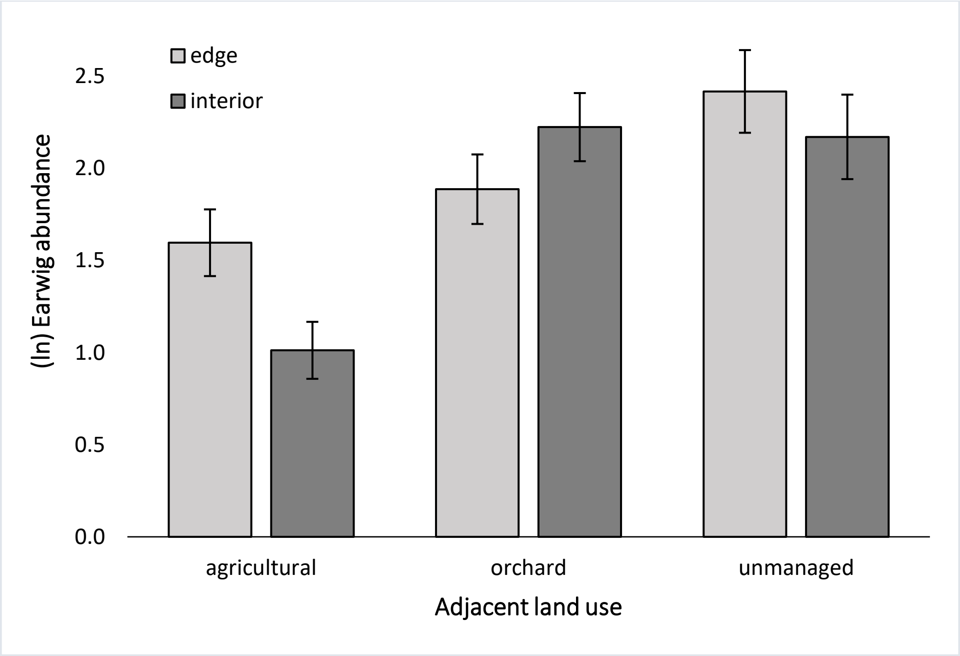


Figure 4.2: *A bar plot showing the influence of the interaction between transect location and the adjacent land use on earwig abundance. Earwig abundance was significantly lower in transects in the interior of agricultural plots compared to those at the edge of unmanaged plots (Wald χ2 = 7.61, p = 0.022). Error bars denote the standard error of the mean.*

4.5.2. The Effects of Insecticide Usage on Earwig Abundance

Insecticide usage during this study was found to vary significantly according to land use diversity and survey year (Generalised linear mixed effects model, $F_{2, 381} = 214.5$, $p < 0.001$). Lower overall insecticide usage was recorded in landscapes with a higher diversity of land use at a 1 km scale (high diversity: mean TFI for insecticides 1.79, s.e.m 0.32, low diversity: mean 2.12, s.e.m 0.24). Further, insecticide usage was higher in 2014 (mean 2.95, s.e.m 0.07) compared to 2015 (mean 0.95, s.e.m 0.06), Fig 4.3. The adjacent land use was not found to be related to insecticide usage.

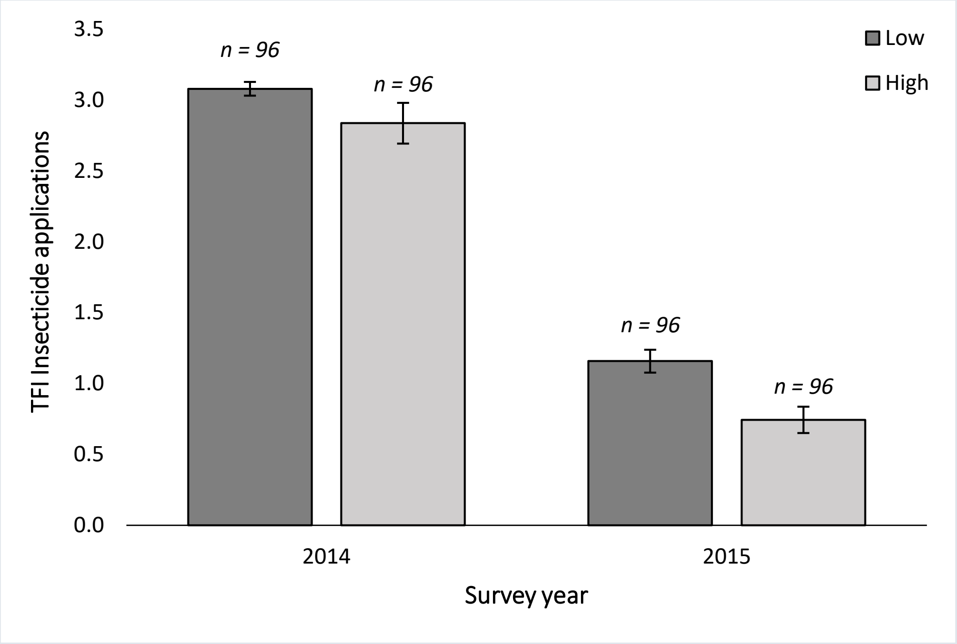


Figure 4.3: *A bar plot showing the interaction between the diversity of land use at the 1 km scale (low vs. high) and survey year on the annual Treatment Frequency Index (TFI) for insecticides. Insecticide use was greater in the first survey year, 2014, in low land use diversity settings (p < 0.001 for both). Error bars denote the standard error of the mean.*

The cumulative TFI for insecticide usage during the study was not found to influence overall earwig abundance. However, as insecticide usage was shown to vary significantly with survey year, TFI indices for each survey year were tested independently. As such the impact of individual insecticides on earwig abundance was established using the TFI values calculated for each product applied in each survey year.

Analysis of the 2014 data found the TFI for overall insecticide usage was not significantly related to earwig abundance. However, increases in the TFI for flonicamid were found to be associated with significantly lower earwig counts overall (Wald $\chi^2 = 14.389$, d.f = 1, $p < 0.001$). The most parsimonious model included thiacloprid and carbamates, though flonicamid was the only insecticide found to significantly influence overall earwig numbers in 2014 (Figure 4.4).

Increased usage of flonicamid was also associated with significantly lower counts of males (Wald $\chi^2 = 7.572$, d.f = 1, $p = 0.006$), females (Wald $\chi^2 = 9.475$, d.f = 1, $p = 0.002$) and juveniles (Wald $\chi^2 = 15.091$, d.f = 1, $p<0.001$) when these groups were tested individually. The numbers of juvenile earwigs recorded was also found to be significantly reduced with increased use of thiacloprid (Wald $\chi^2 = 5.00$, d.f = 1, $p = 0.025$, Fig. 4.5).

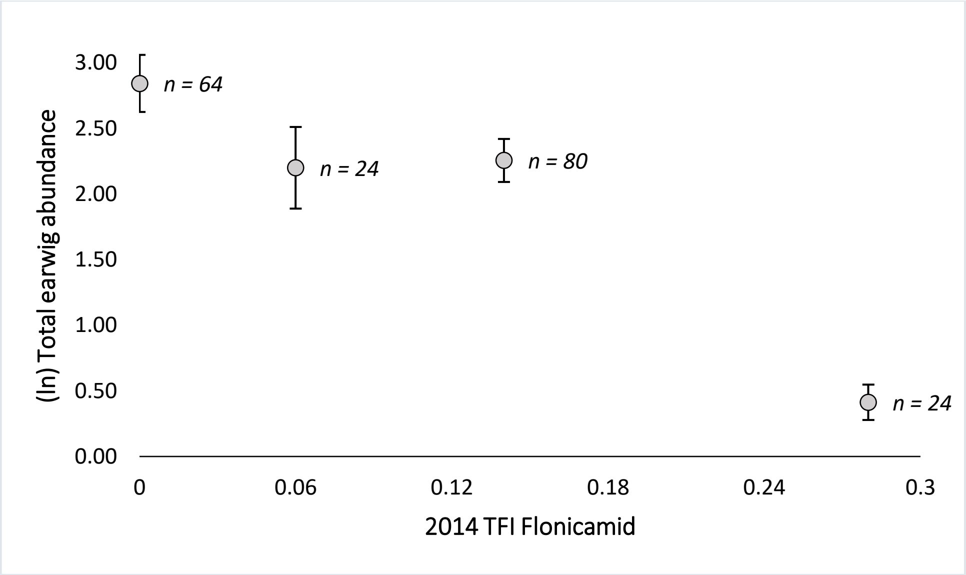


Figure 4.4: *A scatter plot showing the impact of flonicamid usage (2014 TFI) on overall earwig abundance in study orchards in 2014. Increased flonicamid usage was associated with fewer earwigs overall (Wald* $\chi2 = 14.389$*,* $p < 0.001$*). Error bars denote the standard error of the mean.*

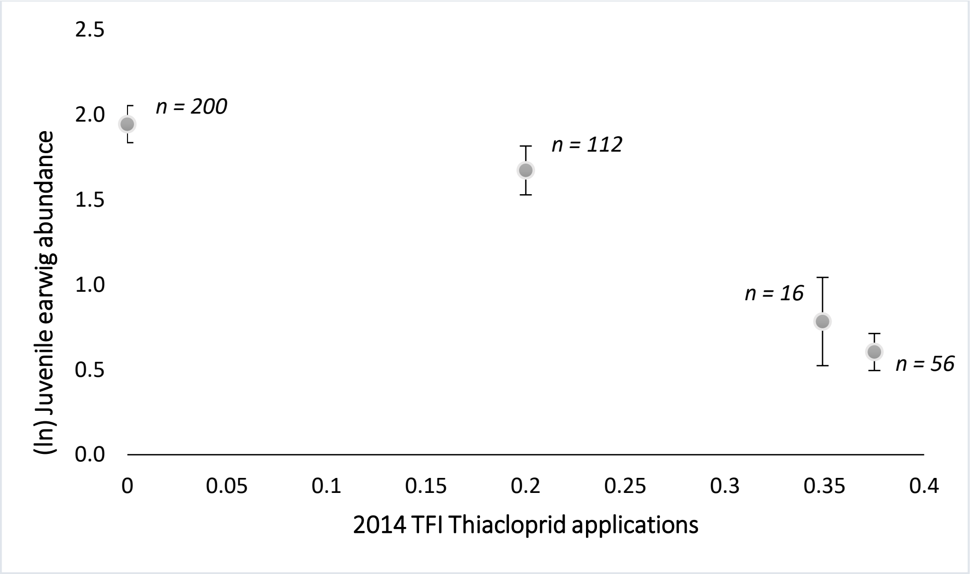


Figure 4.5: *A scatter plot showing the impact of thiacloprid usage (2014 TFI) on the abundance of juvenile earwigs in study orchards in 2014. Increased thiacloprid usage was associated with fewer juvenile earwigs (Wald χ2 = 5.00, p = 0.025). Error bars denote the standard error of the mean.*

Analysis of the 2015 data found the TFI for overall insecticide usage was significantly related to earwig abundance, with fewer earwigs recorded overall where the cumulative TFI was higher (Generalised linear mixed effects model, $F_{1, 46} = 4.25$, $p = 0.045$, Fig 4.5). When assessing the impacts of chemicals independently, increased use of thiacloprid was found to be associated with significantly lower earwig counts overall, (Wald $\chi^2 = 22.57$, d.f = 1, $p < 0.001$), Fig. 4.6). Further, significantly lower counts of males (Wald $\chi^2 = 15.16$, d.f = 1, $p < 0.001$), females (Wald $\chi^2 = 13.21$, d.f = 1, $p < 0.001$) and juveniles (Wald $\chi^2 = 18.68$, d.f = 1, $p < 0.001$) were also observed with increased use of thiacloprid when these groups were tested individually.

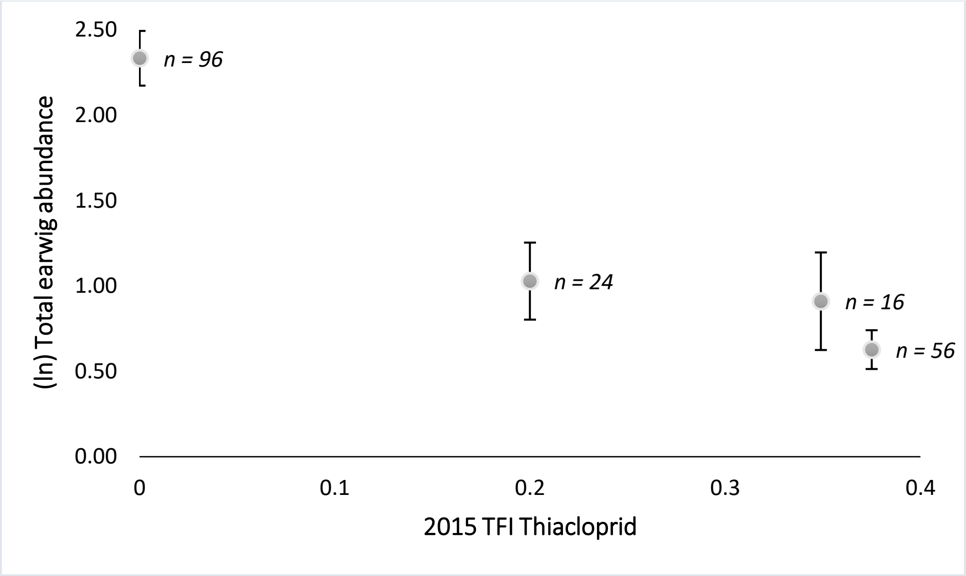


Figure 4.6: *A scatter plot showing the impact of thiacloprid usage (2015 TFI) on the total abundance of earwigs in orchards in 2015. Increased thiacloprid application was associated with fewer earwigs (Wald χ2 = 22.57, p < 0.001). Error bars denote the standard error of the mean.*

The most parsimonious model included methoxyfenozide and carbamates, however thiacloprid was the only insecticide found to significantly influence overall earwig numbers in 2015.

Using Kruskal-Wallis tests, applications of thiacloprid were found to be higher ($\chi^2 = 15.768$, d.f = 1, $p < 0.001$) in 2015 (mean TFI 0.16, s.e.m 0.01) than in 2014 (mean TFI 0.09, s.e.m 0.01). Applications of flonicamid were higher ($\chi^2 = 32.034$, d.f = 1, $p < 0.001$) in 2014 (mean TFI 0.10, s.e.m 0.01) compared to 2015 (mean TFI 0.05, s.e.m 0.00).

4.5.3. The Effects of Earwig Abundance and Land use at Landscape and Local Scales on Natural Pest Control in Orchards

Live aphids were used as bait to assess the influences of earwig abundance and land use on pest control in orchards. The diversity of land use at the landscape scale and the adjacent land use type were found to significantly influence levels of aphid predation. Further, greater numbers of earwigs were found to relate to higher levels of aphid predation.

Predation (i.e. the number of aphids consumed per transect) was found to be significantly greater (Wald χ^2 = 12.11, d.f = 1, $p < 0.001$), in landscapes with a higher diversity of land use at a 1 km scale (high diversity: mean aphids consumed 66.46%, s.e.m 3.40, low diversity: mean aphids consumed 53.63%, s.e.m 3.66).

There was no significant difference in predation between orchards according to adjacent land use types or transect location. A significant interaction effect was observed however between adjacent land use type and transect location where predation was found to be higher in the 'edge' transects of plots adjacent to additional orchard planting or unmanaged land (Wald χ^2 = 10.55, d.f = 1, $p = 0.005$). Post-hoc analysis using a Tukey's HSD test showed a significant difference between predation recorded at the edge of study plots adjacent to orchard planting, and that recorded in the interior of study plots adjacent to unmanaged land ($p = 0.039$). The difference in predation levels recorded between edge and interior transects of plots next to unmanaged land was not found to be significant, however the result was nearing statistical significance ($p = 0.082$), which may hint at a biological relevance (Fig. 4.7).

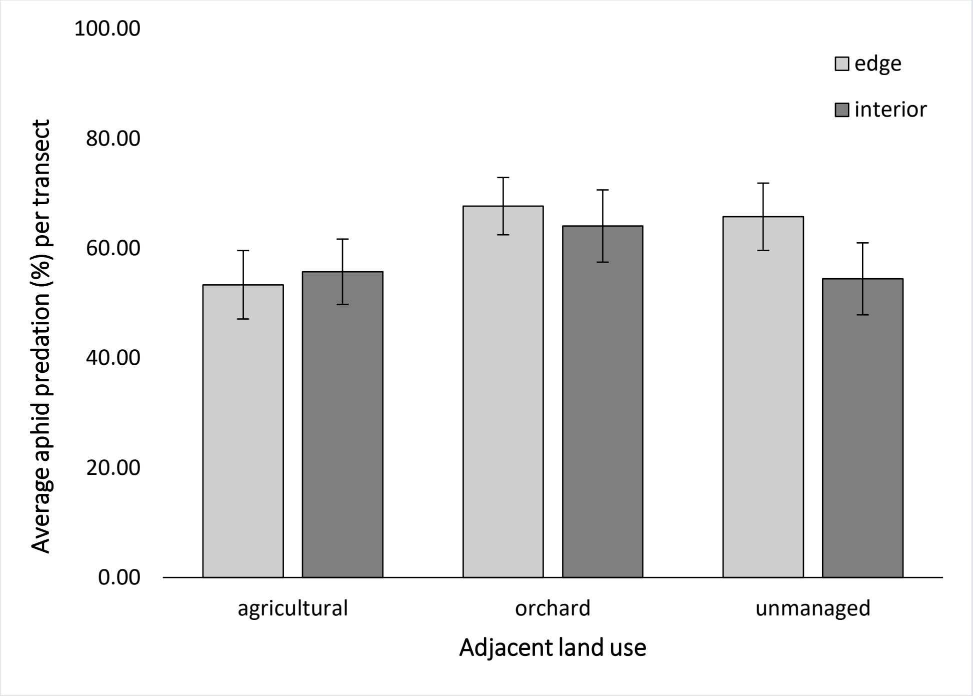


Figure 4.7: A bar plot showing the influence of the interaction between adjacent land use and the diversity of land use at the 1 km scale on aphid predation. The percentage of aphids predated per transect was recorded to be significantly higher in edge transects of edge of study plots adjacent to orchard planting, compared to that recorded in the interior of study plots adjacent to unmanaged land (Wald χ2 = 10.55, p = 0.005). Error bars denote the standard error of the mean.

Higher numbers of earwig were found to be related to significantly higher levels of aphid predation (Wald $\chi^2 = 73.29$, d.f = 1, $p < 0.001$, Fig. 4.8). Post-hoc testing showed the number of earwigs to account for a much greater proportion of variance in the model than the interaction of adjacent land use and transect location. When tested individually, the abundance of adult males (Wald $\chi^2 = 26.10$, d.f = 1, $p < 0.001$), adult females (Wald $\chi^2 = 23.55$, d.f = 1, $p < 0.001$) and juvenile earwigs (Wald $\chi^2 = 78.21$, d.f = 1, $p < 0.001$) were also found to be related to significantly greater levels of predation. It should be noted that the number of earwigs recorded in refugia provides an indicator of population density and not an absolute measure. As such, earwig related predation may be observed within tree canopies where refugia data indicate earwig counts of zero earwigs.

The most parsimonious model included survey month which was found to relate to aphid predation with the proportion of aphids predated increasing from May (mean 36.48, s.e.m 4.77) through to August (mean 87.11, s.e.m 3.34). Earwig abundance and survey month are intercorrelated, as survey month reflects the natural growth in earwig populations. As such, survey month is not considered an independent factor relating to predation levels here.

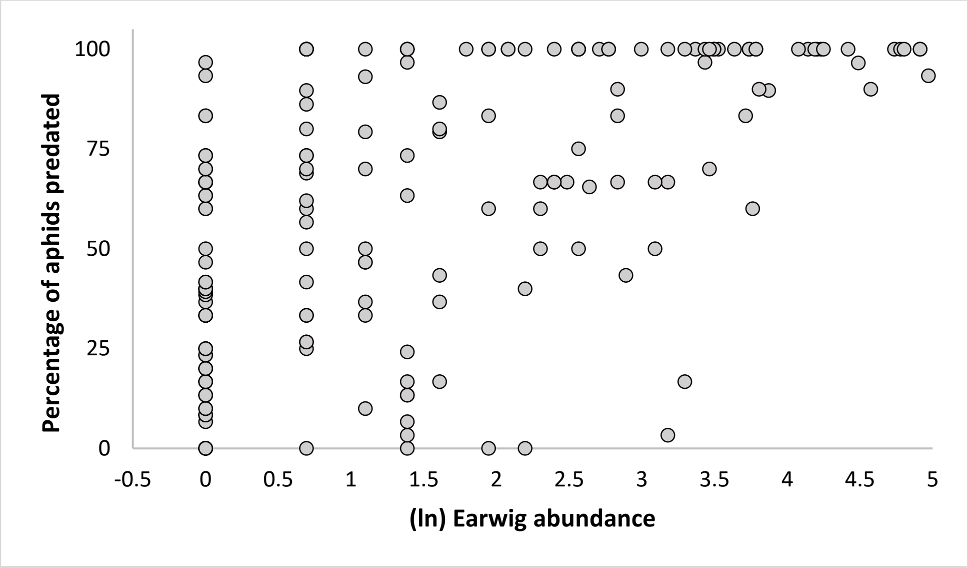


Figure 4.8: *The percentage of aphids predated in relation to the earwig abundance. Higher numbers of earwigs were found to be related to greater levels of aphid predation (Wald χ2 = 78.21, p < 0.001).*

4.6. Discussion

4.6.1. Comparison of earwig sampling techniques

The Agriculture and Horticulture Development Board in the UK recommends tree beating as a means of assessing a range of pests and beneficial arthropods in apple trees (Agriculture and Horticulture Development Board, 2011). As such the method is used by growers and agronomists alike to estimate pest pressure in particular, and to a lesser degree the presence or absence of beneficial arthropods. However, owing to the nocturnal habit of earwigs, this sampling technique is likely to underestimate earwig populations. Consequently, the majority of studies looking at the populations of earwigs present in orchards employ refugia of some description as the primary method for estimating populations (e.g. Philips, 1981, Moerkens *et al.* 2009, Quarrell et al., 2017). This study took the opportunity to compare the efficacy of diurnal and nocturnal tree beating with refugia sampling as means of estimating populations.

Beyond the initial set up phase, refugia sampling requires considerably less survey effort in the field, both in terms of the time and equipment required to obtain data, when compared to tree beating as traps can remain in-situ indefinitely. What is more, neither beating method produced high counts of earwigs, such that greater survey effort produced fewer individuals. Further, all factors shown to influence earwig populations recorded in the canopy were also observed from the refugia data, indicating these samples are representative of the wider population.

These data show correlations between the numbers recorded in the canopy and those recorded in refugia, suggesting beat sampling can be indicative of earwig populations. However, the numbers recorded using diurnal beating in this study were too low to allow detailed consideration of the drivers of earwig populations. While nocturnal beating produced greater numbers for a comparable survey effort, this method poses certain logistical problems and still does not provide the same catch as that which can be derived using refugia.

Based on the results presented here, refugia sampling appears to be the most effective means of estimating earwig populations within orchards, providing reliable estimates of abundance. It should be noted that refugia could be considered an enhancement for earwigs in orchards and as such may boost populations, potentially skewing results. However, in this study there was no indication that the refugia were causing a long-term increase in numbers, as abundance actually fell between year 1 and year 2. Further, numbers recorded via canopy sampling correlated with refugia throughout the study. Canopy sampling recorded earwigs along the length of the 100 m study transect while refugia were only present on three trees in that transect. It is thought therefore that if refugia were drawing earwigs from further afield we would see a) increasing populations between year 1 and year 2 and b) a mismatch between canopy abundance and that recorded in refugia. Based on this it is considered that the refugia are providing a resource for the local population only.

Nonetheless, a simple refuge is an easy and low-cost method of supplying opportunities for earwigs in orchards. These devices can facilitate beneficial exchanges as described by Sauphanor and Sureau (1993) and Gomez and Kolliker (2013) such as increased familial and mating interactions and greater protection from predation. Beyond catering to the aggregative tendencies of *F auricularia*, refugia will also offer shelter from farm management, particularly where plantations are young and natural refuge opportunities will be lower.

The use of shelters and dispersal of earwigs is considered by some to be largely attributable to the availability of food resources. For example Lamb and Wellington (1975) state that earwigs will not continue to use a shelter once local food has been exhausted. However, Moerkens *et al.* (2009) found that earwigs can survive for several weeks without food and that more sheltering opportunities can therefore alter dispersal behaviour. If earwig dispersal from refugia is dictated by food resources, it may be possible to move established populations within refugia and 'deploy' them in areas of the orchard experiencing a pest outbreak. However, adequate provision of shelters in the immediate area combined with a food boom resulting from the pest outbreak should encourage earwigs to remain within the vicinity of the outbreak. Dib *et al.* (2016) trialled a small scale release of mass reared earwigs as a potential control for *Dysaphis plantaginea* however, several constraints were identified. The efficacy of this as a method of natural pest control would benefit from additional research in future.

4.6.2. The effects of land use at landscape and local scales on earwig abundance

Neither the diversity of land use at a landscape scale, nor adjacent land use type were found to influence earwig abundance within orchards when considered in isolation. However, adjacent land use was found to affect earwig populations when considered together with the distance from that land use. Specifically, earwig populations recorded at the 'edge' of study plots adjacent to 'unmanaged' land uses (i.e. natural and semi-natural habitats) differed significantly from those recorded in the interior of study plots next to intensive agricultural land uses.

While there was no direct effect of land use diversity at the landscape scale, the greater numbers observed in orchards in more diverse landscapes suggest there may be a broader influence of land use diversity. In this study, more homogenous landscapes were dominated by agricultural land. Agricultural land is subject to higher disturbance and increased chemical inputs which will result in lower levels of food and habitat resources. As a consequence, homogenous landscapes are less likely to support earwigs. In this study fewer earwigs were indeed observed in orchards in landscapes with a lower diversity of land use. Though the effect was not significant, the variation lends weight to the hypothesis that earwig populations respond to land use.

The low dispersal ranges of *F. auricularia* will inevitably limit the scale at which such effects are observable. Stutz and Entling (2011) found earwig densities to be higher in cherry orchards that were less isolated from woody habitats. However, the impact of woody/semi-natural habitats on earwigs was not detectable at a wider landscape scale of 500 m. In this study the effects of land use on earwig populations was found to be significant when examined at a more local scale as an interaction between adjacent land use and the distance from that land use. Fewer *F. auricularia* were recorded where orchards abut agricultural land, with higher numbers recorded in orchards adjacent to natural/semi-natural land uses or additional orchard planting. In conjunction, populations were higher in transects nearer to these non-crop land uses suggesting migration of individuals from offsite habitats into orchards that is somewhat restricted by distance. Senoussi *et al* (2011) found the abundance and richness of orchard arthropod communities decreased with distance from planted species-rich hedgerows in pear orchards. In turn, Dib *et al.* (2016) found numbers of earwigs decreased with distance from orchard hedgerows. The neighbouring land use appears to give rise to *F. auricularia* actively foraging within orchards. In agricultural settings this is likely to be a result of lower opportunities in agricultural land forcing earwigs to forage or seek refuge in neighbouring orchards. However, in line with Dib *et al.* (2016), the larger populations near natural/semi-natural land uses are likely to reflect the benefit of non-orchard planting in boosting earwig populations within orchards.

4.6.3. The effects of earwig abundance and land use at landscape and local scales on natural pest control in orchards

From the findings presented in this study, it would appear the same principles apply to earwigs as seen with other generalist predators in relation to land use. In agreement with research highlighting the role of landscape heterogeneity in supporting natural enemy mediated pest control in agriculture (Bianchi *et al.*, 2006; Chaplin-Kramer *et al.*, 2011; Rusch *et al.,* 2016), canopy predation was found to be lower in orchards in more homogenous landscapes. Adjacent land use and the distance from those land uses were also influencing factors; though not significant, there was evidence of a trend towards greater predation in the edge transects of orchards adjacent to natural/semi-natural or orchard land uses.

Earwigs are known predators of aphids and other pests and have been shown to contribute to pest suppression in the field. A number of studies have identified earwigs as key predators of the woolly apple aphid; Mueller *at al.* (1988) observed that higher density earwig populations discovered and eliminated *E. lanigerum* colonies more rapidly than lower density populations, while exclusion of earwigs led to increases in *E. lanigerum* infestation of 20 – 25%. Nicholas *et al*. (2005) also found *F. auricularia* to be the principal control agent for *E. lanigerum* under an IPM system. Gut content analysis of a number of generalist predators also found codling moth to form a substantial part of the earwig diet, being recorded in 14.7% of the earwigs tested (Unruh *et al.*, 2016). *F. auricularia* have also been shown to consume rosy apple aphid in the lab (Dib *et al.,* 2011) and apple leaf curling midge in the field (He *et al.,* 2008).

It must be stated that other arboreal predators may have contributed to aphid removal observed in this study. In a field experiment in citrus orchards, Romeu-Dalmau *et al.* (2012) excluded 'earwigs', and 'earwigs and other predators' from the canopies to assess predation contribution. The authors concluded that earwigs were responsible for pest suppression as aphid density was negatively correlated with earwig abundance and aphid colonisation was lower where earwigs were present under these controlled conditions. Although such data was not collected here, the predation observed in this study was found to strongly relate to earwig numbers recorded in orchards, which would seem to agree with the findings of others showing earwigs to be effective predators.

Martin *et al.* (2015) found pest control to be dependent on landscape complexity and natural enemy interactions. In our study, the land use factors associated with predation largely reflect the factors found to influence earwig abundance in orchards. Indeed, the interaction between adjacent land use and earwig population size was found to influence orchard predation. The findings indicate that earwig mediated pest control is influenced to an extent by land use in the surroundings.

4.6.4. The effects of insecticide usage on earwig abundance

In this study the use of insecticides was found to be influenced by the diversity of land use at the landscape scale; with greater diversity being associated with lower insecticide use.

This agrees with research indicating reduced landscape heterogeneity is associated with reduced natural enemy communities, and in turn poorer natural pest control (Rusch *et al.,* 2016) and therefore increased spray applications by growers. Insecticides use was not found to be related to adjacent land use. Growers' spray records suggested a tendency to spray larger areas than an individual orchard block to ensure pest outbreaks are supressed (pers. obs.). It is considered therefore that the lack of variation according to adjacent land use reflects that tendency.

In contrast to other similar studies (Marliac *et al.,* 2015a; Thomson and Hoffmann, 2006), no estimations of the relative toxicity of chemicals applied (i.e. the cumulative toxicity index based on the International Organisation of Biological Control [IOBC] mortality rankings) were made in this study. The species used as test subjects under IOBC rankings are dominated by Hymenoptera and Homoptera. Further, the chemicals investigated in those studies are only occasionally applied in the UK, and many have already been banned. As such, it was thought TFI estimations would provide new information on the toxicity of insecticides to the study species in the field.

In this study, insecticide applications were found to negatively influence the earwig populations recorded in orchards, with different chemicals associated with declines in each survey year. In 2014 flonicamid was shown to have the greatest effect on earwig numbers, while in 2015 thiacloprid was shown to have the greatest effect.

In a review of the knowledge on neonicotinoids, Pisa *et al* (2014) showed consistent negative effects, both lethal and sub-lethal, on a range of non-target invertebrate families including pollinators and natural enemies. Both flonicamid and thiacloprid are neonicotinoid class insecticides. The impacts of thiacloprid on earwigs has been demonstrated by a number of authors (Logan *et al*., 2011; Maher and Connolly, 2009; Shaw and Wallis, 2010) all concluding reduced survivorship with thiacloprid application. Fountain and Harris (2015) found thiacloprid to reduce nymphal development in laboratory tests and lead to less active foraging in trees when assessed in the field. The results presented here agree with previous researchers, with lower earwig populations recorded with increased use of thiacloprid. Lower numbers of juveniles were recorded in 2015 with increased thiacloprid use, and although not significant, thiacloprid application remained in the most parsimonious model for juvenile earwigs in 2014.

There are fewer studies looking at flonicamid and its impacts on non-target species, and earwigs in particular. In an arable study, Jensen *et al.* (2011) found flonicamid to be a selective aphicide, causing no significant reductions in the Coleoptera or Hymenoptera tested. In contrast, when looking at the impacts on earwigs, Vogt *et al.* (2010) found flonicamid to be associated with earwig declines of 40%. This decline was the lowest observed in a study of four chemicals; nonetheless it does represent a significantly negative decline. In a more recent field trial Fountain and Harris (2015) also observed reduced foraging activity with increased flonicamid use. The findings presented here add weight to the evidence that flonicamid is harmful to earwigs.

Interestingly chlorpyrifos was not found to relate to earwig abundance in this study. The reasons for this are not immediately clear as chlorpyrifos has been shown to have negative effects on earwig numbers. Exposure to chlorpyrifos was shown to reduce the predation activity of earwigs under laboratory conditions (Malagnoux *et al.,* 2015a). In a similar study Maher and Connolly (2009) postulated a cumulative effect of chlorpyrifos exposure on earwig survivorship, as mortality increased with time under laboratory conditions. This was also observed in Fountain and Harris (2015), with all individuals dying after two weeks exposure.

It may be that the detrimental effects are only discernible with prolonged exposure which is unlikely to occur in the field. Fountain and Harris (2015) is the only study directly looking at the effects of chlorpyrifos on earwigs in the field. Although the study noted fewer females and nymphs in refugia in plots where chlorpyrifos was applied, it is unclear if this took account of population shifts across the season (i.e. decrease in juvenile earwigs as the population matures).

This study suggests earwigs in the field are not as vulnerable to chlorpyrifos applications as expected. The negative effects of chlorpyrifos may be difficult to determine in the field as earwigs may not come into contact with chlorpyrifos or may not maintain exposure for sufficient time to cause a measurable effect. Further, if chlorpyrifos use is reducing earwig activity, as opposed to having an instant lethal effect, exposed earwigs may reasonably be expected to be present in refugia, such that the effects are not apparent from refugia data.

4.7. Summary and recommendations

Earwigs have consistently been found to have pest control capability in orchards. The findings presented here, showing a decline in aphids where earwig numbers are higher, would seem to support previous studies suggesting that earwigs act as pest control agents in conventionally managed apple orchards. Some of the variability in earwig populations in apple orchards noted by other researchers appears to be attributable to land use diversity. This could be exploited by growers to enhance natural pest control services in orchards through the provision of additional resources for earwigs. The effects of adjacent land use on earwigs appears to diminish beyond approximately 25 m, as such unmanaged areas of woody or scrub habitat could be incorporated into orchards to break-up plantations, enhancing the ecosystem for earwigs and providing resources for other natural enemies to boost natural pest control.

It has also been demonstrated that refugia provide an efficient method of measuring populations, which require less survey effort than conventional tree tapping methods, while still producing valuable results. Simple earwig refugia could be deployed in orchards to enhance opportunities for sheltering earwigs, exploiting their natural aggregative tendencies and potentially reducing exposure to insecticides.

The results of this study confirm that harmful effects of flonicamid and thiacloprid which have been observed in laboratory studies are evident in the field. While the need for some chemical pest control is unavoidable, careful timing of applications will reduce impacts on beneficial species.

Chapter 5 The effects of mowing frequency and height of cut on communities of predatory arthropods in a commercial dessert apple orchard.

5.1. Abstract

In general terms, taller grassland supports a greater number of individuals and diversity of arthropod species than short mown grassland. Yet to reduce competition from weeds and reduce opportunities for pests, the conventional method of ground management in orchards is to mow frequently to a low height. Not only is such frequent management costly, it diminishes sward structure and plant diversity, reducing natural enemy abundance and therefore potentially limiting pest control.

By comparing different mowing regimes, this experiment assessed whether minor alterations in orchard floor vegetation management could enhance communities of natural enemies in a young commercial Braeburn/Gala apple orchard in Kent, UK. Canopy communities were unaffected by treatment in this study. However, epigeal predators in the understory increased significantly under reduced mowing. Any reduction in the intensity of mowing produced positive results. Maintaining standard mowing frequency (2 – 3 times per month) but raising the cutting blades to 15 cm produced similarly positive results as mowing seasonally (i.e. only once a year). The results presented here suggest small alterations in management; raising the blades of the mower to 15 cm, or reducing mowing frequency, could have significant positive implications for predator numbers in apple orchards, which may in turn influence pest control.

5.2. Introduction

Commercial apple orchards are intensively managed cropping systems, which typically are subject to some of the highest chemical application rates in agriculture, necessary to manage pests and reduce competitive effects from other vegetation (Garthwaite *et al.,* 2016; Simon *et al.*, 2010). Commercial dessert apple varieties are shallow rooted plants which will suffer ill effects from competition for water and nutrients when vegetation is allowed to remain in the tree rows (Granatstein and Sánchez, 2009) and so these areas are treated with herbicide regularly. In contrast, vegetation in the inter-row alleys (henceforth 'alleys') is allowed to persist as it provides protection of the soil from damage by farm machinery, reduces erosion and run-off, and supports nutrient cycling (Granatstein and Sánchez, 2009). However, there is concern that non-crop vegetation may encourage pests, so alleys are generally un-sown and frequently mown (Horton *et al.,* 2003). Some pest species are associated with particular plant species in the ground vegetation, for example, Rosy Apple Aphid *Dysaphis plantaginea*, relies on plantain *Plantago spp.* (mainly *P. lanceolata*) for its development (Dib *et al.*, 2010). Such relationships have led growers to be fearful of allowing 'weed' species to proliferate, however, research indicates that maintaining even a limited ground cover can boost numbers of natural enemies in orchards (Markó and Keresztes, 2014; Rieux *et al.*, 1999).

Non-crop habitats surrounding arable farms can be associated with increases in beneficial arthropods, without leading to increases in pest numbers, suggesting that enhancements both within and around a crop are likely to deliver benefits to the crop (Bianchi *et al.,* 2006; Chaplin-Kramer *et al.,* 2011). Orchards, although heavily managed, are more persistent habitats than annual crops and they support relatively stable arthropod communities (Brown, 1993). It may be the case therefore, that, unlike those of arable crops, orchard arthropod communities are more responsive to farm level management than the composition of the wider landscape.

The abundance of floral resources in the spontaneous, self-sown, vegetation in orchard alleys can influence the orchard arthropod community (Rosa García and Miñarro, 2014). This relationship could potentially be exploited to increase ecosystem services through appropriate management of the alleys. Several researchers have investigated the use of cover crops to boost ecosystem service in orchards. In general, the literature highlights beneficial results from such interventions. Gomez *et al.* (2017) showed cover crops in an olive orchard enhanced soil structure and reduced run-off, as well as boosting numbers of natural enemies. In peach orchards, Wan *et al.* (2014) found a cover crop of *Trifolium repens* led to reduced pest numbers and higher numbers of natural enemies in trees, with a higher natural enemy:pest ratio overall. Aphidophagous predators such as Syrphids, Coccinellids, Chrysopids and parasitoids can increase with increased floral resources when those flowers have accessible open nectaries (Campbell *et al.*, 2012; Gontijo *et al*., 2013; Tschumi *et al.*, 2016; Campbell *et al.*, 2017). The findings of research undertaken by Bone *et al.* (2009) are an exception to this trend, as the sowing of cover crops in alleys in that study did not increase natural enemy activity, but was associated with higher levels of pests in the trees.

In an arable study, spider abundance responded positively to increases in the percentage cover of forbs in sown strips, an increase thought to be largely attributable to increased prey availability (Blake *et al.*, 2013). In orchards, understory planting with a forb-rich mix has been found to boost spider numbers in the canopy, with effects varying depending on hunting guild; also numbers of stalkers (Salticidae) rose sharply in flower rich swards whereas web builders (Theridiidae and Araneidae) appeared unaffected (Markó and Keresztes, 2014). Vegetation structure will undoubtedly play a part in influencing arthropod abundance and diversity owing to variations in micro-climate, hunting and refuge opportunities or, in the case of spiders, web-building support (Buri *et al.*, 2016). Marko and Kadar (2005) showed a structurally and botanically diverse mosaic in the orchard ground cover was associated with greater abundance and diversity in carabid assemblages.

In general, tall grassland supports a greater number of individuals and diversity of arthropod species than short grassland (Morris, 2000). However, regular mowing to a short height (< 9 cm on average, J. Fitzgerald, pers. comm. 2013) is the conventional method of ground management in commercial dessert apple orchards in the UK. It is a non-selective form of vegetation management that reduces canopy structural complexity and can destroy other structural features of potential benefit to arthropods such as ant hills or other small scale refugia (Morris, 2000). In commercial dessert apple orchards, regular mowing has been found to reduce natural pest control by predatory phytoseiid mites and may even be associated with increased fecundity in pest mites *Tetranychus urticae* (Funayama, 2014). A simple reduction in the frequency of mowing, from two or three times per month to just once, was found to lead to a greater cover of grasses and forbs in the alleys of a commercial orchard sward (Horton *et al*., 2003). Though little variation was noted in epigeal arthropod communities, some natural enemy taxa were found to be higher in less frequently mowed plots.

Similar patterns have been found in other agricultural systems. For example, Cizek *et al.* (2012) suggest that the effects of reduced mowing in intensive hay meadows will vary according to taxa. In their 2011 study, increased mowing was found to be associated with higher numbers of spiders and carabids, though in cut vegetation the spiders recorded tended to have restricted habitat requirements, while the carabid species in frequently mown swards comprised more disturbance tolerant species. Gobbi *et al.* (2015) expanded on these findings showing carabid communities shifting towards a less specialised, and more disturbance tolerant assemblage in their study. Buri *et al.* (2016) have made similar findings, showing that timely cutting can lead to increases in spiders. Lafage and Pétillon (2014) found both spider and carabid assemblages to be negatively affected by increased frequency of cutting.

Despite the aforementioned negative impacts of mowing on epigeal arthropod communities, an increase in carabid activity was noted up to 5 days following a grass cut in peach orchards (Sonoda *et al.,* 2013). The authors attributed this temporary boost in activity to natural enemies seeking out dead or injured arthropods following a mow. It follows that carabids may be more resilient to this type of disturbance as the majority found in orchards are likely to be nocturnal in habit (Kutasi *et al.*, 2004). Indeed studies by Cizek *et al.* (2012) and Gobbi *et al.* (2015) both identify a less specialist carabid assemblage associated with increased mowing, indicating a community level tolerance of disturbance. However, as Simon *et al.* (2010) have noted, an increase in natural enemies on the ground may not provide a highly functional community for pest control in orchards. Movement between ground and canopy by predators will be limited by several factors (Horton *et al.*, 2003; Schellhorn *et al.*, 2014; Simon *et al.*, 2010). However, an increase in grass and forb cover has been shown to be associated with higher natural enemies in the orchard canopy (Campbell *et al.*, 2017), which would have a direct effect on pest control. In addition, as many orchard pests spend part of their lifecycles on the ground (i.e. *E. lanigerum, T. aequatus, D. mali* and *C. pomonella)*, they are potentially vulnerable to predation by more generalist epigeal predators (Boreau de Roincé *et al.,* 2012, Cross *et al.,* 2015; Unruh *et al.,* 2016) and so there is an argument for enhancing this community regardless of readily identifiable effects within the tree canopy. Mowing is undertaken in orchards to a degree that is considered unnecessary to reduce competitive effects from vegetation or limit secondary hosts for pests (J. Fitzgerald, pers. comm. 2013). Reduction in the mowing regime will reduce levels of disturbance and enhance vegetation structure and composition in the alleys. As such, a reduction in mowing may offer a cost-effective method of enhancing natural enemy communities in the orchard. It is likely that modifying mowing frequency and height will have knock-on benefits for pest control in the orchards. As Horton *et al.* (2003) found, even a reduction in mowing frequency of around 30% can cause an increase in natural enemies.

5.3. Study aims

This experiment aimed to assess whether minor changes in orchard alley vegetation management can significantly enhance the numbers of predatory arthropods in the tree canopies and on the orchard floor, and consequently contribute to pest regulation services in commercial apple orchards.

5.4. Methodology

5.4.1. Experimental design

The study was undertaken within a commercial apple orchard at the 220 ha demonstration farm at East Malling Research, Kent, England (51.291231 N, 0.442135 E). The orchard comprises approximately 1.8 ha of a Braeburn - Gala mix planted in 2012. The alleys comprise spontaneously-established vegetation dominated by grasses. The ground under the tree rows remained bare throughout the course of the study, being frequently treated with herbicides to remove vegetation.

To the north the orchard is bordered by grassland, to the west by a field used for cultivation of strawberries and to the south and east by *Alnus sp*. windbreaks and additional tree fruit (apples and pears), Figure 5.1. Prior to the beginning of this experiment, the orchard had been subject to 'business as usual' mowing (2 – 3 times a month between April and October) and chemical application during the previous year.

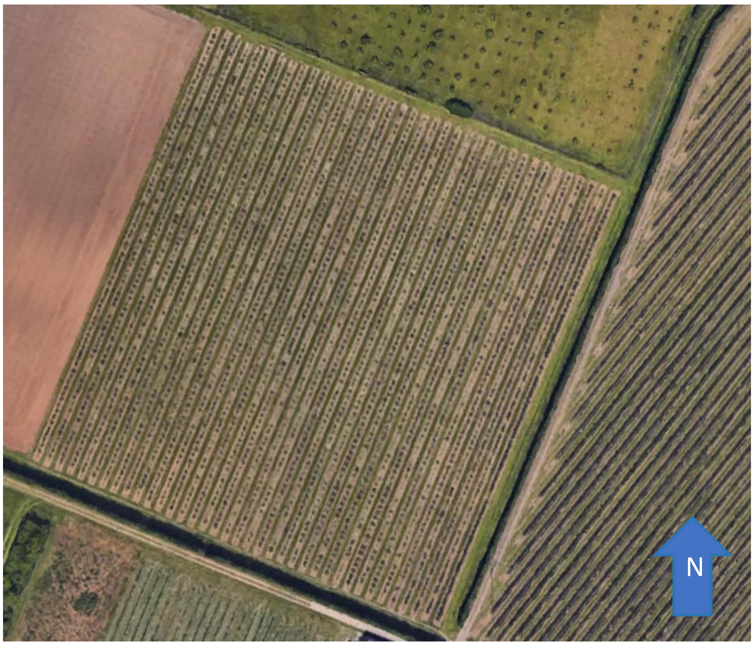


Figure 5.1: *Aerial image showing the experimental plot and adjacent plots at East Malling Research station.*

The experiment was based on a randomised complete block design comprising four treatments with four replicates of each arranged in a Latin square design. The treatments comprised one control and three alternative mowing regimes. [1] control 'business as usual' at the farm, a mow 2-3 times a month between April and October with the sward cut to a low height (≤ 9 cm), [2] mowing at the same frequency as control, with the blades raised to a height of 15 cm, [3] a monthly mow between April and October with the sward cut to a low height (≤9 cm) and [4] a biannual cut immediately pre-blossom (April) and pre-harvest (September), with the sward cut to a low height (≤9 cm) Fig 5.2. Survey plots were nine tree rows wide and eighteen trees in length, approximately 30 m x 30 m.

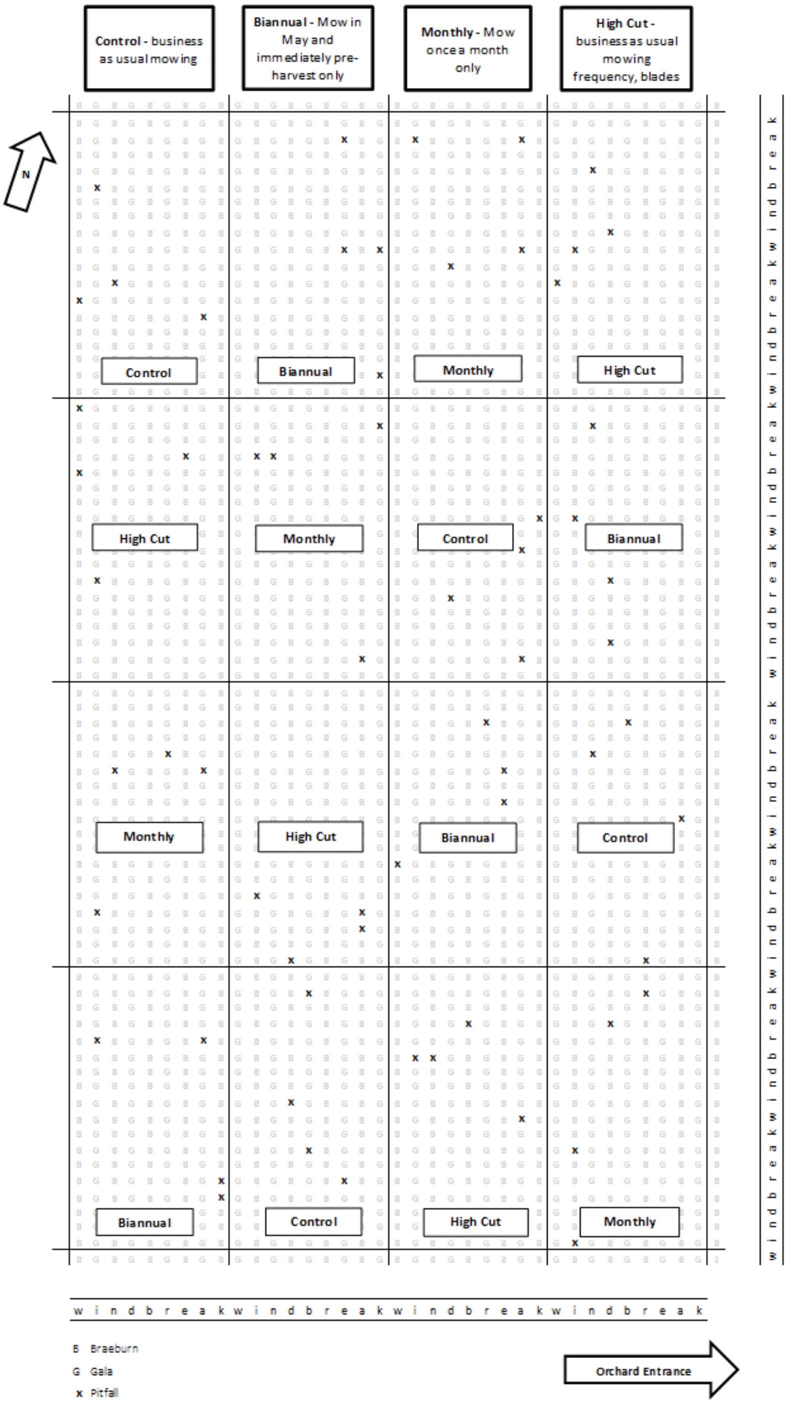


Figure 5.2: *Mowing experiment plot layout showing pitfall locations and treatment allocations.*

5.4.2. Assessing epigeal communities

Epigeal arthropod communities were quantified using pitfall traps. Four randomly-located pitfall traps were set within each treatment plot. Two traps were set within the grass alleys of each plot, two within tree rows. Traps consisted of a removable plastic inner cup set within a harder outer cup (diameter of 80 mm, depth 105 mm). Traps were set flush with the ground and filled to a depth of about 70 mm with a 70 % dilution of propylene glycol, as preservative, and a small amount of detergent to break surface tension.

Pitfall traps were opened for 7 days at a time, at approximately 20 day intervals during the apple growing season, between May and August 2013 and 2014. Trap catch was collected with samples drained and stored in 70 % ethanol. Catch from the pitfall traps in the tree rows in each plot was combined, as was that from the grass alleys. Predators were identified to family and, with the exception of the genus *Amara,* all carabids were identified to species following the nomenclature of Luff (2007).

5.4.3. Assessing canopy communities

The communities of arboreal predators present were assessed using 'tap sampling', undertaken by vigorously striking a single tree branch twice with a padded stick and collecting the dislocated invertebrates into a net (1 m x 0.5 m) with a collecting bag attached. This was repeated on 60 trees, randomly selected by following a rough zig-zag path through each treatment plot. Specimens were collected using a large handheld net with a collecting bag attached and identified to family. Tap sampling was undertaken on dry days between May and August in 2013 and 2014.

5.4.4. Vegetation survey

The composition of the vegetation in the alleys was recorded monthly from a 1 m^2 quadrat surrounding each pitfall trap (in the case of alley traps) or from the alley immediately adjacent to the tree row (in the case of row traps). The percentage cover of grasses, forbs and bare ground was estimated using vertical projection. The mean vegetation height was also estimated from each quadrat based on the maximum vegetation height (cm) recorded from four randomly placed measuring sticks. Quadrat data was collected at the beginning of each 7-day pitfall survey round.

All survey work was repeated monthly between May and August in 2013 and 2014.

5.4.5. Analysis

Statistical analyses were undertaken in R (R Development Core Team, 2016). The 'lme4' package (Bates *et al.,* 2016) was used to undertake general linear models to establish the influence of treatment on vegetation and natural enemies. Owing to their random allocation, some pitfall traps lay on the edges of treatment blocks. To ensure independence, the data from any such traps were removed prior to analysis of treatment effects. The pitfall catch did not vary between the traps set in the grass alleys and those in the tree rows so catch was totalled across each survey plot. Monthly catch was totalled for each year for each survey plot.

The explanatory variables tested were survey year and treatment, i.e. [1] 'Control', a mow 2-3 times a month with the sward cut to a low height, [2] 'High cut', mowing at the same frequency as the control, with the blades raised to a height of 15 cm, [3] 'Monthly', a single monthly mow with the sward cut to a low height and [4] 'Biannual', a twice yearly cut to a low height. The interaction term between year and treatment was also included. Although the structure and composition of the sward is related to 'treatment', for completeness individual metrics relating to the structure and composition of the sward were also tested, namely; vegetation height, the percentage cover of grasses and the percentage cover of forbs.

The response variables tested for canopy predator communities were the abundance of spiders (Araneae), ladybirds (Coleoptera: Coccinellidae), earwigs (Dermaptera: Forficulidae) and lacewings (Neuroptera: Chrysopidae) as well as a combined total predator count. For epigeal predator communities the response variables tested were the abundances of spiders, carabids (Coleoptera: Carabidae) and staphylinids (Coleoptera: Staphylinidae), and again a combined total predator count. Counts were log 10 transformed for analysis.

The 'Anova' function in package 'car' produces an analysis of deviance with Wald chi-square type II tests and was used to make inferences about the model parameters. *Post hoc* testing was undertaken using Tukey's HSD tests.

Models were validated graphically using residual plots to ensure assumptions of normality and homogeneity of variance were met.

5.5. Results

The orchard was sampled monthly over two apple growing seasons in 2013 and 2014. The arboreal predator community was small overall, with canopy sampling producing a total of 386 predatory arthropods from all plots over the two survey seasons. The catch consisted of 151 spiders, 120 coccinellids, 105 forficulids (all *F. auricularia*) and 10 chrysopids (all *Chrysoperla carnea*).

A Kruskal-Wallis test found significantly more arboreal predators in the second year of study (χ^2 = 26.894, d.f = 1, p < 0.001); 297 (mean 18.56, s.e.m 1.44) recorded in 2014, compared to 89 (mean 5.56, s.e.m 0.64) recorded in 2013, Fig. 5.3.

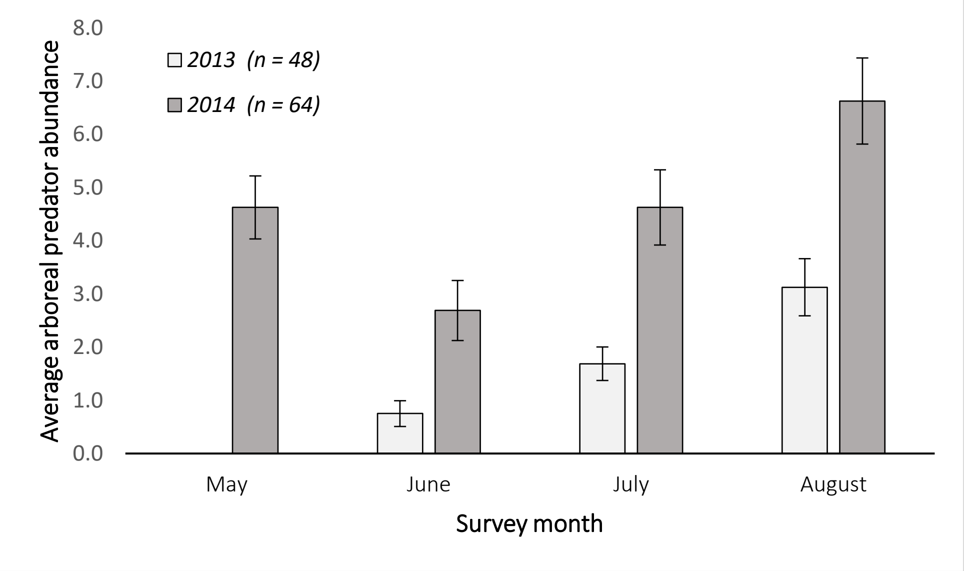


Figure 5.3: *A bar plot showing the difference in arboreal predator abundance per month (i.e. total from 60 taps per plot per month) in each survey year. Predator abundance was higher in 2014 (χ2 = 26.894, p < 0.001). Note there was no May 2013 survey. Error bars denote the standard error of the mean.*

Pitfall trapping yielded a total of 2,467 predatory epigeal invertebrates across the experiment, comprised of 1,121 carabids, 1,188 spiders and 158 staphylinids, Fig. 5.4. Higher numbers of epigeal predators were recorded (χ^2 = 4.622, d.f = 1, p =0.032) in 2013, 1358 (mean 368.4, s.e.m 4.53) than in 2014, 1109 (mean 77, s.e.m 3.16).

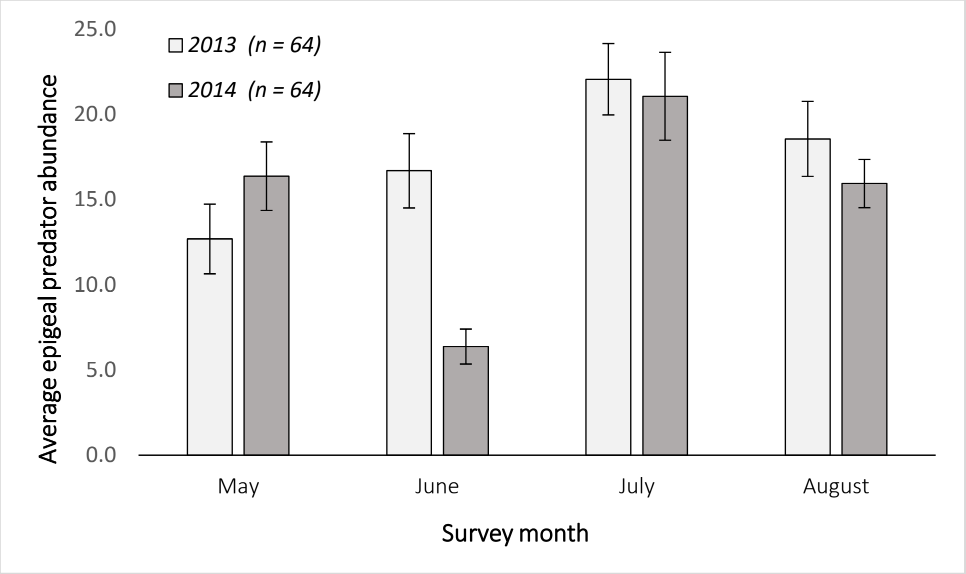


Figure 5.4: *A bar plot showing the difference in epigeal predator per month (i.e. total from 4 traps per plot per month) in each survey year (χ2 = 4.622, p = 0.032). Error bars denote the standard error of the mean.*

Only carabids were identified to species. Catch varied somewhat between years with slightly more carabids recorded in 2013, with 613 (mean 166.9, s.e.m 2.43) compared to 508 (mean 34.19, s.e.m 2.57) in 2014. A total of 15 carabid species were recorded. However, four of those were recorded just once, namely *Anchomenus dorsalis, Asaphidion flavipes, Leistus spinibarbis* and *Notiophilus biguttatus* and will not be considered further.

The carabid community was dominated by a small number of species, with *Harpalus rufipes, Nebria brevicollis, Harpalus affinis and Pterostichus madidus* representing 83% of the total catch. *H. rufipes* dominated the catch in both years, though number of *N. brevicollis, H. affinis and P. madidus* varied somewhat between years, (Table 5.1).

Table 5.1: *The numbers of carabids for the most abundant species caught across all treatments in 2013 and 2014.*

Species	2013	2014	Total
Amara sp.	21	43	64
Bembidion lampros	39	19	58
Calathus fuscipes	10	9	19
Harpalus affinis	62	88	150
Harpalus rufipes	263	229	492
Loricera pilicornis	3	4	7
Nebria brevicollis	166	27	193
Notiophilus rufipes	1	4	5
Poecilus cupreus	18	6	24
Pterostichus madidus	27	73	100
Pterostichus melanarius	3	6	9

5.5.1. Differences in the structure and composition of orchard understory vegetation

The height of the vegetation recorded in the alleys largely corresponded to each treatment, i.e. the tallest sward was recorded in the biannually mown plots, the shortest sward was recorded in the Control plots (Linear mixed effects model, *F = 21.23, df = 3, p < 0.001*). The vegetation height was similar across the two survey years, but composition varied, with grass cover increasing and cover of forbs and bare ground decreasing (Table 5.2).

Table 5.2: *Average vegetation composition (percentage cover) and height (cm) as recorded in 2013 and 2014.*

	% cover Grass		% cover Bare		% cover Forbs		Average height *(cm)*	
Year	***2013***	***2014***	***2013***	***2014***	***2013***	***2014***	***2013***	***2014***
Control	71	83.7	14	4	30.9	12.5	9	9
High cut	57.5	88	17	1.4	41	13.3	20	20
Monthly	64	85	22.9	3.6	25.7	11.7	20	10
Biannual	68.6	87.8	17.5	1.5	27	12.9	27	21

The cover of grass increased across all treatments from 2013 to 2014 (Linear mixed effects model, *F = 41.03, df = 1,* p <0.001), while forbs (Linear mixed effects model, *F = 27.85, df =*

1, p <0.001) and bare ground (Linear mixed effects model, *F = 126.49, df = 1,* p <0.001) decreased. In 2013 the greatest coverage of forbs, 41%, was found within the 'High Cut' plots, i.e. those plots cut at the same frequency as the control, but to a height of 15 cm. This treatment also saw the greatest drop in forb coverage between 2013 and 2014. By the second year the sward composition varied little across treatment, although there were significant differences in mean vegetation height between treatments, Figure 5.4.

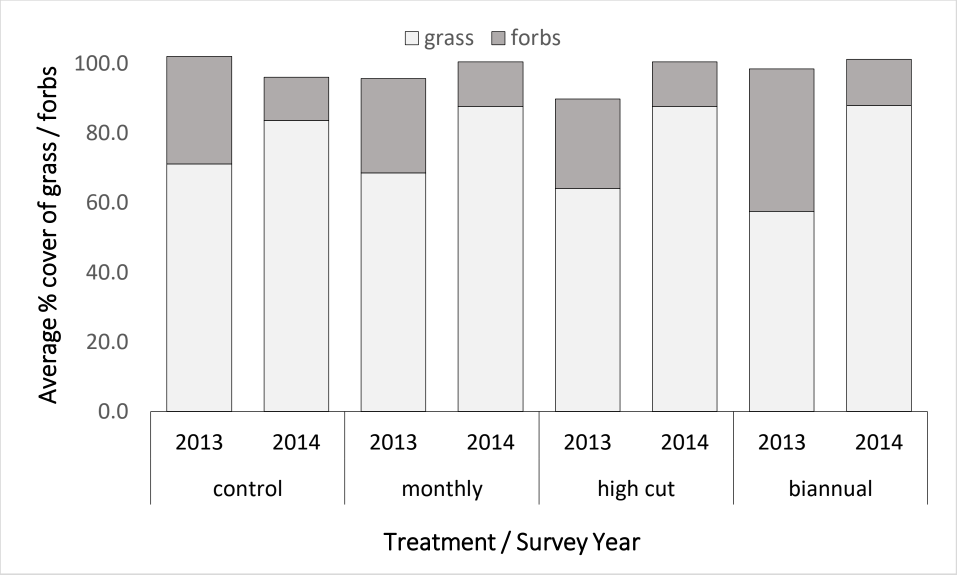


Figure 5.4: *A stacked bar plot showing the average percentage cover of grasses and forbs per treatment each survey year. There is a general trend of increased grass coverage and decreased forb coverage between the first and second survey years.*

5.5.2. Differences in arboreal predator communities

Survey 'year' was the main factor found to influence the abundance of predators in the tree canopies across treatments. Total predator abundance was significantly greater in 2014 (Wald χ^2 = 76.38, d.f. = 1, p < 0.001) compared to 2013. The arboreal predator community was not significantly influenced by 'treatment', although differences in total abundance of predators between treatments were approaching significant (Wald χ^2 = 7.073, d.f. = 3, p = 0.069). Post-hoc analysis using a Tukey's HSD test showed this near significant difference (p = 0.061) to be between the Biannual plots and High Cut plots, with slightly higher numbers of predators recorded in biannually mown plots.

The total abundance of arboreal predators was not found to be affected by either grass, forb and bare ground coverage or the height of vegetation.

5.5.3. Differences in epigeal predator communities

As with arboreal communities, survey 'year' was a predictor of epigeal predator communities. In contrast to the arboreal communities, however, epigeal predator abundance, i.e. the combined abundance of carabids, spiders and staphylinids, was found to be significantly greater in 2013 (Wald χ^2 = 5.48, d.f. = 1, p = 0.019). No significant differences were found between years in the abundances of carabids, spiders or staphylinids when tested individually.

In addition, both the total count of predators and the activity-density of carabids was significantly influenced by 'treatment'. Total predator abundance was significantly greater in all three interventions in comparison to the 'Control' (Wald χ^2 = 13.06, d.f. = 3, p = 0.004), Fig. 5.5, with the 'High Cut' supporting the greatest numbers.

Carabid activity densities in the 'High Cut' and 'Biannual' treatments were found to be significantly greater than that recorded in 'Control' plots (Wald χ^2 = 9.85, d.f. = 3, p = 0.019), Fig. 5.6.

The individual metrics relating to vegetation structure and composition were not found to show any statistically significant relationships with epigeal predator communities.

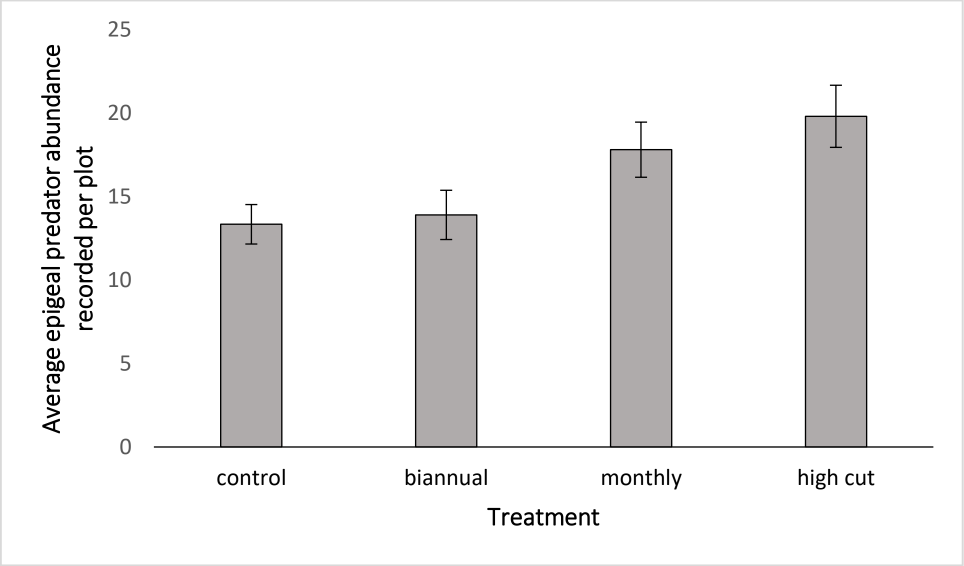


Figure 5.5: A bar plot showing the difference per treatment of the average abundance of epigeal predators recorded per plot (i.e. total catch from 4 pitfalls in each plot). Higher numbers of predators were recorded in the 'Monthly' and 'High cut' treatments when compared to the 'Control' (Wald χ^2 = 13.06, p = 0.004). Error bars denote the standard error of the mean.

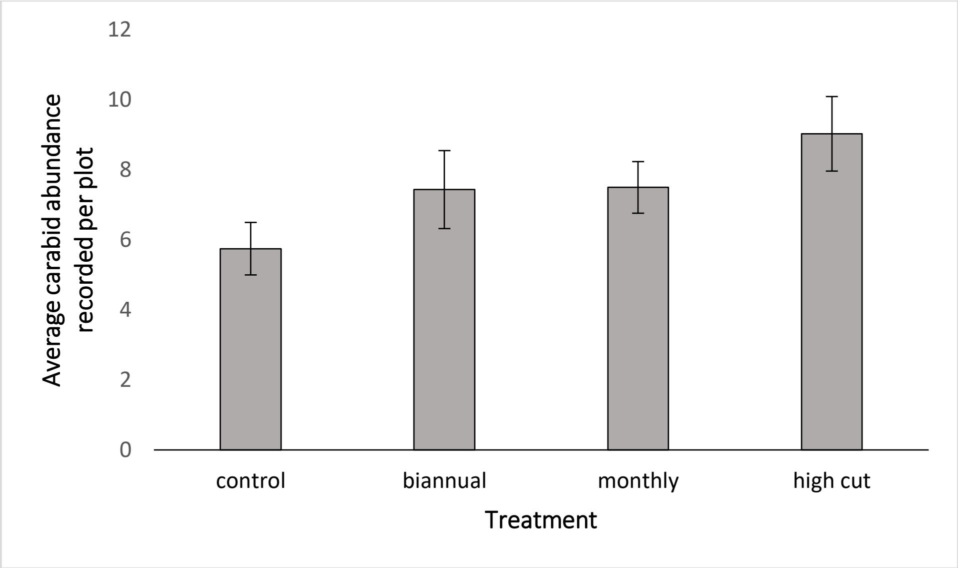


Figure 5.6: *A bar plot showing the difference per treatment of the activity-density of carabids recorded per plot. Activity-density of carabids was greater in the 'Monthly' and 'High cut' treatments when compared to the 'Control' (Wald* $\chi^2 = 9.85$*, p = 0.019). Error bars denote the standard error of the mean.*

5.6. Discussion

Epigeal predators responded positively to reduced intensity of mowing across all three treatments, with higher numbers recorded in all interventions compared to the Control. The differences appear to be largely driven by the carabid beetles. The predator community appears to respond equally well to seasonal mowing as to mowing at a regular frequency but at an increased height, with little difference between the two treatments. Of the three interventions, monthly mowing (i.e. reduced frequency) at the standard height is associated with the lowest increase in predators. The results suggest that, for carabids at least, the detrimental effects of intensive mowing could be mitigated by raising the height of the blades slightly.

The majority of carabid species are nocturnally active, indeed, over 80% of the observed assemblage in this study are nocturnally active species; with *H. affinis, H. rufipes, N. brevicollis and P. madidus* comprising the species most frequently caught. Such species may be less vulnerable to injury from day-time management activities as they seek shelter during the day (Hedde *et al.,* 2015). Despite this, the lower counts observed in the Control plots indicate that frequent, tight mowing of the sward (i.e. <9 cm on average) is not favourable to carabids, in agreement with the findings of Funayama (2014). Mowing frequently and to a low height is likely to cause direct injury or death as well as reduce sheltering opportunities and potentially increase the risk of predation. However, the higher activity-density observed in the High cut treatment would seem to suggest that a sward height increased by just 5cm more than the standard height is enough to prevent loss of predators. This may indicate that carabids are benefitting from the knock-on effects of mowing. Sonoda *et al.* (2013) showed activity-density increased in the days after mowing, attributing the boost to increased food availability for beetles scavenging on injured or dead prey species. Mowing to a low height, but less frequently, as was the case in the Monthly treatment, also appears to be more favourable to carabids than the standard regime.

The low-cut mowing, though undoubtedly detrimental to carabid communities, may be sufficiently infrequent to prevent a total loss of individuals while providing increased feeding opportunities and potentially attracting foraging beetles from the periphery of the orchards.

The orchard selected for study was relatively young, being in its second year when this experiment began in 2013. As the ground cover in the orchard is spontaneous, self-seeded vegetation, much of the variation in composition can be attributed to natural variation in the seed bank and the subsequent competitive interactions between establishing species. While mowing intensity has been altered for the purposes of this study, nonetheless the entire sward has been mown to some extent in the three years since planting; uniformly in the year prior to this study, and then at varying levels over the course of this study. As such, the entire sward will have been subjected to damage caused by mowing. Grasses tend to be more tolerant of this type of damage, having adapted to grazing and other management (Strömberg, 2011). Therefore, the shift in composition from a grass-forb mix towards a grass dominated sward may be reflective of a sward establishing in the face of regular management, whereby grasses come to out-compete forbs with lower tolerance of mowing. The plant community appears to be approaching an equilibrium where forb cover is universally low, and robust grasses dominate.

Although the variation in sward composition was marginal in year two, with forb coverage greatly reduced across the orchard, the ‘High cut’ treatment maintained a greater proportion of forbs, second only to the seasonally cut treatment. In addition, the proportion of bare ground in the High cut treatment was lower than both the Control and Monthly cut treatments, while vegetation height was maintained at approximately 20 cm on average. These results suggest that increasing the height of mower blades could enable a more structurally diverse sward to be maintained.

Several authors have highlighted the benefits of greater forb coverage and variable sward structure to beneficial invertebrates in orchards (Gómez *et al.,* 2017; Markó *et al.*, 2013; Wan *et al.,* 2014). However, the community of arboreal predators was unaffected by changes in ground cover management in this study. It may be that the arboreal communities are simply unaffected by sward height. Those studies which have looked at predators in trees in relation to sward height have generally been using floristically diverse swards, as well as increased vegetation height. Flowers are known to benefit certain predatory arthropods (Campbell *et al.* 2017) through the provision of additional food resources. Forb coverage was generally low in the study orchard, and as such the lower disturbance and increased structural complexity that appears to have benefitted the epigeal community, was presumably of little benefit to the arboreal communities present.

As discussed, the study orchard was relatively young, and both the plant and invertebrate communities were likely to be still establishing at the time of study. It is considered that the experimental design was robust, following standard orchard canopy and ground invertebrate sampling methodologies. The treatments were applied as indicated. Predator abundance was higher in year two, which may indicate an influx of predators to the orchard in response to food availability, potentially in the form of pest species or be due to abiotic factors not recorded. East Malling is a long running farm and research station and, as such the entire area will have been subject to high levels of chemical and physical management for many years. It is possible that the low numbers of arboreal invertebrates observed may reflect low source populations in the wider farm as a whole. Observation of the orchard when both the trees and ground cover are more established would be likely to provide clearer results on the effect of altered mowing on arboreal predators.

5.7. Recommendations

The results presented here suggest that a small change in mowing regime could potentially boost epigeal predatory communities in commercial apple orchards, in particular the activity-density of carabid beetles. As Mathews *et al*. (2004) and Renkema *et al.* (2012) have shown, carabids will predate certain pest species (e.g. *E. lanigerum*, *D. mali* and *C. pomonella*) which spend a portion of their life cycles on the ground (Boreau de Roincé *et* *al.* 2012, Cross *et al.* 2015; Unruh *et al.* 2016), and as such are potentially useful orchard predators. Indeed, the findings set out in Chapter 3 of this work show polyphagous carabids will predate pest species in the orchard.

Other orchard predators are known to respond positively to reduced mowing (Markó *et al*., 2012) and failure to detect such an effect here is considered more likely to be indicative of low source populations, potentially due to orchard age. Further studies in more established dessert orchard plantations could address this issue.

This study demonstrates that raising the height of the cutting blades to approximately 15 cm will support more carabids than a lower cut sward. This is likely to be due to lower instances of death or injury of sheltering ground beetles and regular boosts in food resources to sustain the population. It may be that combining reduced frequency mowing (i.e. monthly) with raised cutting blades offers similar, or perhaps greater benefits, while also reducing management costs for the grower. Further study is recommended to establish an optimal combination of height and frequency to maximise natural enemies in orchards. Nonetheless, it is considered that this technique would be a useful tool in the growers' pest control arsenal, leading to better natural pest control service provision and potentially reduced management costs.

Chapter 6 Discussion

6.1. Discussion

This study set out to investigate the influences of farm management and land use diversity on communities of generalist predators in commercial apple orchards. The findings are reviewed here. Limitations of the study are also considered together with recommendations for growers, and suggestions for further study.

6.2. Land use diversity, orchard predators and predation

6.2.1. Carabids

Overall, the abundance and diversity of carabid communities recorded in this study were not influenced by adjacent land use type. The exception to this being study plots adjacent to unmanaged (i.e. semi-natural/natural) land uses. In these plots, the proportion of species with agricultural associations is greater in the in the assemblage in the interior transects compared to those at the edge. This variation suggests that the assemblage found at the edge of these unmanaged plots comprise more of an 'ecotone' community, experiencing a degree of spill-over to or from the adjacent land use and the orchard. The assemblage found in the interior (i.e. ≥ 50 m from the adjacent land use) of these plots was more similar to those recorded in the other study plot types, (orchard or agricultural), indicating that this is the prevailing orchard community composition, more similar to those found in agricultural land uses than less intensively managed land uses.

The ecological characteristics of carabid communities, as expressed by weighted means of different traits, were influenced by the diversity of land use at the landscape scale, i.e. an area of 1 km radius surrounding the study orchard. Lower land use diversity was associated with smaller beetles and more spring breeding species, both traits shown to increase with increased disturbance (Hill *et al*., 2017) as landscapes of low land use diversity in most published studies are those associated with intensive agricultural systems. High land use diversity was associated with more polyphagous species in this study. This was unexpected, as a higher representation of generalists in beetle assemblages is generally thought to be associated with lower land use diversity (Cizek *et al*., 2012; Gobbi *et al*., 2015). The mechanisms driving this result are unclear, though it may be reflective of the disturbance found within orchard environments. The findings with regards to CWMs are valuable in terms of carabid research, as they support the theory that CWMs are a more useful method of evaluating carabid communities (Gagic *et al*., 2015) than examination of diversity or species composition alone. Based on traditional measures of activity-density or diversity, the variance in communities were not detectible in this work. Indeed, even the more descriptive functional diversity measures (Response Traits Functional Diversity and Effects Traits Functional Diversity) would not have provided the same insights. The lack of variance in functional diversity according to land use in this study is considered to be indicative of a degree of functional redundancy in the assemblage (Oliver *et al*., 2015), possibly indicating communities are relatively stable. Alternatively, it may be that species niches are differentiated according to traits not used in this study (e.g. preferences for abiotic conditions, Pakeman and Stockan, 2014).

The assemblages observed in this study share similarities with those found in found in studies in other agricultural systems, being dominated by disturbance tolerant species, i.e. early breeding species, smaller in size with flexible diets (Letardi *et al*., 2015). However, the lack of association with adjacent habitats and apparent functional redundancy suggests the communities are relatively stable. It is thought that this is indicative of the longevity of the orchard environment. This semi-permanency may support a relatively stable arthropod community, as crops are usually in place for 15 – 20 years, potentially longer. Further, orchard plantations are usually rotated within the same farm, such that when one plantation is grubbed up, another neighbouring plot is likely to still be in place to accommodate displaced arthropods. Furthermore, while the assemblage is reflective of managed land, the species present, specifically the polyphages, have been shown to be effective predators of pest species in this study. Carabids have been somewhat overlooked in orchard IPM, but the findings presented here demonstrate the potential benefit of this largely predatory group.

6.2.2. Earwigs

There was no significant relationship between earwig populations and land use diversity at the scale of 1 km, though a trend towards greater catches in more diverse land use settings was noted. It was hypothesised that earwigs might benefit from a greater proportion of natural/semi-natural habitats in a landscape. However the limited dispersal range of the species is likely to limit the range over which any such effect is detectable (Stutz and Entling, 2011). An interaction between adjacent land use and the distance into the orchard, i.e. transect location, was identified. Greater numbers were recorded in edge transects of plots adjacent to natural/semi-natural land uses, while plots next to agricultural land supported fewer earwigs. This would seem to support the hypothesis that higher earwig abundance is associated with less intensively managed land uses, but this effect may only be detectable within 50 m or less, in line with the typical dispersal range of 20 m – 30 m. Nonetheless it is considered that some of the variability in earwig populations in apple orchards discussed by other authors (e.g. Cross *et al.*, 2015; Dib, Sauphanor and Capowiez, 2016) could be attributed to land use diversity.

The arboreal predation levels recorded in this study were found to be influenced by the diversity of land use at a 1 km scale. In line with other research (Bianchi *et al.*, 2006; Chaplin-Kramer *et al.*, 2011; Rusch *et al.*, 2016) more diverse landscapes were associated with higher levels of arboreal predation. The majority of studies on the effect of surrounding land use on the abundance of natural enemies in agricultural systems have focussed on arable crops, so the findings presented here add to the existing knowledge, demonstrating the role of land use heterogeneity in natural enemy mediated pest control in apple orchards.

Earwig abundance was strongly related to increased aphid predation in this study. This cannot be confidently attributed solely to earwigs as other predators were not excluded in this study. Nonetheless, though not significantly influenced by land use diversity at the 1 km scale, mean earwig catch varied according to land use settings, with more earwigs recorded in more diverse settings. This may be the cause of higher predation levels observed in study plots in more diverse land use settings. The relationship between aphid depletion and earwig numbers supports the findings of previous studies regarding this predator and adds to the evidence demonstrating earwigs to be effective pest control agents in orchards. There was also a trend, though not significant, of greater predation in the edge transects adjacent to unmanaged or orchard habitats, which reflects the variance in abundance of earwigs in relation to adjacent land use in the study orchards. Indeed, the interaction between adjacent habitat and earwig population size was found to influence orchard predation. The findings indicate that earwigs are important orchard predators and that earwig mediated pest control is likely to be influenced to an extent by land use.

6.3. Orchard management and predator communities

6.3.1. Insecticide usage

In this study the use of pesticides was found to be related to the diversity of land use at the landscape scale; with greater diversity being associated with lower pesticide use. This agrees with the literature indicating reduced landscape heterogeneity is associated with reduced natural enemy communities, and in turn poorer natural pest control and consequently an increased reliance on chemicals within farms (Rusch *et al.*, 2016). Adjacent land use was not related to pesticide use in the study orchards which is considered to reflect the tendency of growers to spray larger areas than may be necessary to ensure that pest outbreaks are controlled.

Insecticide use was associated with a reduction in earwig numbers in both survey years. In 2014 the use of flonicamid had the greatest effect on earwig numbers, while in 2015 thiacloprid had the greatest effect on abundance, reflecting the levels of use of each chemical in each year. A moratorium was imposed on neonicotinoid use across the European Union in December 2013, and in 2018 this was raised to a ban (Stokstad, 2018) owing to the well documented negative impacts of this class of insecticides on beneficial arthropods (Pisa *et al.*, 2014). Flonicamid is associated with lower numbers and decreased foraging activity in earwigs (Fountain and Harris, 2015; Vogt *et al.*, 2010). A number of researchers have also demonstrated negative effects of thiacloprid on earwigs in the field and laboratory. The results presented here lend support to those findings, demonstrating that the use of either can reduce earwig abundance in apple orchards. Flonicamid is already included in the neonicotinoid ban, though the use of thiacloprid is still under review (European Commission, 2018). However, the consistent negative effects on beneficial species observed in this and other work would suggest thiacloprid should be included in the neonicotinoid ban.

It is notable that chlorpyrifos use was not associated with reduced earwig abundance in this study, though it has been shown to have detrimental effects on earwigs in laboratory studies and other fruit crops. It may be the case that earwigs in the orchard are not exposed to chlorpyrifos for a sufficient duration to cause notable reductions in abundance. Further, since earwigs must remain active for a long enough period to allow them to return to the refugia to be counted if they have been exposed, it would appear that sub-lethal effects, if occurring, are not leading to reductions in abundance in the orchards. It may be that the presence of refugia provides sufficient shelter to enable populations to avoid the worst effects of this chemical. Use of chlorpyrifos has been banned in orchards in the UK, so further study will not be possible or particularly instructive with regards to chlorpyrifos. However, it is considered that further research into the potential of refugia to mitigate the negative effects of chemical use would be beneficial.

Evidence was also found of an effect of insecticide use on carabids. Thiacloprid was associated with reduced size and greater functional diversity of response traits. Thiacloprid is a neonicotinoid compound which has not yet been banned for use in the EU, as its toxicity is still under investigation (European Commission, 2018). Though negative effects have been noted in other studies, and indeed on earwigs in this study, the findings presented here show an effect on carabid communities, but not necessarily a detrimental one. There was no indication that use of thiacloprid influenced abundance so although the use of this compound is effectively selecting for smaller species, the overall resilience of the community does not appear to be affected.

Chlorpyrifos was associated with lower response and effects traits functional diversity. Activity-density and the abundance of polyphagous species were also found to increase with chlorpyrifos use. These results are considered likely to reflect a greater level of activity in response to food availability rather than an actual boost in numbers. This effect has been documented elsewhere where food shortages after insecticide application trigger higher levels of foraging to compensate (Wan *et al*., 2014). The nocturnal habit of the majority of carabids observed in this study may reduce exposure to chemicals, in turn reducing detrimental impacts from insecticide usage. Further, assuming that the assemblage is relatively stable in orchards, there is an implied level of tolerance of this type of disturbance in those species which are present.

6.3.2. Mowing height and frequency

All of the orchards used in the main study were subject to frequent, close mowing, as is the norm in commercial dessert apple orchards, resulting in an average sward height of approximately 9 cm. The results from the mowing manipulation study showed that epigeal predator communities responded positively to less intensive management of the vegetation in the orchard alleys. The increases were largely driven by increases in the activity-density of the carabid beetles.

Specifically, raising cutting blades to 15 cm was associated with increases in epigeal predators. Raising the blades like this likely prevents death or injury to carabids by maintaining sward structure. Further, high mowing may kill other arthropods dwelling higher in the sward, causing a boost in food availability for carabids at ground level. There was no indication of an influence of this altered management on arboreal communities.

The results of the mowing intervention highlight the benefit of less intensive management of the orchard floor in supporting epigeal predatory communities. As orchard carabid communities were demonstrated to be effective predators of pests in the wider study, this result is an important finding for IPM in orchards.

6.4. Earwig monitoring

Monitoring earwig populations in orchards could be a useful tool for growers to aid well informed management decision making. Three different earwig sampling techniques were trialled in this study; diurnal tap sampling, nocturnal tap sampling and monitoring of artificial refugia. Although there is a correlation between estimates from diurnal tap sampling and refugia, results suggested that regular monitoring of artificial refugia provided the most useful data. Further, beyond initial set-up, this method is considered to be the most efficient technique in terms of time. Earwigs are not commonly found in trees during the day, when tree tapping is usually undertaken, leading to underestimations of populations, However, they will naturally congregate in refugia where they can easily be monitored. Refugia also provide sheltering opportunities for earwigs in orchards, offering protection from farm management and meeting the earwigs' tendency to aggregate.

6.5. Limitations and recommendations

Greater spatial and temporal replication would undoubtedly clarify the relationships between land use and orchard predators. Though the orchards studied in this work are considered to be representative of the majority of Kentish dessert apple plantations, the annual variations in predator numbers would suggest an influence of a broader range of factors, including weather or other abiotic factors. Increased replication could allow such factors to be taken account of.

It is evident that earwigs and carabids will actively predate aphids when present. However, further research is needed to quantify the levels of pest control that could be achieved in the field. This is particularly true for carabids as they are less likely to encounter apple pests than arboreal predators. In this study, spray records were used as a proxy for pest assessments. However, detailed pest monitoring could be employed alongside predator surveys to develop a greater understanding of the interactions between land use, key predators and pests. Likewise, studying natural enemy dynamics in the absence of chemical pest management would doubtless increase understanding.

Malone *et al.* (2017) looked at the effects of the whole suite of chemicals applied in orchards and found compounds used for fungal diseases and management of understory vegetation to be related to natural enemy populations. Fungicides will remove food resources for a range of smaller, fungivorous invertebrates such as Collembola, which will reduce food in the system for predatory species. There is also some evidence that herbicides can negatively affect arthropod predators, with exposure leading to altered locomotion and reductions in long term survivorship (Evans *et al*., 2010). Given the level of spraying activity found in orchards, additional analysis is recommended to fully understand these effects on predators in an orchard environment.

Using a modified TFI, i.e. a metric not based on *a priori* assumptions of toxicity, appears to have provided meaningful results, with chemicals found to be detrimental in other studies, also remaining as terms in the most parsimonious models in this study. The results of this study confirm that harmful effects of flonicamid and thiacloprid which have been observed in laboratory studies are evident in the field. While the need for some chemical pest control is unavoidable, careful timing of applications that avoid vulnerable life stages, will reduce impacts on beneficial species. It is interesting to note that the effects of chlorpyrifos on both carabids and earwigs in this study are positive or neutral. The use of chlorpyrifos has been restricted in the UK since 2016 (HSE, 2018) and is frequently associated with negative effects on beneficial invertebrates. While the boost in carabid activity-density is thought to reflect increased foraging activity in the wake of food shortages, rather than a real increase in numbers, there was no evidence of a decline through time with applications of chlorpyrifos. In fact, based solely on the evidence presented here, the use of chlorpyrifos appears somewhat less detrimental to these predatory species than some other compounds (i.e. neonicotinoids) currently available. Chlorpyrifos is banned currently, as are most neonicotinoids, nonetheless more work is needed to understand the effects of permissible pesticides on orchard predators under field conditions as there is some discrepancy between laboratory and field effects. Further, whether the inclusion of refugia in orchards confers a degree of protection from spray activities for earwigs would also be worth exploring in more detail as they are readily used by earwigs in orchards and provide an easy to implement and cheap enhancement for the species.

Refugia have been demonstrated to provide both an efficient method of measuring populations, as well as a habitat resource for orchard earwigs. Growers could deploy simple earwig refugia in orchards to enhance opportunities for sheltering earwigs, exploiting their natural aggregative tendencies and potentially reducing exposure to pesticides.

It is clear from this study that orchard earwigs are positively influenced by the presence of adjacent semi-natural/natural habitats, but that this effect diminishes with distance from that non-crop habitat. This knowledge can be exploited by growers to enhance natural pest control services in orchards. Based on the findings presented here, the inclusion of unmanaged areas of woody or scrub habitat within the orchard plantation will offer enhanced opportunities for earwigs, and boost pest control in apple canopies. Enhancing these features with tussocky grasses will add benefit for orchard carabid communities.

The work presented here also demonstrates that raising the height of the cutting blades of a mower to approximately 15 cm will support more epigeal predators, particularly carabids than a lower cut sward. This is likely to be due to lower instances of death or injury of sheltering ground beetles and regular boosts in food resources to sustain the population. It may be that combining reduced frequency mowing (i.e. monthly) with raised cutting blades offers similar, or perhaps greater benefits, while also reducing management costs for the grower. Further study is recommended to establish an optimal combination of height and frequency to maximise natural enemies in orchards. It is also recommended to repeat the study in a more established orchard plantation to reduce the effects of changes in vegetation structure and composition in the early years after establishment. Nonetheless, based on these findings it is considered that this alternative mowing regime would boost numbers of epigeal predators which may assist in orchard IPM, without increasing costs associated with set-up or management.

Through the provision of additional habitat resources for carabids and earwigs, and by adopting a more sensitive mowing regime it is thought that growers will be able to increase the levels of natural pest control in orchards. Though chemical pest control is unavoidable, these small enhancements should boost orchard predator populations by offering refuge and other resources, and ultimately help move towards a lower chemical dependence. Further, owing to the semi-permanent nature of the orchard habitat, these provisions should allow a more stable community to establish and persist over a number of years, whilst also adding to the cumulative diversity of land use at a broader scale.

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Appendices

Appendix 1: Carabid functional and behavioural traits per species recorded in this study.

Species / Traits	**Food of Adults** - *(1) Collembola specialists, (2) generalist predators, (3) mixed diet, (4) mostly plant*	**Activity** - *(D) diurnal, (N) nocturnal)*	**Breeding Season** - *(A) Autumn, (S) Spring*	**Size** - *(1) ≤5mm, (2) 5 – 9mm, (3) 9 – 15mm, (4) ≥15mm*	**Wings** - *(1) apterous, (2) dimorphic, (3) macropterous)*	**Head Width** (mm)	**Eye Width** (mm)	**Antennae Length** (mm)	**Pronotum Width** (mm)	**Pronotum Height** (mm)	**Femora Length** (mm)	**Metatibia Length** (mm)	**Metatarsal Length** (mm)	**Femora Width** (mm)	**Mass** (mg)
Abax parallelepipedus	2	N	A	4	1	3.81	0.48	8.33	6.19	2.91	3.47	3.31	2.81	0.91	72.49
Acupalpus meridianus	2	N	S	1	3	0.69	0.09	0.70	0.86	0.66	0.75	0.51	0.56	0.16	0.72
Amara	4	D	S	2	3	1.38	0.25	2.50	2.74	1.84	1.92	1.75	1.56	0.54	7.52
Anchomenus dorsalis	2	N	S	2	3	1.01	0.30	4.32	1.53	1.46	2.30	2.06	2.02	0.40	4.45
Anisodactylus binotatus	4	-	S	3	3	*no data available*									
Asaphidion flavipes	1	D	S	1	3	0.77	0.39	2.02	1.08	1.05	1.35	1.16	1.09	0.27	1.53
Bembidion gilvipes	2	N	S	1	2	0.40	0.13	1.24	0.65	0.49	0.73	0.49	0.55	0.15	0.45
Bembidion lampros	2	D	S	1	2	0.69	0.26	1.76	1.02	0.84	1.05	0.90	0.80	0.23	0.88
Bembidion lunulatum	2	D	S	1	3	0.76	0.19	1.73	1.13	0.95	1.08	1.00	0.80	0.24	0.94
Bembidion obtusum	2	DN	A	1	2	0.60	0.14	1.44	0.90	0.66	0.83	0.67	0.61	0.19	0.64
Bembidion properans	2	D	S	1	2	0.69	0.26	1.76	1.02	0.84	1.05	0.90	0.80	0.23	0.98
Calathus ambiguus	2	N	A	3	3	1.58	0.34	4.81	3.01	2.42	2.99	3.04	2.85	0.70	13.17
Calathus erratus	2	N	A	3	2	no data available									
Calathus fuscipes	2	N	A	3	1	1.79	0.38	5.39	3.39	2.76	3.45	3.40	3.31	0.81	19.09
Calathus melanocephalus	2	N	A	2	2	1.18	0.24	3.87	2.34	1.75	2.17	2.10	2.09	0.49	5.58
Carabus violaceus	2	N	A	4	1	3.46	0.70	12.10	6.55	6.06	8.08	7.76	6.84	1.38	152.78
Clivina fossor	2	DN	S	2	2	0.86	0.16	1.80	1.57	1.32	1.19	0.97	0.81	0.31	3.17
Curtonotus aulicus	4	N	A	3	3	2.63	0.42	4.91	4.11	3.09	3.40	3.16	2.95	0.97	23.44

Appendix 1: Carabid functional and behavioural traits per species recorded in this study – continued.

Species / Traits	**Food of Adults** - *(1) Collembola specialists, (2) generalist predators, (3) mixed diet, (4) mostly plant*	**Activity** - *(D) diurnal, (N) nocturnal)*	**Breeding Season** - *(A) Autumn, (S) Spring*	**Size** - *(1) ≤5mm, (2) 5 – 9mm, (3) 9 – 15mm, (4) ≥15mm*	**Wings** - *(1) apterous, (2) dimorphic, (3) macropterous)*	**Head Width** (mm)	**Eye Width** (mm)	**Antennae Length** (mm)	**Pronotum Width** (mm)	**Pronotum Height** (mm)	**Femora Length** (mm)	**Metatibia Length** (mm)	**Metatarsal Length** (mm)	**Femora Width** (mm)	**Mass** (mg)
Harpalus affinis	4	N	S	2	3	1.85	0.30	3.66	3.13	2.36	2.51	2.39	1.87	0.78	13.05
Harpalus latus	4	N	S	2	3	*no data available*									
Harpalus rufipes	3	N	A	3	3	2.68	0.48	5.45	4.30	3.36	3.85	3.71	3.09	1.04	26.07
Laemostenus terricola	-	N	A	4	1	*no data available*									
Leistus spinibarbus	1	N	A	3	2	1.21	0.39	4.04	2.26	1.53	2.77	2.51	1.79	0.35	9.01
Loricera pilicornis	1	DN	S	2	3	1.02	0.32	3.97	2.06	1.59	2.47	2.15	1.98	0.44	5.11
Nebria brevicollis	2	N	A	3	3	2.23	0.48	6.50	3.55	2.77	3.92	3.46	3.42	0.62	19.13
Notiophilus biguttatus	1	D	S	1	2	1.37	0.50	1.64	1.58	1.18	1.49	1.23	1.24	0.31	2.20
Ophonus ardosiacus	4	N	-	2	3	*no data available*									
Poecilus cupreus	2	D	S	3	3	1.75	0.39	4.93	3.63	2.70	3.15	3.02	3.17	0.85	15.62
Pterostichus cristatus	2	N	A	3	1	*no data available*									
Pterostichus madidus	3	N	A	3	1	2.75	0.47	6.34	4.40	3.50	4.09	4.07	3.41	1.10	42.35
Pterostichus melanarius	2	DN	A	3	2	2.89	0.48	6.36	4.67	3.60	4.23	4.01	3.40	1.12	36.97
Pterostichus vernalis	2	N	S	2	2	1.04	0.24	3.19	2.06	1.57	1.85	1.76	1.55	0.45	4.37
Stomis pumicatus	2	N	S	2	1	1.09	0.23	3.75	1.74	1.48	1.84	1.60	1.39	0.45	6.00
Trechus quadristriatus	1	N	A	1	3	0.67	0.13	2.05	1.09	0.81	1.17	0.97	0.77	0.30	1.03
Trechus rivularis	2	-	A	1	2	*no data available*									
Trechus secalis	1	N	A	1	1	0.70	0.09	2.14	1.07	0.85	1.15	0.91	0.77	0.28	0.97

Appendix 2: Treatment Frequency Index values.

The TFI is a method of establishing an annual measure of chemical use. In this study the TFI was calculated to provide an overall value for all of the insecticides and acaricides applied per plot/year, as well as an overall value of the fungicides applied per plot/year. In addition, the TFI was also calculated for each individual chemical applied per plot/year. The results are provided in Tables 1 – 6 below.

Table 1 Insecticides and Acaricides 2013

Plot - (A) agricultural, (O) orchard, (S) unmanaged *TFI = total number of pesticide treatments, weighted by the ratio of the dose used to the recommended dose (Kg or L per hectare)*						Insecticides, Acaricides									
Farm	Plot	Year	TFI All Pesticides	TFI Insecticides, Acaricides	TFI Fungicides	*chlorantraniliprole (L)*	*chlorpyrifos (L)*	*fenoxycarb (Kg)*	*flonicamid (Kg)*	*indoxacarb(Kg)*	*pirimicarb (Kg)*	*spirodiclofen (L)*	*tebufenpyrad (Kg)*	*methoxyfenozide (L)*	*thiacloprid (L)*
Batteries	A	2013	33.93	4.57	29.36	0.18	4.00	0.00	0.14	0.00	0.00	0.00	0.00	0.00	0.25
Batteries	O	2013	32.43	3.57	28.86	0.18	3.00	0.00	0.14	0.00	0.00	0.00	0.00	0.00	0.25
Batteries	S	2013	34.93	5.57	29.36	0.18	5.00	0.00	0.14	0.00	0.00	0.00	0.00	0.00	0.25
Densted	A	2013	39.37	7.57	31.80	0.18	7.00	0.00	0.14	0.00	0.00	0.00	0.00	0.00	0.25
Densted	O	2013	35.91	5.57	30.34	0.18	5.00	0.00	0.14	0.00	0.00	0.00	0.00	0.00	0.25
Densted	S	2013	38.14	5.57	32.57	0.18	5.00	0.00	0.14	0.00	0.00	0.00	0.00	0.00	0.25
Griffins	A	2013	25.68	1.65	24.03	0.00	1.00	0.00	0.28	0.00	0.00	0.00	0.00	0.00	0.37
Griffins	O	2013	25.68	1.65	24.03	0.00	1.00	0.00	0.28	0.00	0.00	0.00	0.00	0.00	0.37
Griffins	S	2013	25.68	1.65	24.03	0.00	1.00	0.00	0.28	0.00	0.00	0.00	0.00	0.00	0.37
HMounts	A	2013	35.32	3.25	32.07	0.00	3.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.25
HMounts	O	2013	27.63	3.25	24.38	0.00	3.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.25
HMounts	S	2013	35.09	3.25	31.84	0.00	3.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.25
Loddington	A	2013	22.70	1.85	20.85	0.27	0.00	0.00	0.00	0.00	0.60	0.60	0.00	0.00	0.38
Loddington	O	2013	30.15	1.76	28.39	0.18	0.00	0.00	0.00	0.00	0.60	0.00	0.00	0.60	0.38
Loddington	S	2013	26.60	1.93	24.67	0.36	0.00	0.00	0.00	0.00	1.19	0.00	0.00	0.00	0.38
OrchLodg	A	2013	19.49	2.69	16.80	0.07	2.00	0.00	0.62	0.00	0.00	0.00	0.00	0.00	0.00
OrchLodg	O	2013	13.09	1.33	11.76	0.07	1.20	0.00	0.06	0.00	0.00	0.00	0.00	0.00	0.00
OrchLodg	S	2013	14.88	2.13	12.75	0.07	2.00	0.00	0.06	0.00	0.00	0.00	0.00	0.00	0.00
Santon	A	2013	38.70	3.50	35.20	0.00	3.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.50
Santon	O	2013	38.70	3.50	35.20	0.00	3.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.50
Santon	S	2013	35.70	1.50	34.20	0.00	1.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.50
WPF	A	2013	32.20	1.89	30.31	0.35	0.00	0.00	0.00	0.00	0.56	0.00	0.00	0.60	0.38
WPF	O	2013	32.20	1.89	30.31	0.35	0.00	0.00	0.00	0.00	0.56	0.00	0.00	0.60	0.38
WPF	S	2013	32.75	1.17	31.58	0.18	0.00	0.00	0.00	0.00	0.02	0.00	0.00	0.60	0.38

Table 2 Insecticides and Acaricides 2014

Plot - (A) agricultural, (O) orchard, (S) unmanaged *TFI = total number of pesticide treatments, weighted by the ratio of the dose used to the recommended dose (Kg or L per hectare)*						Insecticides, Acaricides									
Farm	Plot	Year	TFI All Pesticides	TFI Insecticides, Acaricides	TFI Fungicides	*chlorantraniliprole (L)*	*chlorpyrifos (L)*	*fenoxycarb (Kg)*	*flonicamid (Kg)*	*indoxacarb(Kg)*	*pirimicarb (Kg)*	*spirodiclofen (L)*	*tebufenpyrad (Kg)*	*methoxyfenozide (L)*	*thiacloprid (L)*
Batteries	A	2014	19.76	3.29	16.47	0.00	3.00	0.15	0.14	0.00	0.00	0.00	0.00	0.00	0.00
Batteries	O	2014	18.32	3.35	14.97	0.00	3.00	0.15	0.00	0.00	0.00	0.00	0.00	0.00	0.20
Batteries	S	2014	19.82	3.35	16.47	0.00	3.00	0.15	0.00	0.00	0.00	0.00	0.00	0.00	0.20
Densted	A	2014	23.07	5.35	17.72	0.00	5.00	0.15	0.00	0.00	0.00	0.00	0.00	0.00	0.20
Densted	O	2014	23.02	4.35	18.67	0.00	4.00	0.15	0.00	0.00	0.00	0.00	0.00	0.00	0.20
Densted	S	2014	23.07	5.35	17.72	0.00	5.00	0.15	0.00	0.00	0.00	0.00	0.00	0.00	0.20
Griffins	A	2014	44.34	3.73	40.61	0.35	2.50	0.00	0.28	0.00	0.00	0.00	0.00	0.60	0.00
Griffins	O	2014	44.34	3.73	40.61	0.35	2.50	0.00	0.28	0.00	0.00	0.00	0.00	0.60	0.00
Griffins	S	2014	44.34	3.73	40.61	0.35	2.50	0.00	0.28	0.00	0.00	0.00	0.00	0.60	0.00
HMounts	A	2014	23.94	3.35	20.59	0.00	3.00	0.15	0.00	0.00	0.00	0.00	0.00	0.00	0.20
HMounts	O	2014	23.43	2.45	20.98	0.00	2.00	0.25	0.00	0.00	0.00	0.00	0.00	0.00	0.20
HMounts	S	2014	23.84	2.35	21.49	0.00	2.00	0.15	0.00	0.00	0.00	0.00	0.00	0.00	0.20
Loddington	A	2014	34.25	1.09	33.16	0.35	0.00	0.00	0.14	0.00	0.00	0.00	0.00	0.60	0.00
Loddington	O	2014	38.78	0.99	37.79	0.35	0.00	0.00	0.14	0.00	0.00	0.00	0.50	0.00	0.00
Loddington	S	2014	43.19	2.40	40.79	0.35	0.00	0.00	0.14	0.25	0.56	0.00	0.50	0.60	0.00
OrchLodg	A	2014	26.54	2.11	24.43	0.14	1.66	0.00	0.06	0.00	0.00	0.25	0.00	0.00	0.00
OrchLodg	O	2014	26.54	2.11	24.43	0.14	1.66	0.00	0.06	0.00	0.00	0.25	0.00	0.00	0.00
OrchLodg	S	2014	26.54	2.11	24.43	0.14	1.66	0.00	0.06	0.00	0.00	0.25	0.00	0.00	0.00
Santon	A	2014	19.62	2.66	16.96	0.17	2.00	0.15	0.14	0.00	0.00	0.00	0.00	0.00	0.20
Santon	O	2014	17.92	2.66	15.26	0.17	2.00	0.15	0.14	0.00	0.00	0.00	0.00	0.00	0.20
Santon	S	2014	19.62	2.66	16.96	0.17	2.00	0.15	0.14	0.00	0.00	0.00	0.00	0.00	0.20
WPF	A	2014	31.90	2.59	29.31	0.35	1.00	0.00	0.14	0.00	0.00	0.00	0.50	0.60	0.00
WPF	O	2014	31.90	2.59	29.31	0.35	1.00	0.00	0.14	0.00	0.00	0.00	0.50	0.60	0.00
WPF	S	2014	35.76	2.59	33.16	0.35	1.00	0.00	0.14	0.00	0.00	0.00	0.50	0.60	0.00

Table 3 Insecticides and Acaricides 2015

Plot - (A) agricultural, (O) orchard, (S) unmanaged *TFI = total number of pesticide treatments, weighted by the ratio of the dose used to the recommended dose (Kg or L per hectare)*						Insecticides, Acaricides									
Farm	Plot	Year	TFI All Pesticides	TFI Insecticides, Acaricides	TFI Fungicides	*chlorantraniliprole (L)*	*chlorpyrifos (L)*	*fenoxycarb (Kg)*	*flonicamid (Kg)*	*indoxacarb(Kg)*	*pirimicarb (Kg)*	*spirodiclofen (L)*	*tebufenpyrad (Kg)*	*methoxyfenozide (L)*	*thiacloprid (L)*
Batteries	A	2015	18.15	1.00	17.15	0.00	1.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Batteries	O	2015	18.15	1.00	17.15	0.00	1.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Batteries	S	2015	20.15	1.00	19.15	0.00	1.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Densted	A	2015	10.10	0.20	9.90	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.20
Densted	O	2015	12.40	0.00	12.40	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Densted	S	2015	9.90	0.00	9.90	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Griffins	A	2015	35.18	1.47	33.71	0.35	0.00	0.00	0.14	0.00	0.00	0.00	0.00	0.60	0.38
Griffins	O	2015	35.18	1.47	33.71	0.35	0.00	0.00	0.14	0.00	0.00	0.00	0.00	0.60	0.38
Griffins	S	2015	35.18	1.47	33.71	0.35	0.00	0.00	0.14	0.00	0.00	0.00	0.00	0.60	0.38
HMounts	A	2015	12.37	0.53	11.84	0.18	0.00	0.15	0.00	0.00	0.00	0.00	0.00	0.00	0.20
HMounts	O	2015	7.97	0.18	7.79	0.18	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
HMounts	S	2015	12.37	0.53	11.84	0.18	0.00	0.15	0.00	0.00	0.00	0.00	0.00	0.00	0.20
Loddington	A	2015	36.14	2.32	33.83	0.18	0.00	0.00	0.14	0.00	0.00	0.60	0.00	1.05	0.35
Loddington	O	2015	37.35	2.28	35.08	0.18	0.00	0.00	0.14	0.00	0.56	0.00	0.00	1.05	0.35
Loddington	S	2015	31.44	2.31	29.13	0.18	0.00	0.00	0.14	0.00	1.02	0.00	0.00	0.60	0.38
OrchLodg	A	2015	14.21	0.20	14.01	0.14	0.00	0.00	0.06	0.00	0.00	0.00	0.00	0.00	0.00
OrchLodg	O	2015	14.21	0.20	14.01	0.14	0.00	0.00	0.06	0.00	0.00	0.00	0.00	0.00	0.00
OrchLodg	S	2015	14.21	0.20	14.01	0.14	0.00	0.00	0.06	0.00	0.00	0.00	0.00	0.00	0.00
Santon	A	2015	8.63	0.00	8.63	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Santon	O	2015	8.63	0.00	8.63	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Santon	S	2015	8.63	0.00	8.63	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
WPF	A	2015	32.80	2.04	30.77	0.35	0.00	0.00	0.11	0.00	0.00	0.00	0.00	1.20	0.38
WPF	O	2015	32.80	2.04	30.77	0.35	0.00	0.00	0.11	0.00	0.00	0.00	0.00	1.20	0.38
WPF	S	2015	36.64	2.42	34.22	0.18	0.00	0.00	0.11	0.00	0.56	0.00	0.00	1.20	0.38

Table 4 Fungicides 2013

Plot - (A) agricultural, (O) orchard, (S) unmanaged

TFI = total number of pesticide treatments, weighted by the ratio of the dose used to the recommended dose (Kg or L per hectare)

				Fungicides									
Farm	Plot	Year	TFI Fungicides	*boscalid pyraclostrobin (Kg)*	*bupirimate n-butanol hydrocarbons (Kg)*	*captan (Kg)*	*copper (Kg)*	*cyflufenamid (L)*	*cyprodinil fludioxonil (L)*	*difenoconazole (L)*	*dithianon (L)*	*dithianon WG (Kg)*	*dodine (L)*
Batteries	A	2013	29.36	0.00	0.00	2.50	0.00	0.00	0.00	0.00	1.50	0.00	2.50
Batteries	O	2013	28.86	0.00	0.00	2.50	0.00	0.00	0.00	0.00	1.50	0.00	2.50
Batteries	S	2013	29.36	0.00	0.00	2.50	0.00	0.00	0.00	0.00	1.50	0.00	2.50
Densted	A	2013	31.80	0.00	0.00	3.00	0.00	0.00	0.00	0.00	2.25	0.00	2.50
Densted	O	2013	30.34	0.00	0.00	3.00	0.00	0.00	0.00	0.00	2.25	0.00	2.50
Densted	S	2013	32.57	0.00	0.00	3.00	0.00	0.00	0.00	0.00	2.25	0.00	2.50
Griffins	A	2013	24.03	0.00	1.20	1.00	0.00	1.00	0.00	0.20	0.00	1.85	0.00
Griffins	O	2013	24.03	0.00	1.20	1.00	0.00	1.00	0.00	0.20	0.00	1.85	0.00
Griffins	S	2013	24.03	0.00	1.20	1.00	0.00	1.00	0.00	0.20	0.00	1.85	0.00
HMounts	A	2013	32.07	0.00	0.00	3.00	0.00	0.00	0.00	0.00	2.25	0.00	2.50
HMounts	O	2013	24.38	0.00	0.00	2.00	0.00	0.00	0.00	0.00	0.75	0.75	2.50
HMounts	S	2013	31.84	0.00	0.00	3.00	0.00	0.00	0.00	0.00	2.25	0.00	2.50
Loddington	A	2013	20.85	0.81	0.00	2.48	0.00	0.80	0.00	0.00	0.00	0.62	2.57
Loddington	O	2013	28.39	0.81	0.00	3.01	0.00	1.00	0.00	0.00	0.00	1.25	5.01
Loddington	S	2013	24.67	0.81	0.00	3.00	0.00	1.00	0.00	0.00	0.00	1.25	5.01
OrchLodg	A	2013	16.80	1.02	0.00	0.60	0.00	0.20	2.20	5.20	0.00	1.50	0.97
OrchLodg	O	2013	11.76	1.02	0.00	0.60	0.00	0.20	0.23	0.36	0.00	1.50	0.00
OrchLodg	S	2013	12.75	1.02	0.00	0.60	0.00	0.20	0.23	3.60	0.00	1.44	0.00
Santon	A	2013	35.20	0.00	0.00	2.00	0.00	0.00	0.00	0.40	2.25	0.50	2.50
Santon	O	2013	35.20	0.00	0.00	2.00	0.00	0.00	0.00	0.40	2.25	0.50	2.50
Santon	S	2013	34.20	0.00	0.00	2.00	0.00	0.00	0.00	0.40	2.25	0.50	2.50
WPF	A	2013	30.31	0.80	0.00	2.00	0.00	1.00	0.00	0.00	0.00	1.25	7.55
WPF	O	2013	30.31	0.80	0.00	2.00	0.00	1.00	0.00	0.00	0.00	1.25	7.55
WPF	S	2013	31.58	0.80	0.00	2.00	0.00	1.00	0.00	0.00	0.00	1.25	7.56

Table 4 Fungicides 2013 – continued.

Plot - (A) agricultural, (O) orchard, (S) unmanaged *TFI = total number of pesticide treatments, weighted by the ratio of the dose used to the recommended dose (Kg or L per hectare)*				Fungicides										
Farm	Plot	Year	TFI Fungicides	*fenbuconazole (Kg)*	*kresoxim-methyl (Kg)*	*liquid copper (L)*	*meptyldinocap (L)*	*myclobutanil cyclohexanone (L)*	*penconazole (L)*	*potassium hydrogen carbonate (Kg)*	*pyraclostrobin dithianon (Kg)*	*pyrimethanil (L)*	*tebuconazole (L)*	*tebuconazole B (L)*
Batteries	A	2013	29.36	4.00	0.00	0.00	0.00	1.86	17.00	0.00	0.00	0.00	0.00	0.00
Batteries	O	2013	28.86	4.00	0.00	0.00	0.00	1.86	16.50	0.00	0.00	0.00	0.00	0.00
Batteries	S	2013	29.36	4.00	0.00	0.00	0.00	1.86	17.00	0.00	0.00	0.00	0.00	0.00
Densted	A	2013	31.80	5.00	0.00	0.00	0.00	2.55	16.50	0.00	0.00	0.00	0.00	0.00
Densted	O	2013	30.34	4.00	0.00	0.00	0.00	2.09	16.50	0.00	0.00	0.00	0.00	0.00
Densted	S	2013	32.57	4.00	0.00	0.00	0.00	2.32	18.50	0.00	0.00	0.00	0.00	0.00
Griffins	A	2013	24.03	0.00	0.00	0.00	0.00	2.78	16.00	0.00	0.00	0.00	0.00	0.00
Griffins	O	2013	24.03	0.00	0.00	0.00	0.00	2.78	16.00	0.00	0.00	0.00	0.00	0.00
Griffins	S	2013	24.03	0.00	0.00	0.00	0.00	2.78	16.00	0.00	0.00	0.00	0.00	0.00
HMounts	A	2013	32.07	5.00	0.00	0.00	0.00	2.32	17.00	0.00	0.00	0.00	0.00	0.00
HMounts	O	2013	24.38	4.00	0.00	0.00	0.00	1.38	13.00	0.00	0.00	0.00	0.00	0.00
HMounts	S	2013	31.84	5.00	0.00	0.00	0.00	2.09	17.00	0.00	0.00	0.00	0.00	0.00
Loddington	A	2013	20.85	0.00	0.20	0.00	0.60	1.28	10.75	0.00	0.00	0.00	0.00	0.75
Loddington	O	2013	28.39	0.00	0.20	0.00	1.20	1.46	13.01	0.00	0.00	0.00	0.00	1.45
Loddington	S	2013	24.67	0.00	0.20	0.00	1.20	1.46	10.00	0.00	0.00	0.00	0.00	0.75
OrchLodg	A	2013	16.80	0.00	0.09	0.00	0.00	0.62	4.40	0.00	0.00	0.00	0.00	0.00
OrchLodg	O	2013	11.76	0.00	0.09	0.00	0.00	0.51	7.00	0.00	0.00	0.00	0.25	0.00
OrchLodg	S	2013	12.75	0.00	0.09	0.00	0.00	0.50	4.82	0.00	0.00	0.00	0.25	0.00
Santon	A	2013	35.20	5.00	0.00	0.00	0.00	2.55	20.00	0.00	0.00	0.00	0.00	0.00
Santon	O	2013	35.20	5.00	0.00	0.00	0.00	2.55	20.00	0.00	0.00	0.00	0.00	0.00
Santon	S	2013	34.20	4.00	0.00	0.00	0.00	2.55	20.00	0.00	0.00	0.00	0.00	0.00
WPF	A	2013	30.31	0.00	0.20	0.00	1.20	0.35	14.50	0.00	0.00	0.00	0.00	1.46
WPF	O	2013	30.31	0.00	0.20	0.00	1.20	0.35	14.50	0.00	0.00	0.00	0.00	1.46
WPF	S	2013	31.58	0.00	0.20	0.00	1.20	1.05	15.00	0.00	0.00	0.00	0.00	1.52

Table 5 Fungicides 2014

Plot - (A) agricultural, (O) orchard, (S) unmanaged *TFI = total number of pesticide treatments, weighted by the ratio of the dose used to the recommended dose (Kg or L per hectare)*				Fungicides									
Farm	Plot	Year	TFI Fungicides	*boscalid pyraclostrobin (Kg)*	*bupirimate n-butanol hydrocarbons (Kg)*	*captan (Kg)*	*copper (Kg)*	*cyflufenamid (L)*	*cyprodinil fludioxonil (L)*	*difenoconazole (L)*	*dithianon (L)*	*dithianon WG (Kg)*	*dodine (L)*
Batteries	A	2014	16.47	0.00	0.00	0.00	0.00	0.00	0.00	0.60	0.75	0.75	0.00
Batteries	O	2014	14.97	0.00	0.00	0.00	0.00	0.00	0.00	0.60	0.75	0.75	0.00
Batteries	S	2014	16.47	0.00	0.00	0.00	0.00	0.00	0.00	0.60	0.75	0.75	0.00
Densted	A	2014	17.72	0.00	0.00	0.50	0.00	0.00	0.00	0.60	0.75	1.50	0.00
Densted	O	2014	18.67	0.00	0.00	0.50	0.00	0.00	0.00	0.60	0.75	0.75	0.00
Densted	S	2014	17.72	0.00	0.00	0.50	0.00	0.00	0.00	0.60	0.75	1.50	0.00
Griffins	A	2014	40.61	1.60	1.20	1.50	0.00	1.00	0.00	0.20	1.10	0.00	0.00
Griffins	O	2014	40.61	1.60	1.20	1.50	0.00	1.00	0.00	0.20	1.10	0.00	0.00
Griffins	S	2014	40.61	1.60	1.20	1.50	0.00	1.00	0.00	0.20	1.10	0.00	0.00
HMounts	A	2014	20.59	0.80	0.00	0.50	0.00	0.00	0.00	0.60	0.00	2.25	0.00
HMounts	O	2014	20.98	0.00	0.00	0.50	0.00	0.00	0.00	8.00	0.00	2.25	0.00
HMounts	S	2014	21.49	0.00	0.00	0.50	0.00	0.00	0.00	0.60	0.00	2.25	0.00
Loddington	A	2014	33.16	2.85	0.00	2.58	0.00	1.01	0.00	0.20	1.00	3.29	4.00
Loddington	O	2014	37.79	0.80	0.00	2.00	0.00	1.01	0.00	0.20	0.00	2.01	5.04
Loddington	S	2014	40.79	2.85	0.00	3.00	0.00	1.01	0.00	0.20	1.00	3.76	5.19
OrchLodg	A	2014	24.43	1.02	0.00	0.63	0.00	0.42	10.44	0.35	1.32	0.00	0.00
OrchLodg	O	2014	24.43	1.02	0.00	0.63	0.00	0.42	10.44	0.35	1.32	0.00	0.00
OrchLodg	S	2014	24.43	1.02	0.00	0.63	0.00	0.42	10.44	0.35	1.32	0.00	0.00
Santon	A	2014	16.96	0.00	0.00	0.50	0.00	0.00	0.00	0.80	1.50	0.75	0.00
Santon	O	2014	15.26	0.00	0.00	0.50	0.00	0.00	0.00	0.80	1.50	0.75	0.00
Santon	S	2014	16.96	0.00	0.00	0.50	0.00	0.00	0.00	0.80	1.50	0.75	0.00
WPF	A	2014	29.31	0.80	0.00	4.00	0.00	1.00	0.00	0.00	0.00	1.00	5.01
WPF	O	2014	29.31	0.80	0.00	4.00	0.00	1.00	0.00	0.00	0.00	1.00	5.01
WPF	S	2014	33.16	0.80	0.00	4.00	0.00	1.00	0.00	0.00	0.00	1.00	5.01

Table 5 Fungicides 2014 – continued.

Plot - (A) agricultural, (O) orchard, (S) unmanaged *TFI = total number of pesticide treatments, weighted by the ratio of the dose used to the recommended dose (Kg or L per hectare)*				Fungicides										
Farm	Plot	Year	TFI Fungicides	*fenbuconazole (Kg)*	*kresoxim-methyl (Kg)*	*liquid copper (L)*	*meptyldinocap (L)*	*myclobutanil cyclohexanone (L)*	*penconazole (L)*	*potassium hydrogen carbonate (Kg)*	*pyraclostrobin dithianon (Kg)*	*pyrimethanil (L)*	*tebuconazole (L)*	*tebuconazole B (L)*
Batteries	A	2014	16.47	0.00	0.00	0.00	0.00	2.57	9.30	0.00	2.50	0.00	0.00	0.00
Batteries	O	2014	14.97	0.00	0.00	0.00	0.00	2.57	7.80	0.00	2.50	0.00	0.00	0.00
Batteries	S	2014	16.47	0.00	0.00	0.00	0.00	2.57	9.30	0.00	2.50	0.00	0.00	0.00
Densted	A	2014	17.72	0.00	0.00	0.00	0.00	2.57	9.30	0.00	2.50	0.00	0.00	0.00
Densted	O	2014	18.67	0.00	0.00	0.00	0.00	2.57	11.00	0.00	2.50	0.00	0.00	0.00
Densted	S	2014	17.72	0.00	0.00	0.00	0.00	2.57	9.30	0.00	2.50	0.00	0.00	0.00
Griffins	A	2014	40.61	0.00	0.40	7.00	0.60	0.66	24.60	0.00	0.00	0.00	0.00	0.75
Griffins	O	2014	40.61	0.00	0.40	7.00	0.60	0.66	24.60	0.00	0.00	0.00	0.00	0.75
Griffins	S	2014	40.61	0.00	0.40	7.00	0.60	0.66	24.60	0.00	0.00	0.00	0.00	0.75
HMounts	A	2014	20.59	0.00	0.00	0.00	0.00	2.34	11.60	0.00	2.50	0.00	0.00	0.00
HMounts	O	2014	20.98	0.00	0.00	0.00	0.00	2.13	8.10	0.00	0.00	0.00	0.00	0.00
HMounts	S	2014	21.49	0.00	0.00	0.00	0.00	2.34	13.30	0.00	2.50	0.00	0.00	0.00
Loddington	A	2014	33.16	0.00	0.20	0.00	0.65	1.85	12.03	2.00	0.00	0.00	0.00	1.50
Loddington	O	2014	37.79	0.00	0.20	0.00	1.20	2.80	15.03	6.00	0.00	0.00	0.00	1.50
Loddington	S	2014	40.79	0.00	0.20	0.00	1.20	1.85	15.03	4.00	0.00	0.00	0.00	1.50
OrchLodg	A	2014	24.43	0.30	0.00	3.42	0.00	0.48	6.05	0.00	0.00	0.00	0.00	0.00
OrchLodg	O	2014	24.43	0.30	0.00	3.42	0.00	0.48	6.05	0.00	0.00	0.00	0.00	0.00
OrchLodg	S	2014	24.43	0.30	0.00	3.42	0.00	0.48	6.05	0.00	0.00	0.00	0.00	0.00
Santon	A	2014	16.96	0.00	0.00	0.00	0.00	2.11	8.80	0.00	2.50	0.00	0.00	0.00
Santon	O	2014	15.26	0.00	0.00	0.00	0.00	2.11	7.10	0.00	2.50	0.00	0.00	0.00
Santon	S	2014	16.96	0.00	0.00	0.00	0.00	2.11	8.80	0.00	2.50	0.00	0.00	0.00
WPF	A	2014	29.31	0.00	0.20	0.00	1.20	1.85	12.00	0.00	0.00	0.00	0.00	2.25
WPF	O	2014	29.31	0.00	0.20	0.00	1.20	1.85	12.00	0.00	0.00	0.00	0.00	2.25
WPF	S	2014	33.16	0.00	0.20	0.00	1.20	1.85	14.00	2.00	0.00	0.00	0.00	2.10

Table 6 Fungicides 2015 – continued.

Plot - (A) agricultural, (O) orchard, (S) unmanaged *TFI = total number of pesticide treatments, weighted by the ratio of the dose used to the recommended dose (Kg or L per hectare)*				**Fungicides**									
Farm	Plot	Year	TFI Fungicides	*boscalid pyraclostrobin (Kg)*	*bupirimate n-butanol hydrocarbons (Kg)*	*captan (Kg)*	*copper (Kg)*	*cyflufenamid (L)*	*cyprodinil fludioxonil (L)*	*difenoconazole (L)*	*dithianon (L)*	*dithianon WG (Kg)*	*dodine (L)*
Batteries	A	2015	17.15	0.00	0.00	11.50	1.25	0.00	0.00	0.20	0.00	0.00	1.50
Batteries	O	2015	17.15	0.00	0.00	11.50	1.25	0.00	0.00	0.20	0.00	0.00	1.50
Batteries	S	2015	19.15	0.00	0.00	13.50	1.25	0.00	0.00	0.20	0.00	0.00	1.50
Densted	A	2015	9.90	0.00	0.00	7.50	0.00	0.00	0.00	0.20	0.00	0.00	0.00
Densted	O	2015	12.40	0.00	0.00	9.50	0.00	0.00	0.00	0.20	0.00	0.00	0.00
Densted	S	2015	9.90	0.00	0.00	7.50	0.00	0.00	0.00	0.20	0.00	0.00	0.00
Griffins	A	2015	33.71	1.60	1.20	18.00	0.00	1.00	0.00	0.40	1.10	0.00	2.50
Griffins	O	2015	33.71	1.60	1.20	18.00	0.00	1.00	0.00	0.40	1.10	0.00	2.50
Griffins	S	2015	33.71	1.60	1.20	18.00	0.00	1.00	0.00	0.40	1.10	0.00	2.50
HMounts	A	2015	11.84	0.00	0.00	8.00	0.00	0.00	0.00	0.00	0.75	0.00	0.00
HMounts	O	2015	7.79	0.00	0.00	6.00	0.00	0.00	0.00	0.20	0.00	0.00	0.00
HMounts	S	2015	11.84	0.00	0.00	8.00	0.00	0.00	0.00	0.00	0.75	0.00	0.00
Loddington	A	2015	33.83	1.20	0.00	12.79	3.25	1.00	0.00	0.60	0.00	0.75	4.50
Loddington	O	2015	35.08	1.20	0.00	12.79	3.25	1.00	0.00	0.60	0.00	0.75	4.50
Loddington	S	2015	29.13	0.60	0.00	10.99	2.86	0.80	0.00	0.40	0.00	1.13	5.91
OrchLodg	A	2015	14.01	0.68	0.00	6.40	0.00	0.21	0.04	0.27	0.00	0.00	0.00
OrchLodg	O	2015	14.01	0.68	0.00	6.40	0.00	0.21	0.04	0.27	0.00	0.00	0.00
OrchLodg	S	2015	14.01	0.68	0.00	6.40	0.00	0.21	0.04	0.27	0.00	0.00	0.00
Santon	A	2015	8.63	0.00	0.00	5.50	0.00	0.00	0.00	0.20	0.00	0.00	1.50
Santon	O	2015	8.63	0.00	0.00	5.50	0.00	0.00	0.00	0.20	0.00	0.00	1.50
Santon	S	2015	8.63	0.00	0.00	5.50	0.00	0.00	0.00	0.20	0.00	0.00	1.50
WPF	A	2015	30.77	0.60	0.15	13.00	0.25	0.74	0.00	0.20	0.00	0.38	3.97
WPF	O	2015	30.77	0.60	0.15	13.00	0.25	0.74	0.00	0.20	0.00	0.38	3.97
WPF	S	2015	34.22	0.60	0.15	13.00	0.25	0.74	0.00	0.20	0.00	0.38	3.97

Table 6 Fungicides 2015 – continued

Plot - (A) agricultural, (O) orchard, (S) ununmanaged *TFI = total number of pesticide treatments, weighted by the ratio of the dose used to the recommended dose (Kg or L per hectare)*				Fungicides										
Farm	Plot	Year	TFI Fungicides	*fenbuconazole (Kg)*	*kresoxim-methyl (Kg)*	*liquid copper (L)*	*meptyldinocap (L)*	*myclobutanil cyclohexanone (L)*	*penconazole (L)*	*potassium hydrogen carbonate (Kg)*	*pyraclostrobin dithianon (Kg)*	*pyrimethanil (L)*	*tebuconazole (L)*	*tebuconazole B (L)*
Batteries	A	2015	17.15	0.00	0.00	0.00	0.00	0.00	1.20	0.00	0.00	1.50	0.00	0.00
Batteries	O	2015	17.15	0.00	0.00	0.00	0.00	0.00	1.20	0.00	0.00	1.50	0.00	0.00
Batteries	S	2015	19.15	0.00	0.00	0.00	0.00	0.00	1.20	0.00	0.00	1.50	0.00	0.00
Densted	A	2015	9.90	0.00	0.00	0.00	0.00	0.00	1.20	0.00	0.00	1.00	0.00	0.00
Densted	O	2015	12.40	0.00	0.00	0.00	0.00	0.00	1.20	0.00	0.00	1.50	0.00	0.00
Densted	S	2015	9.90	0.00	0.00	0.00	0.00	0.00	1.20	0.00	0.00	1.00	0.00	0.00
Griffins	A	2015	33.71	0.00	0.40	2.50	0.60	0.66	2.00	0.00	0.00	1.75	0.00	0.00
Griffins	O	2015	33.71	0.00	0.40	2.50	0.60	0.66	2.00	0.00	0.00	1.75	0.00	0.00
Griffins	S	2015	33.71	0.00	0.40	2.50	0.60	0.66	2.00	0.00	0.00	1.75	0.00	0.00
HMounts	A	2015	11.84	0.00	0.00	0.00	0.00	0.00	1.59	0.00	0.00	1.50	0.00	0.00
HMounts	O	2015	7.79	0.00	0.00	0.00	0.00	0.00	1.59	0.00	0.00	0.00	0.00	0.00
HMounts	S	2015	11.84	0.00	0.00	0.00	0.00	0.00	1.59	0.00	0.00	1.50	0.00	0.00
Loddington	A	2015	33.83	0.00	0.40	0.00	0.96	2.37	1.00	0.00	5.00	0.00	0.00	0.00
Loddington	O	2015	35.08	0.00	0.40	0.00	0.96	2.37	1.50	0.00	5.00	0.00	0.00	0.75
Loddington	S	2015	29.13	0.00	0.40	0.00	0.96	2.49	1.00	0.00	0.00	0.85	0.00	0.75
OrchLodg	A	2015	14.01	0.41	0.00	2.28	0.00	0.39	2.71	0.00	0.00	0.62	0.00	0.00
OrchLodg	O	2015	14.01	0.41	0.00	2.28	0.00	0.39	2.71	0.00	0.00	0.62	0.00	0.00
OrchLodg	S	2015	14.01	0.41	0.00	2.28	0.00	0.39	2.71	0.00	0.00	0.62	0.00	0.00
Santon	A	2015	8.63	0.00	0.00	0.00	0.00	0.00	0.93	0.00	0.00	0.50	0.00	0.00
Santon	O	2015	8.63	0.00	0.00	0.00	0.00	0.00	0.93	0.00	0.00	0.50	0.00	0.00
Santon	S	2015	8.63	0.00	0.00	0.00	0.00	0.00	0.93	0.00	0.00	0.50	0.00	0.00
WPF	A	2015	30.77	0.00	0.60	0.00	0.97	2.05	1.00	6.01	0.00	0.85	0.00	0.00
WPF	O	2015	30.77	0.00	0.60	0.00	0.97	2.05	1.00	6.01	0.00	0.85	0.00	0.00
WPF	S	2015	34.22	0.00	0.60	0.00	0.97	2.50	1.00	9.01	0.00	0.85	0.00	0.00