Is Sustainable Soil Management Good for Farmland Birds?

Sofia Biffi, BSc, MRes

School of Biology

University of Leeds

Submitted in accordance with the requirements for the degree of

Doctor of Philosophy

September, 2020

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Abstract

Conventional arable systems are designed to produce food at high levels of efficiency, but usually require large amounts of energy and chemical inputs per unit of land, with cascading detrimental effects throughout the field agroecosystem. Thus, agricultural intensification processes have been linked with the decline of farmland bird and invertebrate populations, and with arable soil degradation. Today, agricultural policies and practices are seeking to develop arable systems that are less dependent on these inputs, and sustainable soil management (SSM) practices have been introduced as a way to maintain soil as a healthy living system and support agricultural production.

This thesis aimed to assess if cover crop and green waste compost amendment have the potential to benefit farmland bird populations during breeding season. In particular, the thesis focused on the association of these practices with belowground and aboveground invertebrate abundance and assemblages, chick food availability, and Skylark *Auleda arvensis* territory settlement during breeding season. This study took place at Hope Farm, a commercial arable farm owned by the RSPB that has been operating since 2015 a trial of the effects of SSM in three experimental split-plot fields.

Cover crops were shown to have a positive association with belowground invertebrate abundance and soil biological quality, and with the abundance of canopy dwelling invertebrates, suggesting that this practice may support invertebrate populations. Green compost, albeit strongly increasing soil organic carbon content, did not have an effect on invertebrates. Increased abundance of invertebrates was not reflected by an increase in in-field food availability to breeding birds, and the distribution of Skylark territories did not vary among plots with different soil treatment management over the course of five years. Thus, the results of this thesis suggest that cover crops and green compost did not benefit farmland birds at Hope Farm during summer. This thesis highlights the importance of evaluating the effects of individual in-field soil management practices expected to target broad environmental goals.

Acknowledgements

I wish to thank Professor Les Firbank, my primary supervisor in this doctoral research and already my Master thesis advisor. My gratitude goes to someone who has always been available and present, and who has made it possible to conduct this research, believing in myself and my abilities. Without his guidance, feedback, and light-heartedness this PhD would not have been achievable. I would also like to thank my co-supervisors Professor Guy Ziv and Professor Bill Kunin, whose breadth of scientific knowledge and ability to bridge research topics has been an inspiration to me.

I would like to gratefully acknowledge the financial support provided by the University of Leeds (University Research Scholarship) and the generous donation from Nigel Bertram (Bertram Scholarship), which have allowed me to pursue my studies.

A heartfelt thank you goes to Dr Rob Field, who has always been available to discuss my work, help in the field, and provide useful guidance. Thank you also to Georgina Bray and Derek Gruar at Hope Farm for sharing their expertise while I was at the farm and providing me with the Skylark survey maps.

A number of people helped along the way. I am very grateful to the teaching laboratories staff, Sarah and Dote in particular, who provided me with so much technical support. A big thank you also to my tireless field assistants, Megan Tresise and Talor Whitham. Thank you to the farmers in Cambridgeshire and Lincolnshire who gave me access to their land. Thank you also to Dr Steve Compton and Dr Rodger Key, who helped me improve greatly my identification skills. I will always remember demonstrating during ground beetles' week fondly.

A warm thank you to all the friends I made in Leeds, particularly to Tom, Myrna, and Dan for our sanity coffee breaks. Dan, your friendship in the past years has been a rock.

Un grazie di cuore va ai miei genitori, che mi sono sempre stati vicini –anche a distanza– e che mi hanno sempre incoraggiata ad impegnarmi nel mio lavoro e a trovare felicità e leggerezza in quello che faccio. Vi voglio bene. Finally, I would like thank my patient and encouraging partner Philipp, this years-long roller coaster ride would have been tremendously less fun without you by my side. Thank you for everything.

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List of abbreviations

AES	Agri-environment scheme
ANOSIM	Analysis of similarities
CFI	Chick Food Index
FBI	Farmland bird Indicator
GLM	Generalised linear model
GLMM	Generalised linear mixed model
LMM	Linear mixed model
PERMANOVA	Permutational multivariate analysis
QBS-ar	Soil biological quality arthropod index
SOM/SOC	Soil organic matter/carbon
SSM	Sustainable soil management

Soil management

CvCo	Cover crop and green compost
Cv	Cover crop only
Co	Green compost only
Nil	Traditional soil management

Chapter 1

Introduction

Agriculture transforms natural habitats into systems designed to promote certain species for human consumption, using energy and chemical inputs to enhance, alter, replace, or bypass specific biological functions. Agricultural intensification processes, aimed at increasing agricultural production per unit of cultivated area, affect – at least to a certain degree, most of the agroecosystem functions, and are recognized as major drivers of loss of biodiversity, soil degradation, and air and water pollution (Foley et al. 2005). The European agro-ecological history has been described as "a dynamic succession of more or less traumatic perturbations of varying duration and amplitude" since the arrival of Man in Europe at the end of the last glaciation (Bignal and McCracken 1996). Since the second half of the XX century, however, agricultural landscapes have undergone rapid and extensive agricultural intensification with great increases of productivity per unit of land and labour, and inevitable effects on biodiversity and soil health. Today, agricultural policies and practices are seeking to ameliorate these negative effects, and sustainable soil management practices have been proposed to increase the sustainability of agricultural systems (Powlson et al. 2011; Garratt et al. 2018).

1.1 Agricultural threats to biodiversity

Agricultural expansion and intensification are recognized as main drivers in the global decrease in terrestrial and freshwater biodiversity (Pimm et al. 1995; Foley et al. 2005; Geiger et al. 2010a; Newbold et al. 2015; Firbank and Biffi 2017). The intensification of European agriculture has been associated with the dramatic decline of farmland bird populations (Donald et al. 2001c), arable weeds (Fried et al. 2009; Andreasen and Streibig 2011), butterfly (van Swaay and Warren 1999; van Swaay et al. 2008), and pollinator populations (Potts et al. 2010). The first cause of biodiversity depletion in agricultural systems is habitat loss through the clearance of natural vegetation, a problem that today primarily threatens the developing world (Pimm and Raven 2000; Laurance et al. 2014).

In established agrosystems, above and belowground terrestrial biodiversity can be directly threatened by the most frequent practices of agricultural intensification: landscape homogenisation, increased chemical inputs, and other changes in crop management, such as timing of harvest, crop rotation, and tillage depth (Firbank et al. 2008). While traditional agricultural landscapes retained a variety of semi-natural habitat patches that allowed species to survive and even thrive in agroecosystems, the loss of these semi-natural features during the transition to intensive landscape is responsible for biodiversity decline (Benton et al. 2003; Tscharntke et al. 2005; Fahrig et al. 2011). In intensive agricultural landscapes, chemical inputs are responsible for biodiversity decline through the application of pesticides and herbicides (Geiger et al. 2010a; Pisa et al. 2015; Woodcock et al. 2017), and excessive fertilisation, which affects terrestrial (Hautier et al. 2009) and aquatic biodiversity (Smith 2003) also outside farmland, as a result of aerial deposition (Phoenix et al. 2012).

In agricultural landscapes, biodiversity is also affected at the local and landscape scale by changes in crop rotation, which reduce the spatial availability of resources (Chamberlain et al. 2000). The cropped species themselves have an impact on biodiversity, as they are subject to specific cultivation and management regimes that can affect the agroecosystem differently (Stoate et al. 2009). Belowground, the biomass and diversity of soil biota are altered as a direct effect of agricultural practices (e.g. tillage) and because of soil physical and chemical degradation (van Capelle et al. 2012).

1.1.1 European agriculture and farmland birds

Most European farmland biodiversity has adapted to landscape changes over a long period of time and has evolved in association with traditional farming practices (Stoate et al. 2001). The transition from woodland to grassland and arable land since the arrival of man in Europe allowed certain ground feeding and/or open country bird species, such as finches, buntings, pigeons, partridges, quails, larks and corvids, to thrive in agricultural systems until the XX century (Shrubb 2003). From the late 1950s, however, the European Common Agricultural Policy (CAP) promoted policies encouraging the rapid intensification of agriculture. Today, the European Union (EU) presents one of the most intensive agriculture systems in the world, as population pressure per hectare of agricultural land is amongst the highest. European farmland bird populations have fallen by 57% between 1980 and 2017 (EBCC 2018), and agricultural intensification is undoubtedly one of the major causes of this decline.

In Europe, and in the United Kingdom in particular, birds have been one of the major focuses of research on agricultural intensification processes and biodiversity, with numerous studies published on how these affect birds abundance and diversity (Potts and Aebischer 1995; Chamberlain et al. 2000; Donald et al. 2001c; Vickery et al. 2001; Reif 2013; Jerrentrup et al. 2017; Bowler et al. 2019). Bird populations are known to be directly affected by the simplification of croplands' structural diversity (Hinsley and Bellamy 2000; Chamberlain and Fuller 2001; Gaston et al. 2003; Gil-Tena et al. 2015), by the biological magnification of toxic pesticides (Walker 1983; Rands 1985) and by the depletion of inveterate and weed food resources through the application of pesticides and herbicides (Donald et al. 2006; Holland et al. 2006).

As a result of intensive agricultural practices, the reduced invertebrate prey availability during breeding season in particular has been identified as one of the main drivers of bird populations decline (Benton et al. 2002; Hallmann et al. 2014; Schrauth and Wink 2018). Indeed, the reduction in aboveground invertebrates observed, for example, in Germany (Hallmann et al. 2017), and in lowland Britain (Shortall et al. 2009) has occurred in parallel to the decline of farmland birds species, as most of these rely on invertebrate protein intake for growth and survival during the early stages of development (Southwood and Cross 2002; Hart et al. 2006; Douglas et al. 2012).

1.2 Soil health and agroecosystems

In intensive arable landscapes, the patterns of soil degradation associated with conventional agricultural practices have been shown to have cascading detrimental effects throughout the field agroecosystem (Matson et al. 1997; Six et al. 2000; Diacono and Montemurro 2010). In particular, soil degradation is tightly connected to loss of soil organic matter (SOM) content, which among the most important indicators of soil fertility and agroecosystem functions (Kibblewhite et al. 2008; M. W. Schmidt et al. 2011; Oldfield et al. 2019).

1.2.1 Causes of arable soil degradation

All agricultural soils undergo a process of alteration from their natural state to maximise production of food and fibre. This alteration starts with the clearance of natural vegetation, which has a strong and immediate detrimental effect on soil properties (Dalal et al. 2005), and continues with mechanical practices and chemical applications to maximise yield. Thus, agricultural intensification is responsible for soil degradation, "a process which lowers the current and/or future capacity of the soil to produce services" (Oldeman 1994). The intensity of agriculture varies significantly across cropping systems, and the resilience of soil to degradation varies across climatic and topographic situations depending on the combination of soil type and agricultural system. Although soil degradation generally affects tropical areas more severely than temperate areas (Bai et al. 2008), soil deterioration processes concern any soil under intensive cropping regimes.

In intensive arable systems, the depletion of SOM can be caused by mineral fertilisers surplus, which increases the rate of soil organic matter (SOM) mineralisation in the soil (FOG 1988; Nicolardot et al. 2001), and by intensive tillage, which exposes SOM to microbial decomposition and erosion (Koch and Stockfisch 2006; J. M. Baker et al. 2007). Higher soil temperatures and moisture conditions associated with anthropogenic climate change can also reduce SOM through respiration, rendering arable soils among the main drivers of global greenhouse gasses emissions (Lal 2004). Since the 1960s the once widespread route of adding organic fertilisers to increase the SOM entering the soil system has decreased in conventional agriculture, while in organic farming it has been shown to benefit soil biota (Martínez-García et al. 2018). High nitrogen surplus rates, combined with the decrease in use of manure as fertiliser, increased cultivation depth, and anthropogenic climate change, contribute to the decreasing trend of SOM content throughout arable landscapes in Europe and the UK (Jones et al. 2012; A. Gregory et al. 2015).

Soil physical degradation is often caused by physical stress, such as farm traffic or high livestock densities, and it is also related to the decline of SOM. Physical degradation is an alteration of the structural attributes of soil, and one of its most common manifestations is the modification and reduction of soil pores' geometry, size, and continuity (Greenland 1981). SOM is largely controls the structure of the soil, such as soil porosity (Tisdall and Oades 1982), thus influencing the movement of water (Franzluebbers 2002), but also of roots, and micro and mesofauna in the soil (Jabbour and Barbercheck 2008). Most commonly, physical degradation takes the form of soil compaction (Hamza and Anderson 2005), the most widespread forms of agricultural soil degradation in Europe (Jones et al. 2012) and in the UK (A. Gregory et al. 2015). Physical degradation of arable soils increases their risk of waterlogging, wind and water erosion, and temperature fluctuations (Belnap 1995; Batey 2009), and is associated with decreased soil biota (Carlesso et al. 2019).

1.2.2 Aboveground and belowground feedbacks

Aboveground and belowground components of an ecosystems should not be considered as separate, as they are regulated by a great number of feedback processes, most of which –particularly within agricultural production systems– are yet not well understood. The complexity of biotic interactions between aboveground and belowground ecosystem com-

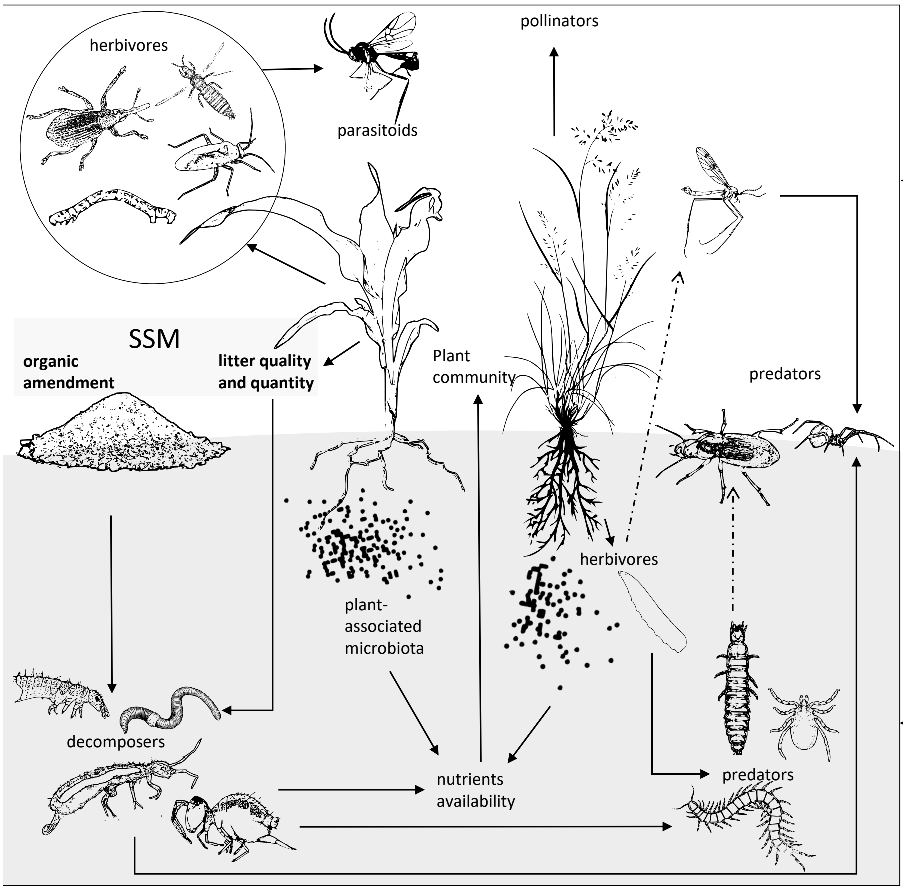

Figure 1.1: Simplified scheme emphasizing trophic relationships and illustrating how some sustainable soil management practices may impact belowground and aboveground invertebrates by increasing quantity and quality of SOM availability. Dash-dot lines represent emergence from larval stages.

ponents is enormous (Wardle et al. 2004a; Dedeyn and Vanderputten 2005), and in the past two decades an increasing number of above and belowground interactions have gained interest in terrestrial ecology, such as the influence of soil biota on plant growth and diversity (Van Der Heijden et al. 2008; Andriuzzi et al. 2015; Filser et al. 2016; Bonkowski and Roy 2005), the effects of aboveground biota on soil health and biota (Strickland et al. 2013), and the links between above and belowground trophic levels (Bever et al. 1997; Wardle et al. 2004b; Bezemer et al. 2005; Johnson et al. 2011; C. Zhao et al. 2013). Most of these studies have been conducted on natural systems, but there is a growing recognition of their potential application for sustainable agricultural production (Veen et al. 2019).

At the basis of the belowground-aboveground feedbacks is SOM and, in arable landscapes, aboveground processes drive belowground communities primarily by affecting the quality and quantity of SOM that enters the soil decomposers food web; in turn, the belowground community affects plant growth potential by regulating nutrient cycling (Wardle et al. 2004a; Veen et al. 2019). The SOM entering an arable soil system is driven by the crop roots and residues, by the identity of crop species present in the rotation (Scheunemann et al. 2015), and –if used– by organic inputs. The identity of cultivated crops has been shown also as a driver of aboveground and belowground biota abundance and diversity in time, as plants can have legacy effects on soil processes and biota long after their removal (Crotty et al. 2015; Wubs and Bezemer 2018; Wubs et al. 2019).

When SOM is available, the soil community is generally more functional and there are more feedback processes with the aboveground community. For example, SOM availability is a driver of abundance of soil macro and mesofauna, which plays a fundamental role in the soil system functioning (Barrios 2007). In addition to affecting plant growth, soil invertebrates are associated to aboveground invertebrates, as they affect nutrient availability for herbivorous species, and predator species, such as ground beetles and spiders (Scheu 2001; Wardle et al. 2004a). Due to these trophic linkages, the enhancement of SOM in arable soils has been suggested as a practices to enhance invertebrate populations, a practice usually studied in terms of pest management (Settle et al. 1996; Lundgren and Fergen 2011; Blubaugh et al. 2016; Riggi and Bommarco 2019).

1.3 Sustainable soil management

Although many of the above-belowground interactions are yet not well understood, there is a growing recognition of the importance of maintaining soil as a healthy living system in supporting agricultural production (Van Der Heijden et al. 2008; Wagg et al. 2014; Veen et al. 2019). In recent years there has been a spread of “sustainable” soil management

(SSM) practices in intensive agricultural landscapes, mostly driven by cost effective measures to partially substitute the use of fertilisers. SSM has been introduced to European agri-environment schemes (Turpin et al. 2017) and is encouraged as a way to increase agricultural soil health and sustainability (Powlson et al. 2011; Bommarco et al. 2013; Garratt et al. 2018). The SSM practices in intensive arable systems include reduced tillage, increased crop diversification, use of cover crops in the cash crop rotation, and use of organic soil amendments (Kremen and Miles 2012; Garratt et al. 2018).

Because of the links between soil health and agroecosystem functions, the general trend of increased soil health with these soil management practices is viewed as an opportunity to enhance invertebrate abundance and diversity on farmland for pest control and agroecosystem functioning (Crotty et al. 2015; Riggi and Bommarco 2019, Fig.1.1). Due to the linkage between invertebrate availability and bird populations, SSM practices have the potential to be beneficial to farmland birds, however, there are surprisingly few publications on the effects of specific SSM practices on farmland biodiversity. Their association with farmland birds and invertebrate abundance and availability, remains mostly unexplored.

1.3.1 Cover crops

Cover crops are grown on fields that would otherwise be bare, and their addition in the cash crop rotation has been shown to positively affect soil chemical and physical properties (Schipanski et al. 2014; Wittwer et al. 2017). They are being increasingly used, especially in the USA, where they are subsidised by the Environmental Quality Incentive Program (EQIP) to maintain water quality and reduce soil erosion. In the EU, as part of the mandatory Member States ”greening” measures under Pillar I, cover crops are one of the most common options for Ecological Focus Areas (Pe’er et al. 2017). In the UK, cover crops have been increasing due to their offer as part of AES and in association with the rise of spring sown crops, which is caused by increasingly wet autumn weather conditions and widespread issues with black-grass (*Alopecurus myosuroides*) throughout the country. Black grass is a highly competitive weed that has to be targeted with broad spectrum herbicides during germination in autumn and winter (AHDB 2019). The switch to spring sown crop has favoured the increased use of cover crops, as uncropped fields over winter can be subject to waterlogging and nutrient leeching (Tonitto et al. 2006; Kaspar and Singer 2011; A. Gregory et al. 2015).

Historically, cover crops have been often used as ”catch” crops to retain nitrates that would otherwise leach out of the soil system and contaminate water, and it has been shown that cover crops can contribute to the long-term nitrogen availability in the soil, depending on the quantity and quality of biomass returned to it (Kuo et al. 1997b). Cover crops can also reduce soil moisture content through uptake and transpiration when alive, however, once terminated the residues on the soil surface can result in higher water retention. If managed correctly, cover crops can increase soil moisture in dry climates (Acharya et al. 2019), and be successful in decreasing water from the fields after overly wet seasons aiding the cultivation process over spring (Qi and Helmers 2010).

The effects of cover crops on soil physical and chemical properties vary depending on the soil type, texture, depth and climate of the cropping system, and on the species used (Kaspar and Singer 2011; Ramirez-Garcia et al. 2015). Several studies have shown that cover crops mixtures can exceed the performance of a single cover crop species, as different species identity can positively interact in the soil system. While the interaction of plant species in grasslands have been widely studied, information on the effects of cover crop mixtures on soil biota is limited (Wendling et al. 2017; Wendling et al. 2019). A study in the UK, however, has identified a mix of up to four species as optimal, as it allows to exploit different functional roles (Storkey et al. 2015). The level SOM increases with cover crops also depends on the species identity. Kuo et al. (1997a), for example, showed that the build up of SOM in the soil after 6 years was higher when using ryegrass than legumes.

Although cover crops have a long history as sustainable soil management in arable systems, and they are often explicitly part of arable biodiversity conservation schemes, data on their effectiveness in enhancing biodiversity is surprisingly lacking (Kaspar and Singer 2011; Shackelford et al. 2019). Some research suggests that they may be beneficial to control pest insects (Tillman et al. 2004; Lundgren and Fergen 2010), and aid pollen reserves for native species (Ellis and Barbercheck 2015). A study on migratory birds in the Midwest showed an increase in abundance of a few bird species while cover crops were present in the field, but did not relate it to an increased prey availability (Wilcoxen et al. 2018).

While they are being widely promoted for wildlife, their beneficial effects on biodiversity are not undisputed. For example, there is some evidence that inter-seeded cover crops can be detrimental to invertebrate communities if planted in fields previously treated with systemic insecticides, as they can retain toxic concentrations and thus be a hazard to non-target species (Bredeson and Lundgren 2019). When investigated, the effects of cover crops on biodiversity are usually studied when the cover crop is present in the field (e.g. Storkey et al. 2015; Wilcoxen et al. 2018). Thus, there is little knowledge on the effect of cover crops on biodiversity during bird breeding season, after the crop termination and the cultivation of a cash crop, when birds' requirements of invertebrate prey are vital.

1.3.2 Green compost

Green compost is a soil amendment derived from municipal botanical waste such as trimmings from perennial and annual plants, grass clippings, vegetables, and fruit, produced by a variety of green spaces both publicly and privately owned and undergoing aerobical humification processes over the course of 8-18 months. Composting leads to the development of a product resembling soil humus in appearance, which may serve as a valuable source of organic matter to increase soil fertility of arable cropland (Cabrera et al. 2005; Hargreaves et al. 2008; Hepperly et al. 2009). This type of compost derived from parks and garden waste has been increasingly used in the UK after the landfill Directive (1993/31/EC) which required the reduction of biodegradable waste to 35% of 1995 levels by 2016 (European Commission 2011). Although the broad effects of this practice on soil physical and chemical conditions are established, we are still lacking evaluations of some of its environmental impacts. For example, some attention has been dedicated to the effects of this compost on microbial and fungi communities, while no studies evaluate its association with soil macrofauna or aboveground biota (Hepperly et al. 2009).

Green compost has been shown to increase soil organic matter content and increase soil pH depending on soil type. Moreover, it has been shown to improve soil porosity, water penetration and air circulation (Celik et al. 2004). Green compost cannot be considered as a nitrogen (N) fertiliser, as it has very low N content, however, it can aid mineral fertilisers (Keeling et al. 2003). It can also increase soil microbial biomass and respiration up to nearly a decade after application (Garcıa-Gil et al. 2000), thus, in some cases, it can help fighting soilborne diseases by introducing competing microorganisms (Tilston et al. 2002). Plant pathogens, however, can also increase in this product during later stages of the humification process (Hassen et al. 2001; Avery et al. 2012), and their presence is not currently being tested in the UK due to lack of validated detection methods (Wrap 2011). Compost application is also often associated to increase in heavy metals concentration (Illera et al. 2000), especially when the compost has been sourced from industrialised regions (Weber et al. 2007). Compost often also contains non-biodegradable materials that cannot be completely extracted by screening.

1.4 RSPB Hope farm

In the UK, the largely unexplored potential of some SSM practices in supporting declining bird populations in arable landscapes, is currently being investigated at Hope Farm. This farm, the main study area of this thesis, is a research farm that pioneered the development of agricultural sustainability in the form of AES, with particular focus on bird

Figure 1.2: Aerial view of Hope Farm, a 181 ha arable commercial farm in Cambridgeshire, UK (00082 55W, 52814 43N) owned by the Royal Society for the Protection of Birds (RSPB) since 2000. Vertical Aerial Imagery (2013), Scale 1:500, Using: EDINA Aerial Digimap Service.

populations (Morris 2006). Due to the longevity of successful agri-environment schemes implementation, which is reflected in higher abundance and diversity of invertebrates and birds on the farm than in the surrounding arable landscapes, and the consistency of its bird monitoring schemes, Hope Farm provides an exceptional insight into the potential effects of SSM practices on farmland birds.

Hope Farm (Fig.1.2 and Fig.1.3) is a 181.4 ha lowland arable commercial farm in Cambridgeshire, UK (00082 55W, 52814 43N). This farm is owned by the Royal Society for the Protection of Birds (RSPB) since 1999 and managed by an agricultural contractor with mineral fertilisation and chemical inputs applied to the crop following the UK standard scheme management (AHDB 2017). It is situated on Hanslope-series calcareous clay loam with poor drainage capacity and high sensitivity to compaction, and it comprises of 170 ha of arable (93% of area), 6 ha of grazed pasture (3%), small woodland patches (<1%) and 10.3 km of hedgerow.

Hope Farm was originally purchased with the objective to “trial, demonstrate and showcase wildlife-friendly intensive farming”, researching the feasibility of successfully running a commercial arable farm using pesticides and inorganic fertilisers, whilst locally reversing declines in farmland biodiversity through the application of agri-environmental measures (Morris et al. 2010; Field et al. 2016; Aebischer et al. 2016). In terms of crop management, from a clay-rich soil traditional three-year rotation with autumn sown crops (winter wheat/winter wheat/oilseed-rape), a four-year rotation with inclusion of both autumn-sown and spring-sown field beans was adopted to improve soil quality (winter wheat/oilseed-rape/winter wheat/field beans). In recent years, however, spring-sown malting barely has sometimes replaced winter wheat in this rotation. Moreover, the adoption of pest-resistant spring-sown varieties (i.e. orange wheat blossom midge resistant wheat), has allowed to avoid spraying broad-spectrum insecticides during breeding season.

Initially, the management of Hope Farm focused primarily on demonstrating the effectiveness of best practice Entry Level Stewardship Scheme (ELS) agreements, and has successively expanded to encompass soil management trials. Suitable habitats for nesting and foraging were provided with wide flower-rich and grass margins (many fields on the farm have 6 m wide margins) and set-asides, in-field skylark plots, managed hedgerows, and wild bird cover mixes in winter (Maddock 2008). The adoption of these practices has proven very successful, with a 226% increase in the Breeding Bird Index and a 1738% increase in the Winter Bird Index between 2000 and 2017 (RSPB 2017). The provision of flower and grass margins, beetle banks, and managed ditches has also resulted in a long term increase in pollinator invertebrate populations, with a 223% increment in Butterfly Index between 2001 and 2018, and 95% higher bumblebee abundance at Hope Farm than a nearby farm with no ELS implemented (RSPB 2018).

Sustainable soil management trial A field scale trial of the effects of green compost soil amendment and cover crops has been set in place at Hope Farm since 2015. Three fields were divided into four plots with different combinations of soil management, with and without cover crops and green waste compost amendment (Fig.1.3). The cover crop used in the trial was a phacelia (*Phacelia tanacetifolia*)/radish (*Raphanus sativus*)/vetch (*Vicia sativa*) mix. These species are recommended as cover crop in the UK, as *Brassicaceae* (i.e. radish) are known for their leaching-reducing deep root system, while *Leguminosae* (i.e. vetch) have nitrogen fixing properties (AHDB 2015). The green waste compost was supplied by a certified compost producer (WRAP 2011) and applied annually at c. 30 t/ha.

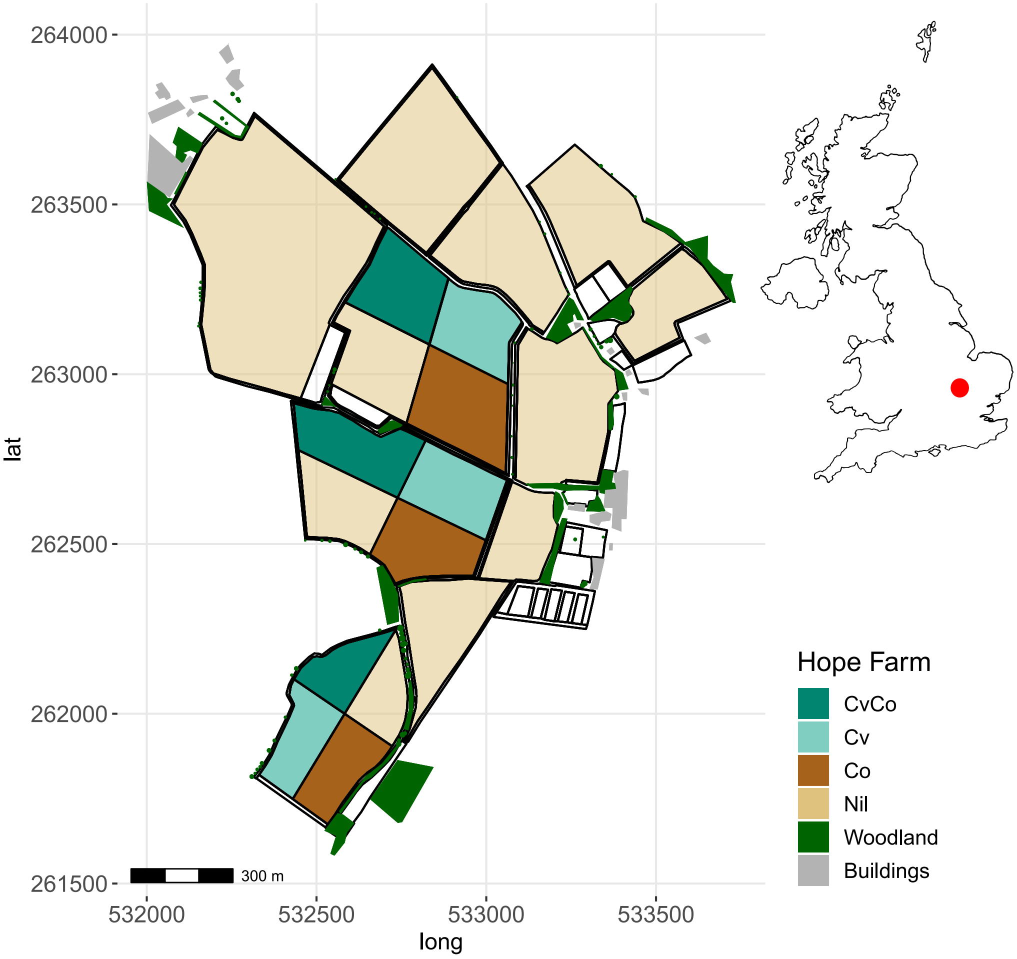

Figure 1.3: Three fields at Hope Farm have been undergoing a SSM trial since 2015, with combinations of cover cropping and green waste compost amendment. From the top, Field 5, Field 4, and Field 3. Approximate location of the farm in the UK indicated by the dot. CvCo = cover crop and compost; Cv = cover crop only; Co = compost only; Nil = no SSM.

1.5 Thesis overview

The association between the use of sustainable soil management in conventional arable fields and resources availability to farmland birds remains largely unexplored. The aim of this thesis is to assess **the potential of cover crops and green waste compost amendment use in a commercial arable system to benefit breeding farmland birds**. In particular, this thesis is focused on how these soil management practices may affect belowground invertebrates (Chapter 2), aboveground invertebrates (Chapter 3), chick food availability (Chapter 4), and territory settlement (Chapter 5) during breeding season. The thesis aim will be achieved through studies that

1. investigate belowground invertebrate abundance and quality to soil management type and micro-habitat characteristics known to affect invertebrates in arable landscapes,
2. investigate aboveground invertebrate abundance and assemblage to soil management type and micro-habitat characteristics known to affect invertebrates in arable landscapes,
3. formulate a new Chick Food Index and use it to assess effective food resources available to bird chicks among soil management types,
4. analyse a long-term dataset on the distribution of Skylarks in fields undergoing different management types.

The four analysis chapters of this thesis test the hypotheses:

1. the use of cover crops in arable fields will be beneficial to farmland birds,
2. the use of green waste compost amendment in arable fields will be beneficial to farmland birds,
3. the interaction between the use of cover crops and green compost will be beneficial to farmland birds.

It should be noted, that the decline of farmland birds in arable landscapes has not only been associated to breeding season conditions and food resources availability, but also to winter ones (Geiger et al. 2010b). Therefore, relating SSM to in-field summer conditions is just one side of the coin.

Chapter 2 – Sustainable soil management effects on belowground invertebrates

This study investigated the patterns in soil invertebrates abundance and soil biological quality (QBS-ar) associated with the use of cover crop and green waste compost amendment in split-plot and paired fields. The aim was to assess if SSM may affect soil meso and macrofauna, thus supporting the physical and chemical functioning of the field soil system (Barrios 2007), and, ultimately support greater invertebrate abundance available to farmland birds during breeding season (Felici et al. 2019). Variations in soil micro-habitat conditions with SSM of soil bulk density, water content, penetrability, and SOM content, were also investigated. SSM, habitat variables, and field boundary measurements of the response variables were then used as predictors of belowground meso and macrofauna abundance, soil biological quality, and palatable preys for birds in arable fields. The study took place in three split-plot experimental fields at Hope Farm, and at four commercial arable farms on similar soil types in Cambridgeshire and Lincolnshire with paired fields with and without SSM. The result of this study showed a positive effect of cover crop on both the total abundance of soil invertebrates and the soil biological quality, suggesting that the introduction of a cover crop in the cash crop rotation can have lasting beneficial effects on soil invertebrates.

Chapter 3 – Sustainable soil management effects on aboveground invertebrates This chapter used field data collected from the experimental fields at Hope Farm to examine the association between cover crop and green waste compost with the aboveground invertebrate community. Separate analyses were conducted on canopy dwelling invertebrates collected with suction sampling, and ground dwelling invertebrates collected in pitfall traps. Within the ground dwelling community, ground beetle species were also analysed separately. Changes in abundance and assemblage of these invertebrate groups among soil management treatments were investigated together with measures of SOM content, soil penetrability, vegetation cover, and response variable measures in the field boundary. Invertebrate abundance and assemblage were investigated to determine if the cover crop and green waste compost amendment may be associated with changes in the aboveground invertebrate community, which is an important driver of ecosystem processes (Yang and Gratton 2014), and provides vital resources to many farmland bird species during breeding season (Potts 1986; Hart et al. 2006; Douglas et al. 2012; Hallmann et al. 2014). The results of this study indicated that cover crops may have positive effects on the abundance of canopy dwelling invertebrates long after the cover crop termination. The study also found a significant negative effect of the interaction term between SSM treatments for ground beetles, which, however, was explained by the population dynamics of this group, which often forms ephemeral clusterings in arable landscapes (C. F. G. Thomas et al. 2002). Due to the high mobility of these insects, thus, it was not possible to attribute the significant association of the treatment interaction with ground beetle abundance to in-field soil management.

Chapter 4 – Sustainable soil management effects on chick food availability Invertebrate abundance is not the sole factor contributing towards chicks survival, but also the availability of a diverse assemblage that allows birds to select invertebrate preys for an optimal nutritional combination (Potts 1986; Borg and Toft 2000). Chapter 4 focused on establishing if SSM was associated with an increase of in-field chick food availability in the experimental fields of Hope Farm. This chapter used field data collected from the experimental fields at Hope Farm to examine the association between cover crop and green waste compost with bird chick food availability for species commonly found in UK's lowland arable landscapes. This motivated the proposal of a new Chick Food Index that attempts to estimate the palatability of invertebrate groups of unknown importance in breeding birds diets, and integrate it with established dietary requirements of farmland birds. Variations in CFI scores in association with SSM were investigated together with measures of SOM content, soil penetrability, and vegetation cover. The results of this study found no evidence of strong patterns of increased aboveground invertebrate chick food availability among the different soil management treatment plots at Hope Farm.

Chapter 5 – Sustainable soil management effects on Skylark territory distribution While the previous chapter showed no variation in resources availability with SSM during breeding season, Chapter 5 focused on how cover crops and compost use may broadly affect farmland birds breeding territory settlement preferences. The Skylark *Auleda arvensis* was used as a model species as it is a ground nesting and feeding species; thus, its behaviour can be directly associated with in-field agricultural management (Donald 2004; Morris et al. 2004; Josefsson et al. 2013). Five years of observational data of Skylark distribution collected by the RSPB at Hope Farm were analysed to assess if cover crop and green waste compost amendment had an effect on the distribution of Skylark territories. To account for pre-existing patters of territory distribution on the farm, territory number was adjusted for disturbance elements in the landscape by considering the amount of suitable area of fields and treatment plots based on Skylark territory distribution in the seven years preceding the SSM trial. This study found no evidence of an effect of green waste compost and cover crop on Skylark territory number, suggesting that these SSM practices do not play a role in breeding habitat selection.

Chapter 6 – Conclusions Finally, this section of the thesis draws together the results shown in the previous chapters to identify their wider implications and suggest future research directions.

1.5.1 Proposed methodologies

While all chapters present novel results which contribute towards a better understanding of the potential of SSM use in arable landscapes to support greater invertebrate and farmland bird abundance, Chapter 4 and Chapter 5 also introduce novel methodologies which may be applied in future agroecological research.

Chapter 4 introduced a method to compute an invertebrate Chick Food Index which allows to take into account all invertebrates observed in field studies as potential resources to farmland bird chicks within impoverished arable landscapes (Conrad et al. 2006; K. L. Evans et al. 2007; Shortall et al. 2009), while taking into account existing information on bird species breeding dietary requirements. This index was proposed, as current estimates of chick food availability in agroecology are based on selected dietary studies derived from faecal analyses. These studies can be biased against highly digestible preys (Moreby and Stoate 2001; Robinson et al. 2018) and towards the invertebrate community available in the site studied. Estimating food availability exclusively from these studies, thus, implicates discarding taxonomic groups that may be available to birds in arable landscapes.

The dietary requirement table used for the development of the CFI was extracted from the unpublished report by Gibbons and Dawson (2004), which compiles the known diet of forty species commonly found in UK's lowland farmland. Although slightly more recent published reviews exist (e.g. Holland et al. 2006), the review by Gibbons and Dawson is the most extensive available to date. There have been a few dietary studies of selected bird species (e.g. Gilroy et al. 2009; D. E. Anderson 2014; McHugh et al. 2016), which could be integrated in the existing review. These, however, were conducted on species for which some level of knowledge of breeding dietary requirements already existed.

The physical palatability trait scoring was based on a food selection study on grey partridge *Predix perdix* chicks by Moreby et al. (2006), which ranked the palatability of colour, size, and speed of preys. In order to score the ranked physical traits, assumptions had to be drawn on the relationship between such traits and their palatability. No studies are available to date that quantify the palatability of different invertebrate traits to birds, and the CFI proposed in this thesis would be improved if it relied on paired selection trials with quantitative measurement rather than on assumptions. Given the difficulties associated with feeding birds a known diet, and the ability of the CFI to capture the varied morphology of invertebrates in the fields and to successfully integrate physical trait scores with dietary requirements, the CFI may be considered an improvement over existing methods.

Chapter 5 proposed a methodology for automatic territory delimitation and count based on territory mapping rules (Bibby et al. 2000), and for calculating suitable area at the field scale. These two methods are both independent of a specific analysis software and they may be applied in future analyses of settlement patterns of territorial, socially monogamous farmland birds. Their combined advantage is that they may allow to eliminate interpretation error when assigning individual observations to territories and increase within and between-sites comparability, and to account for known drivers of farmland bird territory settlement in the landscape.

Automating the territory delimitation process allows users to apply territory mapping rules consistently, independently of the specific map or year being analysed. Traditional territory mapping in fact can be subject to interpretation error because of the difficulties in maintaining consistent standards when interpreting different bird distribution maps (Best 1975). Suitable area was calculated by subtracting from total field area buffers of varying widths around known disturbance elements in the landscape *via* an optimisation process. The suitable area calculation requires information on known disturbance elements in the landscape for territory settlement of the species studied. This may exclude species with lesser known ecology; the Skylark is indeed a well-studied species, but so are many other farmland birds in the UK. For example, corn buntings are known to prefer to settle in the vicinity of isolated trees (Fischer and Schöps 1997), avoid forest edges and urban patches (Lilleør 2007), while their nests can be found up to 20 m from roads (Meyer et al. 2007). Finally, suitable area was calculated without including disturbance elements cause by biological interactions, such as interspecific competition and predation, which are can affect bird territory settlement (Donald 2004). Extension of the method to include these disturbance elements could be proposed by future studies.

Chapter 2

Effects of sustainable soil management on belowground invertebrates

2.1 Introduction

The role that soil organisms can play in intensive arable landscapes has historically received little attention in the literature, because in agricultural systems human inputs have replaced the natural processes that regulate soil structure, nutrient cycles, and pathogen control. Soil meso and macrofauna, however, play a fundamental role in the chemical and physical functioning of natural, semi-natural, and intensive agricultural soil systems (Barrios 2007). They reduce soil organic matter (SOM) particle size and facilitate decomposition by soil bacteria and fungi, thus aiding organic matter and nutrients mineralisation. They also support SOM dynamics when redistributing it vertically by constructing burrows and mounds in the soil profile and through the movement of ingested soil, a process known as bioturbation (Wilkinson et al. 2009). Bioturbation modifies soil structure, increasing porosity and water holding capacity (Siddiky et al. 2012). The regulation of soil functioning is also driven by bottom-up effects of plant cover on soil fauna (Wardle et al. 2004a). Vegetation cover diversity and composition have been shown to modify soil meso and macrofauna by directly affecting the quantity and quality of resources available in the food web to detritivorous and herbivorous invertebrates (Wardle et al. 2006; Crotty et al. 2014; Crotty et al. 2015; Tiemann et al. 2015). Thus, the impoverishment of plant cover and organic matter content contributes to the long-term decline in the number of invertebrates in intensive arable systems (Tiemann et al. 2015), and this reduced faunal activity, in turn, may hinder lowered nutrient release and primary production.

Sustainable soil management is implemented to improve soil self-regulating capacity in agricultural systems to ultimately increase soil fertility and primary production in the field (Garratt et al. 2018). Both cover crops and green compost amendment have the potential to benefit soil meso and macrofauna, as they affect SOC content and soil structure. The potential increase in soil meso and macrofauna with sustainable soil management could aid farmland bird populations which rely on the provision of invertebrates during breeding season.

Aim Although soils have been widely studied and classified in their physical and chemical characteristics, we are still lacking knowledge of soil biota (Barrios 2007), and the understanding of sustainable soil management reflects this disparity, so that less attention has been dedicated to its effects on soil meso and macrofauna (Schipanski et al. 2014; Shackelford et al. 2019). The aim of this chapter was to investigate how cover crop and green compost amendment may affect soil meso and macrofauna abundance and quality in the field, and ultimately support larger invertebrate populations available to farmland birds in arable landscapes during breeding season.

2.2 Methods

2.2.1 Study sites

This study took place at Hope Farm and at four conventional farms located in Cambridgeshire and Lincolnshire, which were included to asses if invertebrate abundance at Hope Farm was of comparable magnitude to that of farms not managed explicitly for wildlife benefits (Field et al. 2016). Arable fields dividend into plots undergoing different soil management were sampled and denoted A–H. Fields were planted with spring barley (*Hordeum vulgare*) or winter wheat (*Triricum aestivum*), with application of mineral fertilisation and chemical input following the UK standard scheme management for farmers (AHDB 2017). The fields were all located on three of the most common clayey-loamy soil types under arable landcover in England (Tab.A.1).

The sampling occurred in fields divided into plots, with each plot corresponding to a soil management treatment combination of cover crop (present or absent) and green waste compost (present or absent). A schematic representation of the plots is shown in Fig.2.1. Fields A–D were the three split-plot replicate experimental fields described in §1.4. These fields were divided in four quadrants, each corresponding to a soil treatment option. One of these fields was sampled twice, once in 2017 (22-27 May) and once in 2018 (21-26 May). These re-sampling was treated separately in the study and denoted as A

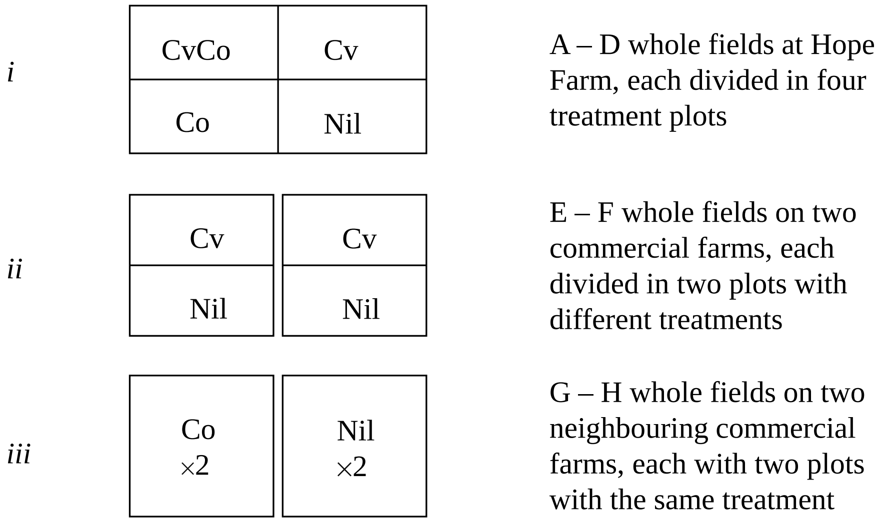

Figure 2.1: Scheme of treatment plots sampled in this study corresponding to each field code letter. CvCo = cover crop and compost; Cv = cover crop only; Co = compost only; Nil = no SSM. Details in table A.1.

and B, respectively. Although A and B are not independent, as the soil treatment plots were spatially constrained, the randomisation of transect placement was possible between the two years. Samples from A and B were both included in this study as they present mesofauna communities differently affected by weather conditions in the months preceding the sampling. Spring 2017 was warm and dry, while the spring of 2018 was characterised by higher precipitations and lower temperatures, as shown in Fig.A.1.

Fields E–H were located on four commercial farms not managed by the RSPB and were sampled between 28 May and 8 June 2018. Fields E and F each had a paired-plot design of conventional rotation and rotation with cover crop (*Phacelia*/black oat/radish or mustard), while G and H were two paired-plot fields on neighbouring farms, G using green compost amendment in both plots and H not using sustainable soil management. All information on the fields is summarised in Tab.A.1.

2.2.2 Sampling design

The field sampled were undergoing different soil treatments, with a treatment being defined as one combination of cover crop (present or absent) and green compost amendment (present or absent). Within each treatment, sampling occurred along three randomly placed transect projecting 32 m from the crop edge towards the field centre (2.2). Along each transects, samples were collected from three locations: in the field boundary, and in the crop field at 4 m and 32 m towards the field centre from the field edge. The boundary was defined as a physical feature that is an interface between the field and

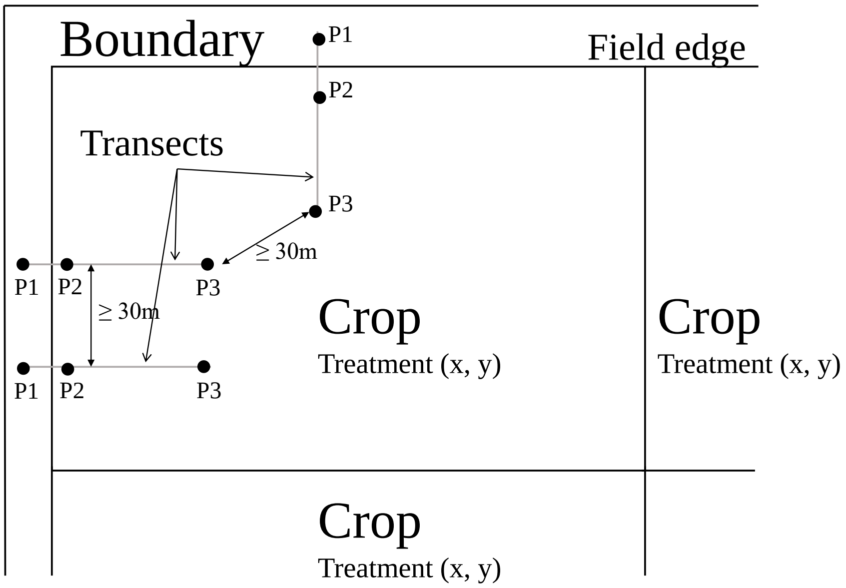

Figure 2.2: Layout of the invertebrate, soil, and vegetation sampling of treatment plot of variable total area. Three transects extended 32 m into the crop, with sample locations 2 m from the field edge in the field boundary (P1), and at 4 m (P2) and 32 m (P3) in the field. The distance between transects was variable, as they were randomly located, but was no less than 30 m apart. Invertebrate samples with suction and pitfall trapping, 100 cm^3 soil cores, and vegetation cover measures were taken at each sampling location. (this is a provisional scheme that needs improvement)

another land cover type. The transects did not extend further into the fields, as it has been shown that biodiversity declines rapidly from field boundaries, and that there is no significant difference between measures detected at 32 m and at the field centre (Marshall 1989; Critchley and Fowbert 2000). The total sampling points were 216 samples, of which 72 in the field boundaries and 144 in the fields. The number of samples per each field and treatment plot is shown in Tab.A.1. Measures of micro-habitat characteristics and invertebrate abundance and diversity were taken at each sampling point.

2.2.3 Soil sampling

Penetrability Soil static penetrability is the measure of ease at which an instrument (i.e. penetrometer) can be pushed into the soil (Bradford 1986), and is used as a measure of soil compaction and as a proxy for soil water content (Green 1988). Higher measurement of penetrability are associated with higher compaction and drier soil. Cone resistance, the resistance to penetration developed by a cone of known basal area, was measured at each sampling location with an Eijkelkamp hand penetrometer (EijkelkampAgrisearch Equipment, NL) with an average measurement error of ±8%. Cone penetrability was estimated as the average of three consecutive measures taken 50 cm apart on undisturbed soil.

Bulk density and water content Soil cores were taken to obtain measures of bulk density, water content, and organic matter content. At each sampling location, two soil cores of 100 cc each were collected from the surface to 5 cm depth using an Eijkelkamp double cylinder soil corer with closed ring holder (Eijkelkamp Agrisearch Equipment, NL). The soil corer was hand-driven when the soil conditions allowed it, and hammer-driven otherwise. The outer cylinder of the soil corer absorbed the impact of the hammer, reducing surrounding soil compaction, while the inner ring preserved an intact sample also under soil dry conditions. The two samples were taken 50 cm apart from undisturbed soil after brushing the soil top layer to remove plant residues. The extremities of the sample protruding from the inner ring were cut to obtain samples of consistent volume throughout. Samples were kept in labelled resalable plastic bags and weighed in the field with a balance accurate to 0.001 g. The gross weight of sample and bag was recorded. Samples were immediately stored at 4°C until further analysis. Bags were weighed at a later stage and subtracted from gross wet weight to record net wet weight ($\mathrm{m}_{\mathrm{wet}}$). Samples were air dried at 40 °C for 24 h and weighed again to obtain measures of moisture content relative to fresh weight. One of the two samples from each point was then oven dried at 105 °C for 24 h and placed in a desiccator for 1 h to cool. Oven dry mass ($\mathrm{m}_{\mathrm{dry}}$) was used to measure bulk density and gravimetric water content as shown in 2.1.

$$\text{Bulk density (g/cm)}^3 = \frac{\mathrm{m_{dry}}}{\text{Sample volume}}$$

$$\text{Water content (g)} = \frac{\mathrm{m_{wet}} - \mathrm{m_{dry}}}{\text{dry}} \tag{2.1}$$

Soil organic matter content Oven dry soil samples were ground to < 2 mm using a mechanical soil grinder (model H-4199, Humboldt Mfg. Co., USA) and homogenised. Average soil organic matter content was determined by loss on ignition of three replicates per sample. Approximately 2 g of soil were heated at 550 °C for 16 h. Standard soil was also ignited to ensure that the temperature of the furnace was homogenous. Loss on ignition was calculated as a percentage of the original weight using the weight loss between 105 °C and 550 °C and averaged across the three replicates.

2.2.4 Invertebrate sampling

Belowground invertebrate samples were taken at each location from a 15 cm depth core sample (300 cc) collected using an Eijkelkamp double cylinder soil corer with closed ring holder (Eijkelkamp Agrisearch Equipment, NL). The samples were kept in labelled resealable plastic bags and refrigerated at 4 °C to protect the invertebrates from thermal shock up to a maximum of 48 hours until processing.

Invertebrates were extracted from the cores into 70% ethanol solution using Berlese-Tullgren funnels of 20 cm in diameter and equipped with a 4 mm mesh. The samples were run over 48 hours to allow the complete extraction of invertebrates from the samples. The behavioural extraction methods of Belerles-Tullgren funnel is one of the traditional methods to effectively and inexpensively extract soil arthropods without contaminating the samples (Crossley and Blair 1991; Sakchoowong et al. 2007). The apparatus works by creating warm and dry conditions through a incandescent lightning source placed 25 cm above the soil sample, driving the soil and litter dwelling invertebrates to move away from the inhospitable conditions towards the bottom of the sample, where a funnel forces them into a collection bottle (Fig. 2.3).

Invertebrate abundance and Soil Biological Quality arthropod index

Litter and soil dwelling invertebrates found in the soil cores were counted and identified to Order or Class level following Tilling (1987). Total abundance and abundance of invertebrates palatable to birds were recorded, with the latter being a subset of the invertebrate

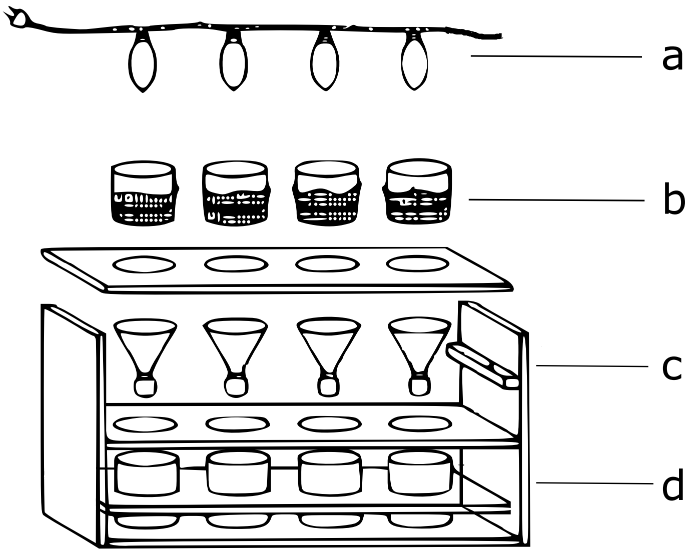

Figure 2.3: Schematic representation of a Belerles-Tullgren funnel set used for invertebrate extraction. Adapted from Crossley and Blair (1991). A = light bar, B = sleeves with 4 mm mesh, C = funnels, D = collection bottles.

community as a whole (Tab.2.1). Invertebrates were also classified following the Soil Biological Quality arthropod index (QBS-ar, Tab. 2.1) developed by Parisi (2001) and Parisi et al. (2005), that has been widely used in the agroecological assessment of soil quality in the past 15 years (Firbank et al. 2017; Menta et al. 2018). Based on this index, a "good quality" soil is stable, porous, and high in organic matter content and biodiversity. Its metric is based on the concept that higher quality soils will present a higher abundance of arthropods morphologically well adapted to the soil. These morphological adaptations comprise the loss of long antennae and hairs, of body pigmentation and visual apparatuses, and of running, flight, and jumping adaptations. Thus, the QBS methods requires to assign to morpho-types most specialized to the soil (e.g. Diplura) a high eco-morphological score (EMI) than to morpho-types not well adapted to the soil (e.g. Thysanoptera). The sum of all EMI scores in a sample gives the QBS-ar score of that sample.

2.2.5 Statistical analysis

The effects of soil treatments on soil habitat covariates and on invertebrate abundance and QBS-ar scores were investigated using generalised linear mixed models. Alternative model sets were formulated for the mesofauna response variables based on different hypotheses and multi-model inference was used in determining the model best fit in describing the observed data. All analysis was conducted in R version 3.6.0 (Fox and Weisberg 2019).

Table 2.1: Eco-morphologic indices (EMIs) for the computation of the QBS-ar soil biodiversity index of arthropod groups found in the study, adapted from Parisi et al. (2005). The sum of all EMI scores in a sample gives the QBS-ar score of that sample. Asterisks indicate taxonomic groups considered palatable to birds.

Group		EMI score	
Collembola	Epigeous: complex pigmentation	1	*
	Epigeous: small size, modest pigmentation	4	*
	Hemi-edaphic: pigmentation present, reduced appendages, visual apparatus developed	6	
	Hemi-edaphics: pigmentation present, furca reduced, reduced appendages	8	
	Eu-edaphic: no pigmentation	10	
	Eu-edaphic: no pigmentation, no furca	20	
Diplura		20	
Pauropoda		20	
Protura		20	
Symphyla		20	
Pseudoscorpiones		20	
Mites		20	
Araneae	> 5 mm	1	*
	< 5 mm, scarse pigmentation	5	
Opiliones		10	
Chilopoda	> 5 mm, developed legs	10	*
	Geophilomorpha	20	
Isopoda		10	
Coleoptera	Larvae	10	*
	Epigeous forms	1	*
	< 2 mm	5	
	Thin, tan-brown tegument	6	
Diptera	Larvae	10	*
	Adults	1	
Hymenoptera	Larvae	10	*
Hymenoptera	Adults	1	
	Formicidae	5	
Lepidoptera		10	*
Psocoptera		1	
Thysanoptera		1	
Other Holometabola	Larvae	10	*
	Adults	1	
Earthworms			*

2.2.5.1 Soil characteristics, biological quality, and invertebrates abundance in the fields and boundaries

The effects of soil management on soil habitat characteristics, and the changes in habitat covariates along the sampling transects were investigated using linear mixed models in the **lme4** package (Bates et al. 2014), with *field code* (A-H) as random intercept. For categorical variables containing more than two levels significant results were further investigated with *post hoc* comparisons of each pair of levels with Tukey's tests using the package **multcomp** (Hothorn et al. 2016). Tukey's tests correct for multiple comparisons thus avoiding additional use of Bonferroni corrections (Ruxton and Beauchamp 2008). GLMMs assumptions were checked with residuals plotting and assumptions are met.

2.2.5.2 Effects of soil treatments and habitat covariates on QBS-ar and invertebrate abundance

Generalized linear mixed models (GLMMs) with Poisson error distributions were used to investigate the effect of soil treatment and habitat covariates on in-field Soil Biological Quality (QBS-ar) and mesofauna abundance. Soil samples QBS-ar scores, total abundance and abundance of invertebrate palatable to birds were taken into account. These response variables were individually modelled using the package **lme4** (Bates et al. 2014).

The candidate model set for QBS-ar measures and invertebrate abundance measures included Cover crop (present/absent), Compost amendment (present/absent), Year (2017 and 2018), Point (4 m and 32 m), and crop type (winter wheat/spring barley) as fixed categorical effects. Penetrability, water and organic matter content, bulk density, and the response variables' measures (i.e. total abundance, abundance of invertebrates palatable to birds, and QBS-ar) as measured in the boundary were modelled as covariates (Tab. 2.2). All models included *field code* (2.2.1) as a random intercept effect to allow for non-independence of multiple samples per site. Moreover, an *observation level* random effect was included when needed to account for model overdispersion, measured by comparing the sum of squared Pearson residuals with the residual degrees of freedom of the full model following Bolker et al. (2009). The assumptions of Poisson error models require that the residual deviance of a candidate model is roughly equal to the residual degrees of freedom, and when the ratio of these parameters (i.e. the dispersion parameter) is > 1, the data is deemed overdispersed. The addition of an observation level random effect has been shown to be a robust method to model overdispersed biological data when using Laplace approximation in GLMM parameter estimation (Elston et al. 2001; Harrison 2014).

A null model model of no variation of the dependent variable was included in the

Table 2.2: Model set examining the effect of soil treatment and habitat covariates on QBS-ar and invertebrate abundance. All models with the exception of the intercept-only model also included the fixed factor Year to account for changes occurring between samples from 2017 and 2018.

N	Model description	Model notation
1	Intercept only	β_0
2	Cover + Compost	$\beta_0 + \beta_1 X_{1i} + \beta_2 X_{2i}$
3	Cover ∗ Compost	$\beta_0 + \beta_1 X_{1i} * \beta_2 X_{2i} + \beta_{1i,2}(X_{1i} * X_2)$
4	Cover + Compost + Crop	$\beta_0 + \beta_1 X_{1i} + \beta_2 X_{2i} + \beta_3 X_3$
5	Cover + Compost + Point	$\beta_0 + \beta_1 X_{1i} + \beta_2 X_{2i} + \beta_4 X_4$
6	Cover + Compost + Point + BY	$\beta_0 + \beta_1 X_{1i} + \beta_2 X_{2i} + \beta_4 X_4 + \beta_5 X_5$
7	Cover + Compost + W + CR	$\beta_0 + \beta_1 X_{1i} + \beta_2 X_{2i} + \beta_6 X_{6i} + \beta_7 X_{7i}$
8	Cover + Compost + Point + CR + BD	$\beta_0 + \beta_1 X_{1i} + \beta_2 X_{2i} + \beta_3 X_{3i} + \beta_7 X_{7i} + \beta_8 X_{8i}$
9	Cover + Compost + CR + BD + LOI	$\beta_0 + \beta_1 X_{1i} + \beta_2 X_{2i} + \beta_7 X_{7i} + \beta_8 X_{8i} + \beta_{10} X_{10i}$
10	Cover + Compost + Crop + Point + BY + W + CR + BD + LOI	$\beta_0 + \beta_1 X_{1i} + \beta_2 X_{2i} + \beta_3 X_{3i} + \beta_4 X_{4i} + \beta_5 X_{5i} + \beta_6 X_{6i} + \beta_7 X_{7i} + \beta_8 X_{8i} + \beta_9 X_{9i} + \beta_{10} X_{10i}$

BD = bulk density; BY = response variable measure in the boundary; CR = soil penetrability; LOI = soil organic content; W = gravimetric water content.

analysis, as model selection through information criteria is restricted to the *a priori* defined model set, and could thus identify a "best" model that may explain very little variation in the dataset (Dochtermann and Jenkins 2011). If the null model was supported to a similar degree to the other candidate models, none of the candidate models was deemed appropriate.

Model fits were compared by calculating the corrected Akaike (AIC_c) information statistic and ΔAIC_c (i.e. a measure of each model relative to the best model) with the **AICcmodavg** package (Mazerolle 2019). Models with the lowest AIC_c values were considered to have the most empirical support, and, although models in the 2-7 range have some support (Burnham et al. 2011), only models with ΔAIC_c between 0 and 2 were deemed to have substantial support. Model averaging was conducted on models with $\Delta AIC_c \leq 2$. The sum of weights of predictor variables, often employed by ecologists to determine the importance of given predictors by summing the the weights of averaged models, was not calculated in this study, as it has been shown to be a poor estimate of variables' importance (Galipaud et al. 2014).

Models were checked for any obvious deviations from homoscedasticity and normality using plotting functions in the **DHARMa** package (Hartig 2019). All models meet the assumptions for general linear mixed modelling testing.

2.3 Results

2.3.1 Soil habitat characteristics in the fields

With the notable exception of soil organic matter content, which was positively affected by compost amendment, soil treatment did not have an effect on the habitat variables measured in this study (Tab. 2.3). Soil organic content was significantly higher in plots with green compost amendment (mean = 9.60%, SD = ±2.21) than without (mean = 8.32%, SD = ±1.58, $\beta = 1.32$, $t_{133} = 3.77$, $p < 0.001$, Fig. 2.5). Penetrability, organic matter, and water content were markedly different between the field boundaries and the crop fields (2.4), while bulk density remained more homogeneous across the sampling points. Surface soils in the field boundaries, despite having higher water content, were significantly harder than in the arable fields. As soil penetrability can be used as an indirect measure of soil moisture, with wet soils less resistant than dry soils (Green 1988), a positive relationship between these two variables appears contradictory. Penetrability, however, may have been affected by the dense and complex root system of boundary vegetation compared to the sparse roots of the crop in its early stages in the field. Organic content was significantly predicted by sampling point (β = -0.76, t_{203} = -5.97, $p < 0.001$):

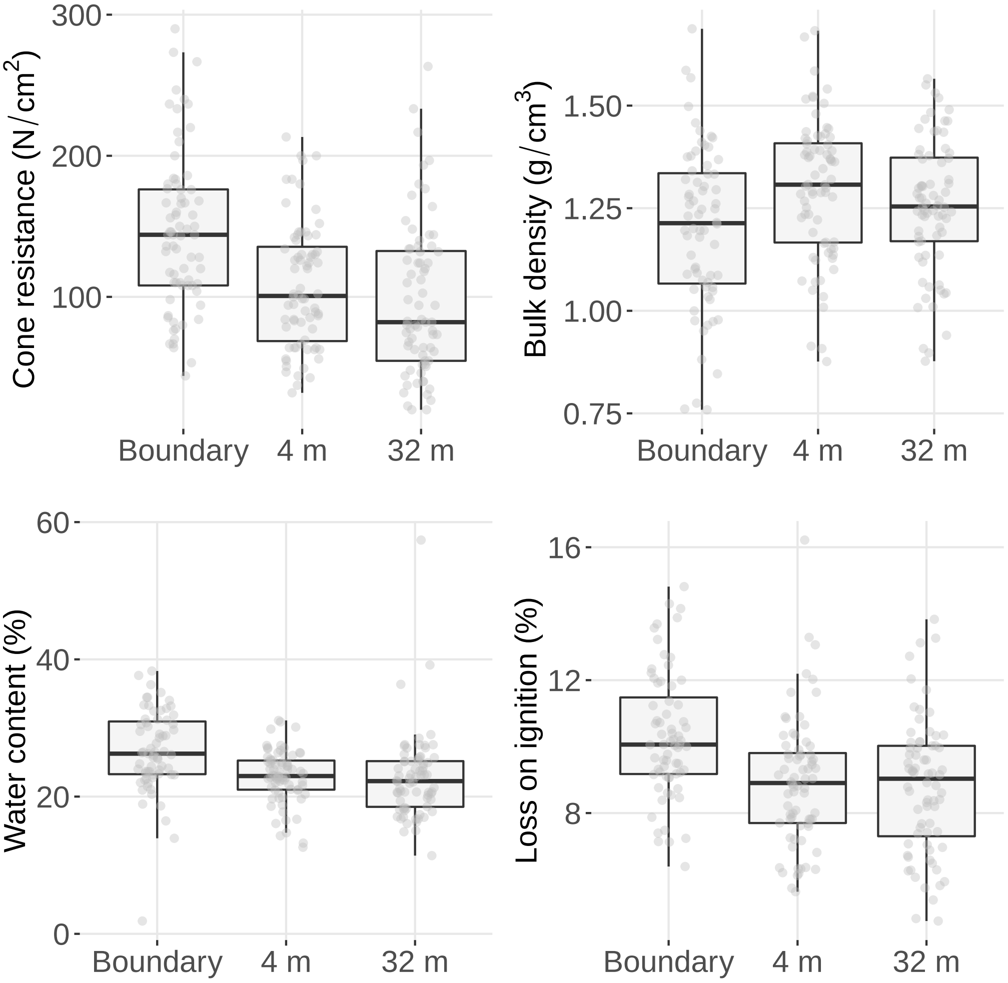

Figure 2.4: Soil habitat characteristics along the transects. Soil samples taken in the boundaries had higher penetrability, water and organic matter content than in-field points. In-field habitat characteristics did not differ significantly between points at 4 m and 32 m from the field edge.

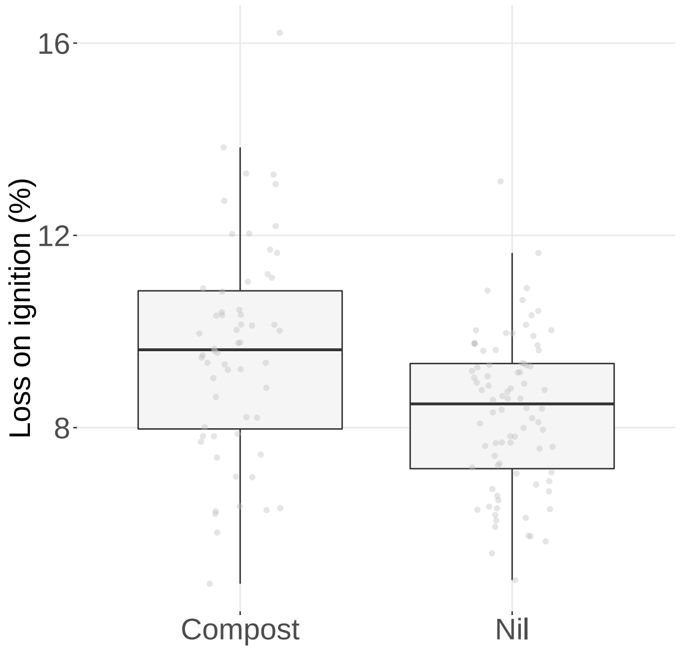

Figure 2.5: Organic matter content measured as loss on ignition in plots with and without compost soil amendment.

the average soil organic content in boundary samples (10.69%, ±3.31) was higher than at 4 m (β = -1.4, z = -5.64, P < 0.001) and 32 m (β = -1.5 , z = -6.15, P < 0.001) from the field edge. In-field organic matter measures did not differ along the transects. In-field average penetrability was 102.3 N/cm^2 (SD = ±49.4), while in the boundary was 143.5 N/cm^2 (SD = ±56.4). Sampling point along the transect was a significant predictor of penetrability (β = <-24.14, t_{207} = -7.64, $p < 0.001$). Higher penetrability measures were recorded in the boundaries than in the fields at 4 m (β = -38.1, z=-6.11, P < 0.001) and 32 m (β = -48.3, z=-7,74, P < 0.001) from the field edge, while there were no significant in-field differences. Gravimetric water content changed along the transects (β = -1.87, t_{202} = -4.43, $p < 0.001$) and was higher in boundary (26.63%, SD = ±6.01), than in in-field samples (22.83%, ±5.17) at 4 m (β = -3.6, z=-4.32, P < 0.001), and at 32 m (β = -3.7, z=-4.54, P < 0.001) from the field edge. In-field water content did not significantly differ. Bulk density did not change significantly among transect sampling point (β = 0.003, t_{203} = 1.72, p = 0.087). The average bulk density recorded in the fields was 1.28 g/cm_3 (± 0.17), and appeared to be generally higher close to the field edge than further into the fields.

2.3.2 Soil meso and macrofauna

A total of 4,416 invertebrates from the 20 taxonomic groups shown in Tab.2.1 were collected, identified and counted. Of these, over 50% of individuals were from the Subclass

Table 2.3: Results of generalised linear mixed models of effects of soil treatment on habitat covariates accounting for differences across fields. Only soil organic content significantly increased with compost amendment.

	Bulk density	SE	df	t-value	P
Intercept	1.29	0.04	133	35.679	<0.001
Cover crop	0.00	0.03	133	0.082	0.935
Compost	-0.02	0.04	133	-0.528	0.599
Cover * Compost	0.00	0.05	133	0.010	0.992

	Penetrability	SE	df	t-value	P
Intercept	89.89	14.00	133	6.421	<0.001
Cover crop	6.25	7.72	133	0.810	0.419
Compost	8.73	8.93	133	0.978	0.330
Cover * Compost	-1.54	12.18	133	-0.126	0.900

	Water content	SE	df	t-value	P
Intercept	23.20	1.23	133	18.893	<0.001
Cover crop	-1.11	1.06	133	-1.047	0.297
Compost	0.85	1.20	133	0.713	0.477
Cover * Compost	1.07	1.66	133	0.644	0.520

	Organic content	SE	df	t-value	P
Intercept	7.95	0.61	133	13.094	<0.001
Cover crop	0.06	0.30	133	0.189	0.850
Compost	1.32	0.35	133	3.774	<0.001
Cover * Compost	0.22	0.48	133	0.469	0.640

Collembola (springtails), and 23% from the Subclass Acari (mites). Of the total invertebrate sample, 740 individuals were classified as palatable to birds. Both abundance and QBS-ar scores were higher in field boundaries than in the crop field, and higher towards the field centre than at the field edge (Fig. 2.6). Field boundaries can hold a larger number of ecological niches for mesofauna than arable fields (Mason et al. 2005; Sechi et al. 2017), partly because of the diversification of plant cover, which positively influences soil organisms richness and diversity (Bennett 2010; Crotty et al. 2015).

Abundance significantly changed along the transects (β = -2.36, t_{204} = -2.38, p = 0.018), with soil samples collected in boundaries presenting more invertebrates (mean = 26.1, SD ±13.4) than those collected at 4 m (mean = 14.5, SD ±10.5, β = -11.36, z=-6.14, P $<$ 0.001) and 32 m (mean = 21.0, SD ±13.7, β = -4.86, z=-2.63, P = 0.023) from the field edge (Fig. 2.6). In the fields, abundance was higher at 32 m than at 4 m (β = 6.50, z=3.55, P = 0.001). QBS-ar scores followed the total invertebrate abundance patterns and were also significantly predicted by sampling point along the transects (β = -47.68, t_{204} = -3.78, p $<$ 0.001). QBS-ar was higher in the field boundary (mean = 269, SD ± 176) than at 4 m (mean = 96, SD ±95, β = -170.21, z=-7,30, P $<$ 0.001) and at 32 m (mean = 170, SD ±169, β = -97.16, z=-6.17, P $<$ 0.001). Again, QBS-ar towards the field centre was significantly higher than close to the field edge (β = 73.06, z=3.17, P = 0.004).

Table 2.4: Relative abundance of the most common invertebrate taxa collected in the soil cores.

	Taxa	Proportion (%)
1	Collembola	53.68
2	Acari	22.99
3	Diptera	6.56
4	Coleoptera	5.75
5	Symphyla	3.45
6	Anellida	2.84
7	Other	1.67
8	Hymenoptera	1.42
9	Chilopoda	0.81
10	Protura	0.56
11	Araneae	0.28

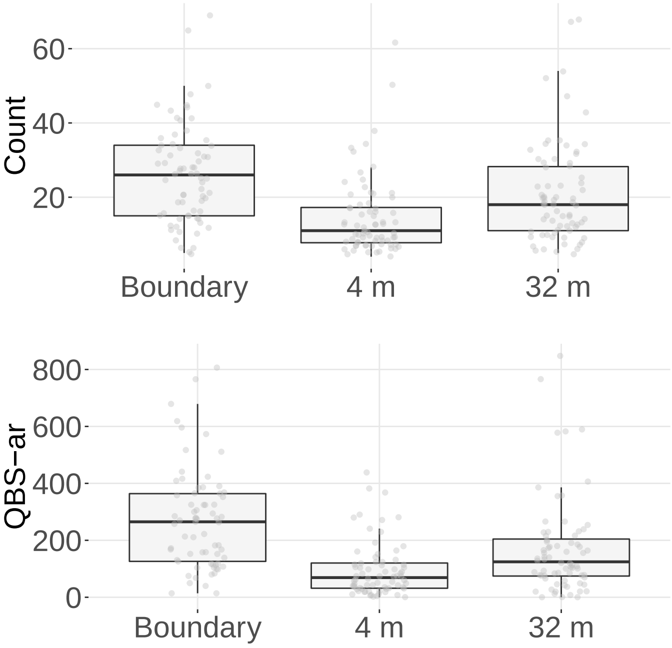

Figure 2.6: Invertebrate total abundance and QBS-ar scores from the soil samples collected in the field boundary, and in the fields at 4 m and 32 m from the field edge. Abundance and QBS-ar were highest in the boundary, and significantly higher at 32 m than at 4 m.

Table 2.5: Comparison of models explaining total abundance of belowground invertebrates. Models are ranked by their AICc and weight. K is the number of parameters in the model, see below for definitions of model terms.

	Model	K	AICc	Delta AICc	AICc weight
5	C+S+P	7	994.09	0.00	0.45
8	C+S+P+CR+BD	9	994.40	0.32	0.39
6	C+S+P+BY	8	996.22	2.13	0.16
10	C+S+P+CT+BY+W+CR+BD+LOI	15	1007.18	13.09	0.00
9	C+S+CR+BD+LOI	9	1011.74	17.66	0.00
2	C+S	6	1012.38	18.30	0.00
7	C+S+W+CR	8	1012.99	18.91	0.00
4	C+S+CT	7	1014.34	20.26	0.00
3	C*S	7	1014.56	20.47	0.00
1	Null	3	1020.15	26.06	0.00

C = cover crop; S = compost amendment; P = sampling point; CT = crop type; BD = bulk density; BY = abundance in the boundary; CR = soil penetrability; LOI = soil organic content; W = gravimetric water content; Null = intercept-only model.

2.3.3 Effects of soil treatments and habitat covariates on meso and macrofauna and QBS-ar

The generalised linear mixed models showed a significant positive response of both in-field total invertebrate abundance and QBS-ar scores to cover crop, while compost addition and increased soil organic matter content did not have any effect. Distance from the field edge was a significant predictor of in-field invertebrates, with points further into the field presenting higher QBS-ar and abundance than points closer to the field edge. Localised measures of penetrability —and to a lesser extent soil bulk density— although not significant, influenced the QBS-ar scores of soil samples, with higher levels of soil compaction showing some detrimental effects on invertebrate community quality. Invertebrate abundance and QBS-ar scores were higher in samples from 2017 (labelled A) than in samples from 2018 (labelled B). The abundance of invertebrates palatable to birds did not appear to vary significantly across the study area.

Comparison of the alternative models showed that those including soil treatments, transect point, penetrability and bulk density were the top-ranked models (ΔAICc ≤ 2) for both total abundance (Tab.2.5) and QBS-ar (Tab.2.6). Two candidate models emerged from the analysis of abundance, thus, their parameter estimates were averaged across them (Tab.2.7); for QBS-ar only one model had ΔAICc ≤ 2 and, therefore, multi-model inference was not used (Tab.2.8). Model predictions showed a strong positive effect ($P = 0.003$) of the use of cover crop on in-field total abundance (Fig. 2.8) and QBS-ar (Fig.2.7). Moreover, samples from points closer to the field centre had higher abundance ($P < 0.001$) and biological quality ($P = 0.003$) than samples collected at 4 m from the field edge. Green compost amendment had no effect on mesofauna abundance ($P = 0.503$) or QBS-ar ($P = 0.113$). Both abundance ($P = 0.009$) and QBS-ar scores ($P = 0.031$) were higher in samples from 2017 (A) than 2018 (B). Model estimates suggested a general negative effect of soil penetrability on the QBS-ar score of soil samples (Fig.2.9). Soil penetrability and bulk density were in the top-ranked models for both response variables, however, their estimates and AIC$_c$ weight suggested that they were mostly relevant in relation to QBS-ar scores. The inclusion of these two habitat characteristics increased the AIC$_c$ weight of the best-fit model of QBS-ar from 0.15 (i.e soil treatments and sampling point alone) to 0.77 (Tab.2.6), and high penetrability measures were associated to lower QBS-ar scores, although this relationship varied across fields (Fig. 2.9). A study on decomposition rates across similar transects (boundary/margin:crop interface/crop) conducted on same soil type as this study showed significantly lower mesofauna abundance closer to the field edge, where large machinery traffic compacts the soil and renders it inhospitable to soil or-

Table 2.6: Comparison of models explaining local values of soil biological quality (QBS-ar). Models are ranked by their AICc and weight. K is the number of parameters in the model, see below for definitions of model terms.

	Model	K	AICc	Delta AICc	AICc weight
8	C+S+P+CR+BD	9	1683.86	0.00	0.73
5	C+S+P	7	1686.63	2.77	0.18
6	C+S+P+BY	8	1688.70	4.83	0.06
9	C+S+CR+BD+LOI	9	1691.84	7.97	0.01
7	C+S+W+CR	8	1693.32	9.46	0.01
10	C+S+P+CT+BY+W+CR+BD+LOI	15	1693.85	9.99	0.00
2	C+S	6	1695.88	12.02	0.00
3	C*S	7	1697.29	13.42	0.00
4	C+S+CT	7	1697.98	14.11	0.00
1	Null	3	1703.64	19.78	0.00

C = cover crop; S = compost amendment; P = sampling point; CT = crop type; BD = bulk density; BY = QBS-ar score in the boundary; CR = soil penetrability; LOI = soil organic content; W = gravimetric water content; Null = intercept-only model.

ganisms (Carlesso et al. 2019). Although soil core extraction explicitly avoided machinery tramlines, and detected no significant difference in penetrability and bulk density at 4 m and 32 m from the field edge (as shown in section 2.3.1), there was a pattern of higher soil compaction closer to the field edge (Fig. 2.4), which may have affected soil biological quality.

Total mesofauna abundance and soil QBS-ar scores in the crop were not influenced by their values in the margin. Although spillover from the boundaries can increase the diversity of larger-bodied soil macrofauna in the field (J. Smith et al. 2008), the abundance of mesofauna in the crop is driven by local habitat conditions caused by management, so that their communities are often isolated from the margins (Frazão et al. 2017; Carlesso et al. 2019). Moreover, soil tillage has been shown to inhibit soil invertebrate spillover from the field boundaries (Médiène et al. 2011).

The best fitting model out of the model set of invertebrate palatable to birds abundance was the intercept only model (A.3), indicating that soil treatment and habitat characteristics were inadequate at describing the distribution of these invertebrates. This result was likely due to the low abundance of invertebrates ascribing to this classification (only

16.7% of the total sample).

Table 2.7: Averaged estimated regression parameters, standard errors, *z*-values and 95% confidence intervals (CI) for variables included in the top-ranked Poisson GLMMs (ΔAICc $\leq$ 2) of in-field total belowground invertebrate abundance. Boldface indicates statistical significance with $p < 0.05$.

	Av. estimate	SE	z-value	CI 95%		*p*
Cover crop	0.25	0.09	3.05	0.08	0.42	**0.002**
Compost	0.06	0.09	0.67	-0.12	0.24	0.503
Year	-0.74	0.27	-2.63	-1.27	-0.22	**0.009**
Point	0.36	0.08	4.31	0.20	0.51	**< 0.001**
Penetrability	-0.29	0.31	-0.95	-0.89	0.31	0.344
Bulk density	-0.44	0.26	-1.69	-0.95	0.07	0.091

Table 2.8: Estimated regression parameters, standard errors, *z*-values and 95% confidence intervals (CI) for variables of the best-fit Poisson GLMMs (ΔAICc $\leq$ 2) of in-field QBS-ar scores. Boldface indicates statistical significance with $p < 0.05$.

	Estimate	SE	z-value	CI 95%		*p*
Cover crop	0.53	0.17	3.03	0.17	0.86	**0.003**
Compost	0.29	0.18	1.58	-0.08	0.65	0.113
Year	-1.01	0.47	-2.16	-1.96	-0.11	**0.031**
Point	-0.48	0.16	3.01	0.18	0.82	**0.003**
Penetrability	-1.16	0.60	-1.92	-2.36	0.02	0.054
Bulk density	-0.87	0.51	-1.68	-1.88	0.14	0.092

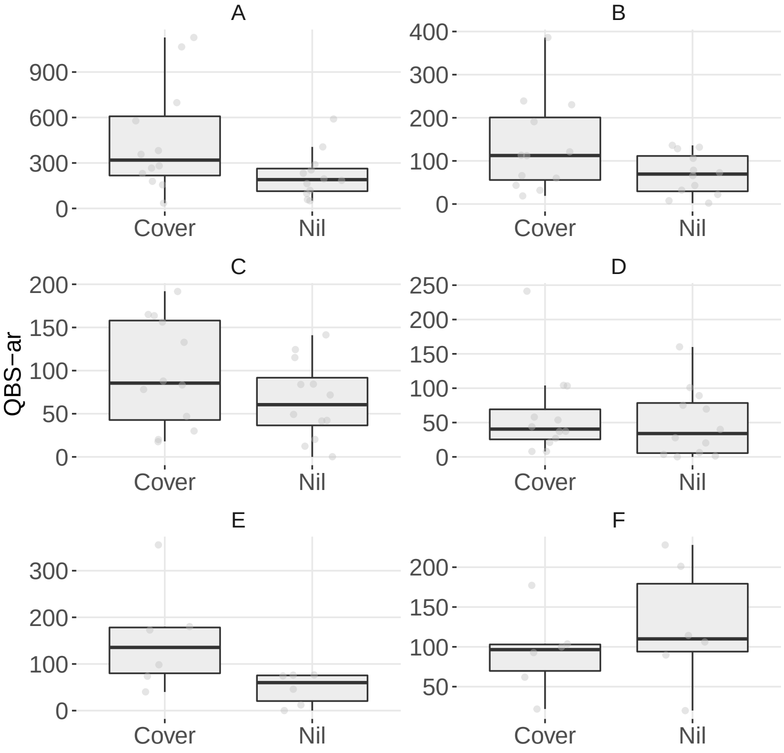

Figure 2.7: QBS-ar scores in plots with and without cover crops.

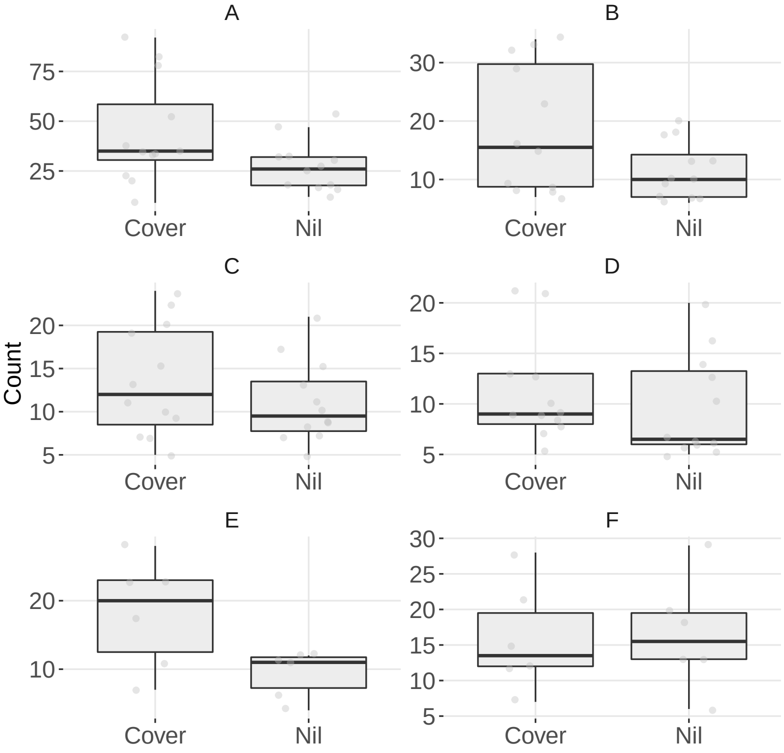

Figure 2.8: Total invertebrate abundance with and without cover crops.

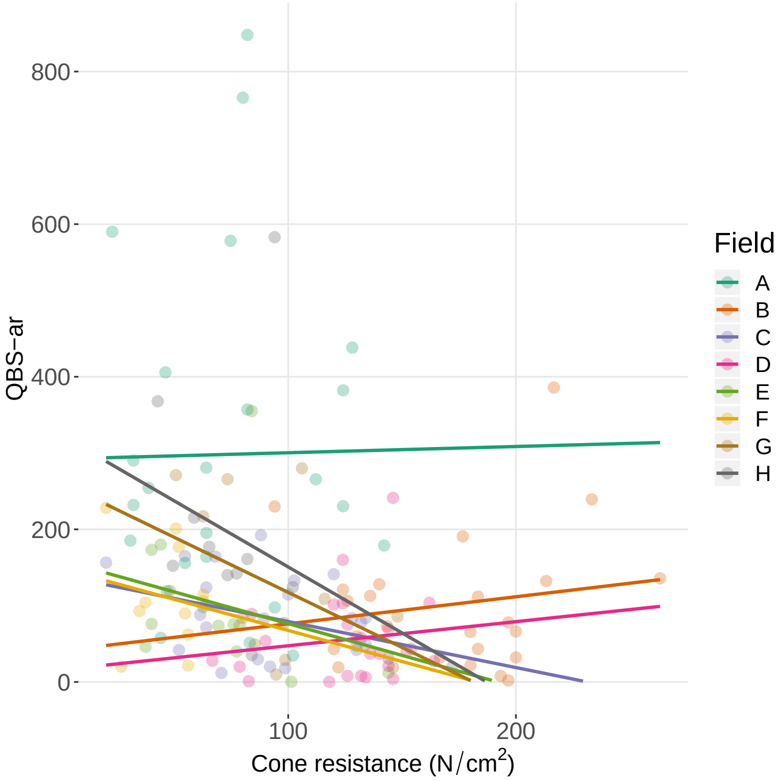

Figure 2.9: QBS-ar scores and soil penetrability (i.e. cone resistance) in each of the study fields. Fields A–D were located at Hope Farm and presented four soil management options (with or without cover crop and compost amendment), E–F presented plots with or without cover crops, and G–H presented plots with or without compost.

2.4 Discussion

Soil meso and macrofauna play a central role in the mineralization and redistribution of organic matter and in the maintenance of soil structure, and provide a source of food for birds in agricultural landscapes. Cover crops and compost can have lasting (i.e. legacy) effects on soil agroecosystems. While we have a better understanding of these effects on nutrient cycling, soil physical properties, and microbial communities, we lack knowledge on their effect on soil meso and macrofauna (Garcıa-Gil et al. 2000; Frasier et al. 2016; Wittwer et al. 2017; Garratt et al. 2018). This study found that cover crop had a positive legacy effect on soil meso and macrofauna, resulting in an increase in total abundance of invertebrates and in soil QBS-ar scores when the cash crop were present in the fields, while green waste compost amendment did not have a significant impact on the invertebrate community.

Soil invertebrate abundance and soil QBS-ar scores were higher in plots with a cover crop mixture in the cash crop rotation long after its removal, even when soil invertebrate communities are not at their peak (Olejniczak 2007). These results show the legacy of cover crop on the invertebrate community, as usually invertebrates are studied while the cover crop is in the field, or shortly after its removal (e.g. Phophi et al. 2017; de Melo et al. 2019). The lasting effects of cover crop on arable soil largely depend on species identity, as different types of cover crops and residues can harbour distinctive invertebrate communities (Gallandt et al. 2005; Crotty et al. 2015). The different cover crop mixes (cereals/brassicas/legumes/phacelia) used in the fields of this study, however, have shown not to alter soil mesofauna composition differently during a UK field trial (e.g. Crotty and Stoate 2019), and abundance and QBS-ar scores in this study were of comparable magnitude across samples of the same year.

Incorporation of cover crop residues in the soil has been shown to have positive cascading effects on mites and springtails populations in the detritivores food web by increasing SOM content (Garrett et al. 2001; Miyazawa et al. 2002; Kautz et al. 2006). In this study, cover crops were found not to increase total SOC in the soil, however, different cover crop species may affect litter carbon differently depending on their decomposability and lignin content (Frasier et al. 2016), and contribute differently to distinct soil organic matter fractions (Duval et al. 2016), an effect that would not emerge from loss on ignition measurements. The higher abundance and QBS-ar scores found in cover crop plots suggest that the amount of readily available decomposable material in the soil was higher in these plots, and that dominant mesofauna taxa morphologically well adapted to the soil (i.e. mites and springtails) followed its distribution.

Green compost amendment in the fields did not have an effects on invertebrate abundance or soil QBS-ar scores (A.2, A.3), even though it significantly increased soil organic content in the fields. A UK study evaluating the effects of 3 years of green/food waste compost on arable land also found that it was very successful at increasing SOC in this relatively short period of time. Compost, however, was found not to provide large quantities of readily decomposable organic matter, a characteristics suggested by its failure to increase soil microbial biomass within that time frame (Bhogal et al. 2018). Additionally, two UK studies found that earthworms (*Lumbricus terrestris*) density was not affected after two and a half years and three years of green waste compost amendment in arable experimental plots (Leroy et al. 2008; Stroud et al. 2016). The results of this chapter align with the findings of these studies, showing that green waste compost does not successfully increase soil meso and marcofauna, potentially due to an insufficient provision of readily available organic matter to the soil biota.

While most fields were mostly unaffected by compost, field C showed a negative effect of this soil treatment on invertebrate abundance and QBS-ar scores caused by a 60% drop in the number of Collembola (A.4), suggesting that compost may have had a localised negative effect on these invertebrates. This local detrimental effect could be explained by the heterogeneity of green waste compost piles caused by the feedstock material used to produce it. Green compost feedstock quality can vary with the level of urbanisation of the area (Weber et al. 2007), season of waste collection, and compost maturation time (Hepperly et al. 2009). While there is no knowledge on the effects of green compost heterogeneity on invertebrate populations in arable soils, a limited number of studies have highlighted how this variation can affect widely differently soil biota when applying biochar (Lehmann et al. 2011; Ameloot et al. 2013; Domene 2016), a soil amendment product of the anaerobic thermal degradation of organic materials also used to improve soil fertility and increase soil carbon storage.

2.5 Conclusions

Soil meso and macrofauna contribute to the physical and chemical health of agricultural soils by aiding organic matter and nutrients mineralisation and redistribution, and modifying soil structure. The potential effects of increased plant diversity and SOM abundance in arable soils obtained with introducing cover crops or green waste compost amendment in impoverished arable soils, however, has received little attention in the literature. This study showed lasting effects of cover crops on the abundance of belowground invertebrates and the QBS-ar scores of soil long after the cover crops' removal from the field. Green compost, instead, did not show an association with belowground meso and macrofauna.

Chapter 3

Effects of sustainable soil management on aboveground invertebrates

3.1 Introduction

Modern conventional intensive arable systems have been associated with pollution and water eutrophication, biodiversity declines, and soil degradation (Stoate et al. 2001; Tilman et al. 2001; Newbold et al. 2015). Concerns raised about their long-term sustainability have seen the development of practices aimed at improving agricultural landscapes and reducing the use of synthetic chemical inputs, such as agri-environment schemes and organic amendments (Bommarco et al. 2013). At the field level, soil management is key in supporting sustainable agricultural systems by enhancing nutrient availability, the growth and yield of crops, and all biological processes associated to high soil organic matter retention (Kremen and Miles 2012; Garratt et al. 2018).

Terrestrial invertebrates are important drivers of healthy ecosystems, as they affect the quantity, quality, and timing of nutrients cycling with direct (i.e. biomass input) and indirect (i.e. food web dynamics) pathways (Yang and Gratton 2014). In agricultural systems, they are largely responsible for pollination (Klein et al. 2007), and contribute to organic matter decomposition, soil structure improvement, and water transfer (Barrios 2007; Bottinelli et al. 2015). Ground beetles and other predator taxa are also an important part of natural pest control (M. B. Thomas et al. 1992), with cascading effects throughout the ecosystem (Hawlena and Schmitz 2010). Last, but not least, invertebrates are an essential dietary component for most farmland birds, particularly during breeding season, when they provide the protein necessary to chicks for growth and resistance to chilling

(Southwood and Cross 2002). Birds, thus, fail to fledge their chicks in arable landscapes impoverished of invertebrates (Potts 1986; Brickle et al. 2000; Boatman et al. 2004; Hallmann et al. 2014). The farmland bird populations decline since the 1980s (EBCC 2018) has occurred in parallel to the crash in invertebrate populations throughout Europe and the UK widely considered to be driven by habitat loss and agrochemical use in conventional agricultural landscapes (Conrad et al. 2006; Shortall et al. 2009; Potts et al. 2010; Hallmann et al. 2017).

Sustainable soil management practices, such as cover crops and green waste compost incorporation, can increase the organic matter content available in the soil to offset the use of chemical fertilisers and promote soil health (Hargreaves et al. 2008; Wittwer et al. 2017). Cover crops are grown in the field during off-season also to reduce soil erosion, aid weeds and pathogens control (Kremen and Miles 2012; Blubaugh et al. 2016), and promote nutrient management (Tonitto et al. 2006; Hallama et al. 2019). Green waste compost is a soil amendment derived from municipal botanical waste used to increase soil organic matter content, pH, and to improve soil structure (Celik et al. 2004).

In a healthy soil system, there are strong links between aboveground and belowground biological communities (Wardle et al. 2004a). In agricultural systems, optimal soil structure can favour emergence of invertebrates (Frampton et al. 2000; Felici et al. 2019), and soil organic matter content positively affects detritivores abundance and diversity (Vreeken-Buijs et al. 1998; Postma-Blaauw et al. 2010), with potential repercussions throughout the invertebrate food chain. The trend of increased soil health attained with sustainable soil management practices, thus, may result in increased invertebrate abundance and diversity in arable fields. The effects of cover crops and green waste compost amendment on aboveground invertebrates, however, remain largely unexplored.

The aim of this chapter was to investigate if phacelia (*Phacelia tanacetifolia*)/ radish (*Raphanus sativus*)/ vetch (*Vicia sativa*) cover crop mix and green waste compost amendment have the potential to positively affect in-field assemblages and abundance of aboveground invertebrate populations during breeding bird season, when prey availability is crucial in determining the survival of farmland bird chicks.

3.2 Methods

To investigate whether cover crop and green compost amendment may positively affect the aboveground invertebrate community of arable landscapes, this study measured invertebrate diversity and abundance and habitat characteristics along transects extending towards the crop centre in three experimental fields managed with four combinations of sustainable soil treatment. Invertebrates were collected with suction sampling and pitfall trapping to obtain an estimate of community composition and abundance of both canopy and ground dwelling invertebrates. Habitat characteristics of soil quality and vegetation cover were examined in plots with or without cover crop and green compost, and compared to measures obtained in the field boundary. The effects of cover crop and compost treatments, together with their interaction, were then analysed with multivariate analysis and generalised linear mixed model to investigate if, respectively, they changed the assemblage of invertebrate taxa and if they increased their overall abundance (3.1).

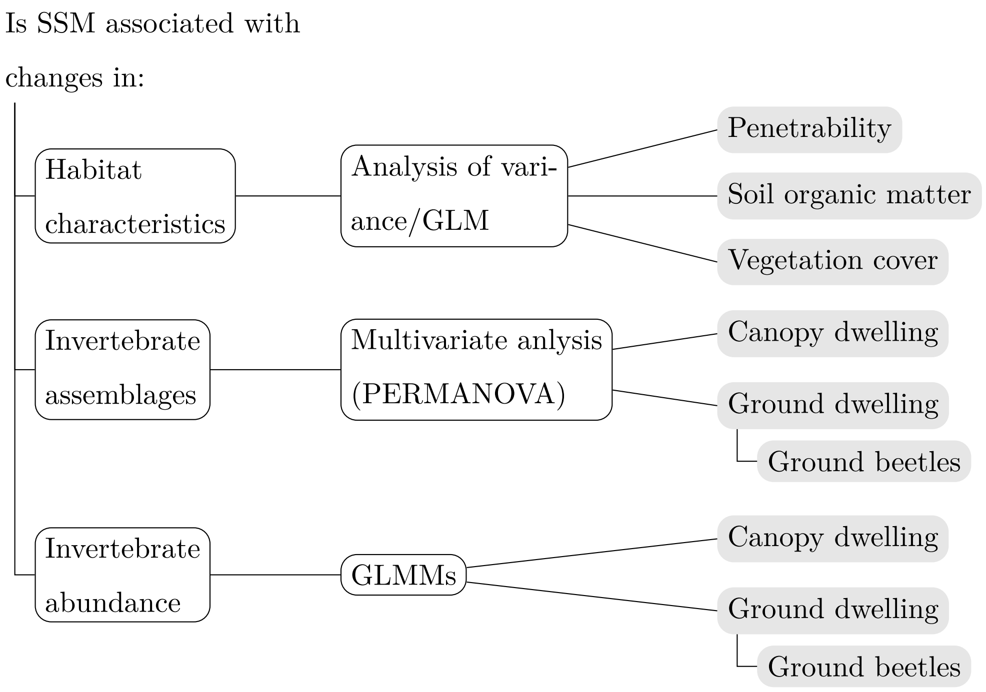

Figure 3.1: Schematic representation of the analysis of this chapter, which investigates the effects of cover crop and compost amendment on in-field habitat characteristics and invertebrate community composition and abundance in the three experimental fields at RSPB Hope Farm (see §1.4).

3.2.1 Fields and sampling design

This study took place in May and June 2018 in three fields at Hope farm that since 2015 have been divided following a split-plot experimental design in four plots undergoing different combinations of crop management: traditional or sustainable soil management with post-harvest green waste compost soil amendment (30 t/ha), and traditional rotation of cash crops or rotation with a cover crop (vetch/radish/phacelia mix) between harvests.

Within each experimental plot, defined as a field quadrant managed with or without cover crop and with or without compost (Fig.3.2), measures of invertebrate abundance and diversity, soil penetrability, soil organic carbon, and vegetation cover and were taken,

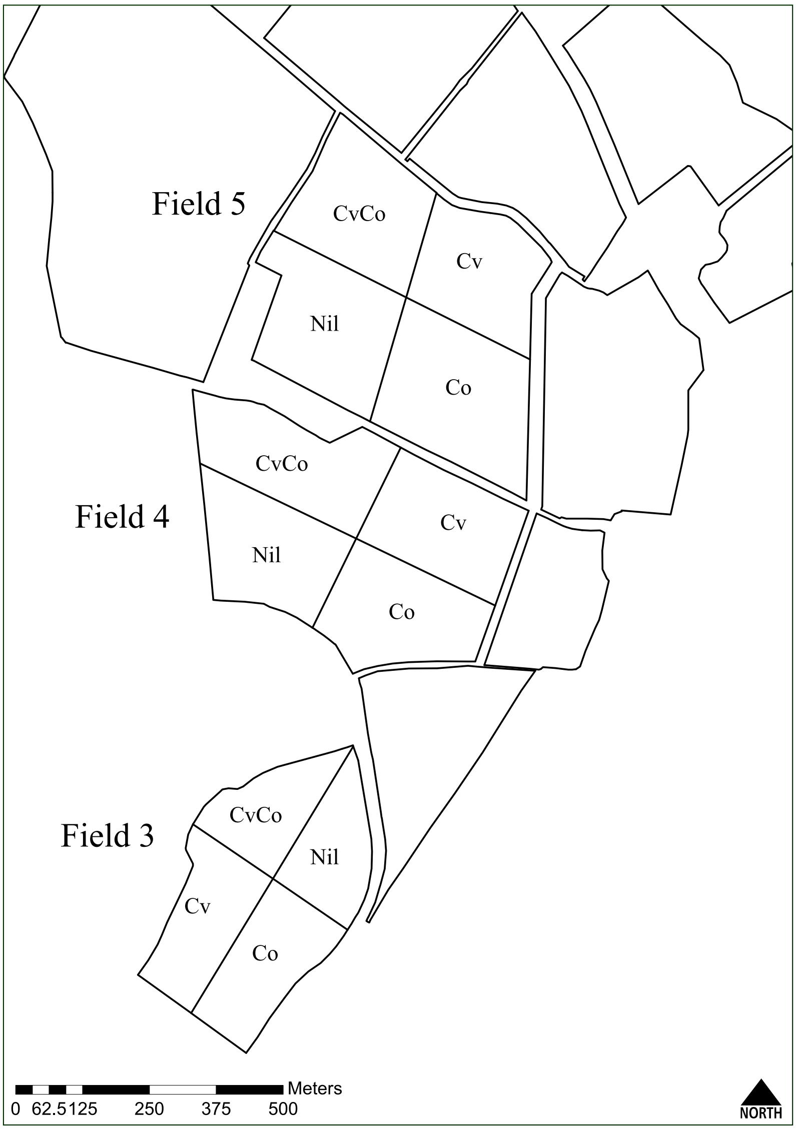

Figure 3.2: Map of the three experimental split-plot fields with combinations of sustainable soil management. **CvCo** plots with both cover crop and compost, **Cv** plots with cover crop only, **Co** plots with compost only, **Nil** marks plots with no sustainable soil treatment.

as described in §2.2.2. Samples were collected at three points (boundary, 4 m and 32 m) located along transects projecting from the field boundary towards the field centre. Three transects were placed in each soil management plot.

Sampling of invertebrates, and measures of vegetation cover and soil penetrability occurred twice along the same transects, once during the week of 21st-27th of May 2018, and then during the week of 18th-24th of June 2018. Soil samples for the determination of organic carbon were taken during the May sampling week, as measures of loss on ignition would not change between the first and second week of sampling. A total of 216 points were sampled, 144 in the fields and 72 in the field boundary. Estimates of organic soil content were obtained from soil samples collected during the first week of sampling.

3.2.2 Soil sampling

Penetrability Soil static penetrability is the measure of ease at which an instrument (i.e. penetrometer) can be pushed into the soil (Bradford 1986), and is used as a measure of soil compaction and as a proxy for soil water content (Green 1988). Higher measurement of penetrability are associated with higher compaction and drier soil. Cone resistance, the resistance to penetration developed by a cone of known basal area, was measured at each sampling location with an Eijkelkamp hand penetrometer (EijkelkampAgrisearch Equipment, NL) with an average measurement error of ±8%. Cone penetrability was estimated as the average of three consecutive measures taken 50 cm apart on undisturbed soil.

Soil Organic Matter Oven dry soil samples were ground to < 2 mm using a mechanical soil grinder (model H-4199, Humboldt Mfg. Co., USA) and homogenised. Average soil organic matter content was determined by loss on ignition of three replicates per sample. Approximately 2 g of soil were heated at 550 °C for 16 h. Standard soil was also ignited to ensure that the temperature of the furnace was homogenous. Loss on ignition was calculated as a percentage of the original weight using the weight loss between 105 °C and 550 °C and averaged across the three replicates.

3.2.3 Vegetation cover sampling

Vegetation cover was measured at each location using a point quadrat frame with ten equally-spaced vertical 3 mm pins, placed randomly in the field boundary sampling point and perpendicularly to crop rows at the 4 m and 32 m in-field sampling locations. The number of touches of living plants along each pin at 5 cm intervals, with a maximum number of touches of 15, was recorded to obtain a measure of vegetation cover within a vertical 8x10 grid. Vegetation contact frequency was estimated by dividing the number of touches recorded by 1200, the maximum number of possible hits in one frame. The crops in the fields were staged according to Tottman et al. (1979).

3.2.4 Aboveground invertebrates sampling

Pitfall trapping is the most common –and often the sole– form of invertebrate sampling used to portray ground dwelling arthropod assemblages in terrestrial habitats (e.g. Buddle et al. 2006; Knapp et al. 2013; Brown and Matthews 2016; Hohbein and Conway 2018; Biffi et al. 2019). It can be used to estimate the "activity-density" of a species, which reflects its abundance as activity during the sampling, and population density in the area sampled (Southwood 1978). In arable landscapes, pitfall traps are often used to investigate the patterns of ground beetles (*Carabidae*) abundance and diversity. Ground beetles are used as indicator species, as they are known to be sensitive to a range of agricultural practices, such as cropping regimes, pesticide application, and management intensity (Cárcamo et al. 1995; Aviron et al. 2005; Hanson et al. 2016). Using pitfall trapping alone to compare ground dwelling invertebrate assemblages, however, may only capture a small proportion of the fauna (primarily large-bodied, mobile predatory species, such as spiders and ground beetles) during a temporal arch that includes species not necessarily available to most birds, such as nocturnal species.

Since its first development (Dietrick 1961) suction sampling has increasingly been used to collect canopy dwelling invertebrates from arable cropland. Suction sampling gathers invertebrates from all locations within a known volume of vegetation, including soil surface, weeds and crop foliage, and, thus, it can obtain an accurate measure of taxa density. This type of sampling collects most of the known invertebrate groups eaten by birds (Holland et al. 2012; Wood et al. 2013), but underestimates large and heavy bodied invertebrates in the field, such as ground beetles species.

Most invertebrates were identified to Order or Family level, while ground beetles were identified to species. Standard identification references were used, such as Chinery (1986), Tilling (1987), D. M. Unwin (1988), Forsythe (2000), D. Unwin (2001) and Luff (2007).

3.2.4.1 Ground dwelling invertebrates

At each sampling location, one pitfall trap was placed to sample ground surface dwelling invertebrates. The traps consisted of plastic cups (diameter 7 cm, height 15 cm) sunk into the ground, flush with the soil surface, and partly filled with 10% saline solution (10 g of rock salt per 100 ml water) to preserve the specimens, and a drop of unscented detergent to reduce surface tension for smaller bodied invertebrates. A 25 mm mesh was secured around the trap to reduce the likelihood of small mammals falling in the trap. A plastic roof kept in place with a metal stake prevented rainwater from flooding the traps, which were also checked every two days for evaporation levels during the first week of sampling (May 2018), and daily during the second week of sampling (June 2018) due to the increased seasonal temperature. Instances of trap disturbance were noted. All trap disturbances (6.9% of all traps at Hope Farm) occurred in one half of Field 3 (Cv and Co plots) and were likely due to the presence of a predator in the area. Traps were emptied after seven days and samples were stored in 70% ethanol prior to identification.

The late-May and late-June sampling weeks were chosen to capture species typically found in early summer that are accessible to farmland birds during the early stages of development. This sampling strategy does not depict late-summer species activity and assemblage, which can differ significantly from the May-June period, as single ground beetle species peak during different times depending on their ecology (Holland 2002). Although longer sampling periods are often desirable, a sampling period of just two weeks has been shown to reflect the species composition of either early or late season species, but not of the entire season (e.g. Niemelä et al. 1990; Holland and Smith 1999). Sampling intervals of seven days have been shown to be the most common across ecological literature, and has been suggested as standard practice for comparability across studies (Brown and Matthews 2016; Hohbein and Conway 2018).

3.2.4.2 Canopy dwelling invertebrates

Canopy dwelling invertebrates were sampled with a Vortis suction sampler (Burkard Manufacturing Co. Ltd., UK, Fig.3.3) with a collecting area of 0.2 m^2. Each individual sample comprised of three bulked sucks lasting 15 seconds each, for a total covered area of 0.6 m^2. The engine was kept running between consecutive sucks. Prior to identification, samples were stored in resalable labelled bags and frozen.

3.2.5 Statistical analysis

Cover crop and green compost amendment were two main interacting treatments and models included the cover crop $*$ green compost term. *Post hoc* pairwise comparisons were conducted after a significant effect of the interaction term. The analysis of canopy and ground dwelling invertebrates assemblage and total abundance included the random factor *field and sampling week* to account for fields differences between the May and June sampling. Analysis of habitat characteristics collected during both samplings included the random factor *sampling week* to account for repeated measures. All analyses were conducted in R.

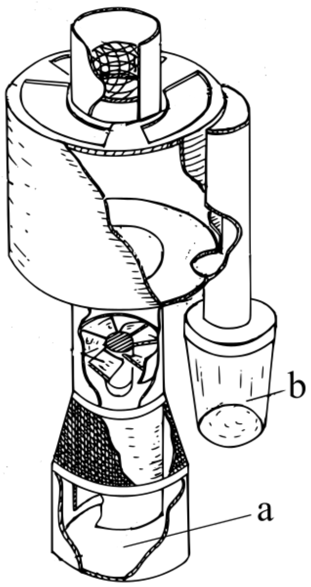

Figure 3.3: Scheme of Vortis insect suction system, modified from Arnold (1994). The sampling tube (*a*) is lowered to the ground for sampling invertebrates from canopy vegetation and soil surface. Invertebrates are collected in a removable insect container (*b*).

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As shown in Fig. 3.1, the analysis occurred in three steps of analysis:

1. The effect of soil treatment on habitat characteristics (§3.2.5.1).
2. The effect of soil treatment and habitat characteristics on **assemblage** of (*i*) canopy dwelling invertebrates, (*ii*) ground dwelling invertebrates, and of these, specifically (*iii*) ground beetles (§3.2.5.2).
3. The effect of soil treatment and habitat characteristics on **abundance** of (*i*) canopy dwelling invertebrates, (*ii*) ground dwelling invertebrates, and of these, specifically (*iii*) ground beetles (§3.2.5.3).

3.2.5.1 Soil characteristics and vegetation cover in the field and boundary

The difference in habitat characteristics associated with soil treatments were examined with non-parametric analysis of variance and generalised linear models. The effects of management on soil organic matter content were investigated using the non-parametric Kruskal-Wallis rank test, followed, in case of significant differences, by Wilcoxon signed-rank tests for *post hoc* pairwise comparisons with Holm-Bonferroni corrections. Soil treatment effects on habitat variables measured during the May and June sampling weeks (i.e. soil penetrability and vegetation cover) were investigated using generalised linear models using `lme` in `lme4` (Pinheiro et al. 2019). Main fixed effects were the presence or absence of cover crop, or of green compost amendment. The random fixed effect of *sampling week* was included to account for potential changes between the May and June sampling.

3.2.5.2 Invertebrate community composition analysis

Multivariate analyses were undertaken to investigate whether soil treatment combinations were associated with differences in in-field invertebrate community and ground beetle community. Abundance of canopy and ground dwelling invertebrates (separated by Order), and abundance of ground beetle species (Carabidae) collected at 4 m and 32 m along the transects were taken into account.

Tests of the multivariate null hypotheses of no differences among a priori defined groups (i.e. combinations of soil treatments) were examined using permutational multivariate analysis of variance (PERMANOVA, M. J. Anderson 2001, 2017), analysis of similarities (ANOSIM, Clarke 1993), and non-metric multidimensional scaling plots (nMDS).

PERMANOVA and ANOSIM are non-parametric methods based on permutation tests and do not rely on stringent assumptions, rendering them particularly suitable for ecological analysis (M. J. Anderson 2001). These tests were performed with 999 random permutations on the basis of non-Euclidean distances. The significance level to reject the null hypothesis was set at $\alpha = 0.05$ in all cases. Permutations were constrained within field and week to account for crossed sampling design.

Permutational multivariate analysis of variance (PERMANOVA) was performed using the `adonis` function in R in the package `vegan` (Oksanen et al. 2019). Soil treatment main effects were included as fixed factors along with their interaction effect, together with categorical (i.e. distance along the transect) and quantitative habitat covariates (i.e. vegetation cover, soil penetrability, and soil organic matter). Count data were standardized by dividing the abundance of each taxon by the total number of individuals caught in each sample. Standardized data were $\ln(x+1)$ transformed to down-weigh the contribution of particularly abundant taxa in relation the less common and rare ones when calculating the similarities between samples (Clarke 1993). A resemblance matrix was calculated using Bray-Curtis dissimilarities (Bray and Curtis 1957) based on the $\ln(x+1)$ transformed standardized data. The Bray-Curtis dissimilarity index is a semi-metric, non-Euclidean distance particularly useful in community ecology to represent shared abundance between populations. If two samples contain no individuals, the standard Bray-Curtis coefficient is undefined. Therefore, because of the sparse abundance of invertebrates at the end of May compared to the end of June, a zero-adjusted Bray-Curtis dissimilarity index was calculated by adding a dummy species with abundance of 1 to all samples, following Clarke et al. (2006), allowing to include empty traps in the analysis.

Given that categorical terms in the PERMANOVA, such as soil treatment, can be significant due to differences in groups (ie. location effect) only, or difference in groups *and* variability across groups (i.e. dispersion effect), Beta-dispersion analysis (`betadisper` in `vegan`) was used to test homogeneity of dispersion among grouping factors M. J. Anderson (2017). For significant treatment effects in PERMANOVAs, *post hoc* pairwise comparisons were conducted to test for faunal assemblage differences among soil treatment combinations, using Holm-Bonferroni p-value adjustments for multiple comparisons. A test of homogeneity of multivariate dispersion (PERMDISP) was conducted on the Bray-Curtis dissimilarity matrices to accompany the PERMANOVAs and identify significant differences in the spread of the groups around their centroids.

The PERMANOVAs were defined as follows:

```
permanova = adonis (community matrix ~
            cover crop * compost +
            distance from edge +
            vegetation cover +
            penetrability +
            organic matter,
            strata = field and sampling week)
```

The robust non-parametric ANOSIM, analogue of analysis of variance (ANOVA), was conducted using the above similarity matrix and the individual soil treatment combinations to provide, together with the nMDS plots, comparison with the result of the PERMANOVA test. ANOSIM provides an R statistic ranging from 0 to 1 for no difference between groups to maximum difference between groups, respectively. The test were performed using `anosim` in the package `vegan`.

The distribution of the data was visualised using nMDS plots using the `metaMDS` function in the `vegan` package. The goal of nMDS is to show accurately the dissimilarity matrix distribution in multidimensional space using a reduced number of dimensions. Unlike Principal Component Analysis, which uses Euclidean distances, it relies on ranked distances, and is therefore not sensitive to data transformation. The plot arranges the points in the ordination while preserving the rank-order distances, and its optimization is quantitatively measured as stress (Kruskal 1964). It is generally accepted that a stress above 0.2 is deemed suspect and a stress value approaching 0.3 indicates that the ordination is arbitrary, while stress values equal to or below 0.1 indicate good fit. This rule of thumb, however, may not apply to particularly large datasets (Dexter et al. 2018).

3.2.5.3 Effects of soil treatment and habitat covariates on invertebrate abundance

Generalised Linear Mixed Models (GLMMs) with penalized-quasi likelihood were used to investigate the potential effect of soil treatment combinations on in-field invertebrate abundance. Total counts of invertebrates collected with suction sampling or pitfall trapping, and abundance of ground beetles (Carabidae) collected at 4 m and 32 m were taken into account. Response variables were individually modelled. Soil treatments (i.e. cover crop and green compost amendment) and their interactions, and distance along the transects were fixed categorical factors. Abundance of invertebrates or ground beetles collected in the field boundary (2 m from the field edge) was included as a quantitative covariate to determine if in-field differences were influenced by abundance outside the field, together with in-field measures of vegetation cover, soil penetrability, and soil organic matter content (3.1). The random term of *field and sampling week* was added to account for differences among fields in time that would otherwise not captured by the model. This random main effect introduced an component of variation in the model, and incorporated the dependency among observation within the same field and collection week. Quasi-Poisson GLMMs with *log*link function were used to account for over dispersed count data using the `glmmPQL` function in the R `MASS` package (Venables and Ripley 2002). The *log*link function ensures positive fitted values, and the Poisson distribution is typically used for count data. Quasi-likelihood models represent a partially non-parametric approach allowing various optimality properties regardless of the precise underlying data distribution, to avoid transformation of the dependent variable (Venables and Dichmont 2004; Bolker et al. 2009). When not significant, the soil treatment interaction was removed for model simplification. *Post hoc* pairwise comparisons were conducted using `emmeans` and `pairs` in the `emmeans` package with Bonferroni adjustment to control for multiple comparisons (Lenth et al. 2018). Model assumptions were verified by plotting residuals against fitted values and each covariate in the model. All tests of null hypotheses were evaluated against a standard of $\alpha = 0.05$.

As Venables and Ripley (2002) do not allow for quasi-AIC calculations for quasi-Poisson models, R^2_{GLMM} was calculated as summary statistics using `rsquared` in `piecewiseSEM` (Lefcheck 2016). R^2_{GLMM} is used to summarize the variance explained at different levels in a model: marginal R^2_{GLMM} for fixed effects and conditional R^2_{GLMM} for fixed and random effects together (Nakagawa and Schielzeth 2013; Nakagawa et al. 2017).

The GLMMs were defined as follows:

```
m = glmmPQL (invertebrates count ~
        cover crop * compost +
        distance from edge +
        invertebrate count in boundary +
        vegetation cover +
        penetrability +
        organic matter +
        (1|field and sampling week)
        )
```

Table 3.1: List of covariates and data type used in the analysis.

Covariate	Type
Cover crop	Categorical (yes/no)
Compost	Categorical (yes/no)
Distance from edge	Categorical (4 m, 32 m)
Abundance in boundary	Continuous
Vegetation density	Continuous
Penetrability	Continuous
Organic matter	Continuous
Cover crop $*$ Compost	

3.3 Results

3.3.1 Soil characteristics and vegetation cover in the field and boundary

Habitat characteristics, with the exception of soil organic matter content, did not appear to be strongly influenced by sustainable soil management. Soil penetrability and vegetation cover did not show consistent patterns with soil treatments, while soil organic matter content was significantly higher in plots with green compost amendment.

In-field soil organic matter measured by loss on ignition was significantly higher when green compost was applied to the soil (Kruskal-Wallis $\chi^2 = 11.64$, $p < 0.001$, Fig.3.4), while there was no difference between plots with and without cover crop (Kruskal-Wallis $\chi^2 = 0.01$, $p = 0.928$). There was a difference in the amount of organic matter in and outside of the field (Kruskal-Wallis $\chi^2 = 14.40$, $p < 0.001$), with the boundary having soil richer in organic content than samples taken at 4 m (Wilcoxon test $p = 0.003$) and 32 m (Wilcoxon test $p = 0.001$, Fig. 3.4).

In-field soil penetrability did not differ between the two round of samplings (Kruskal-Wallis $\chi^2 = 3.53$, $p = 0.060$). Soil penetrability recorded during the second round of sampling, however, was generally higher (Appendix Fig. B.1). A linear mixed model with *sampling week* as random intercept factor to account for increased penetrability over time showed no significant effect of cover crop (β = 9.11, t_{141} = 0.57, p = 0.569) or compost (β = -2.42, t_{141} = -0.15, p = 0.879) on soil penetrability.

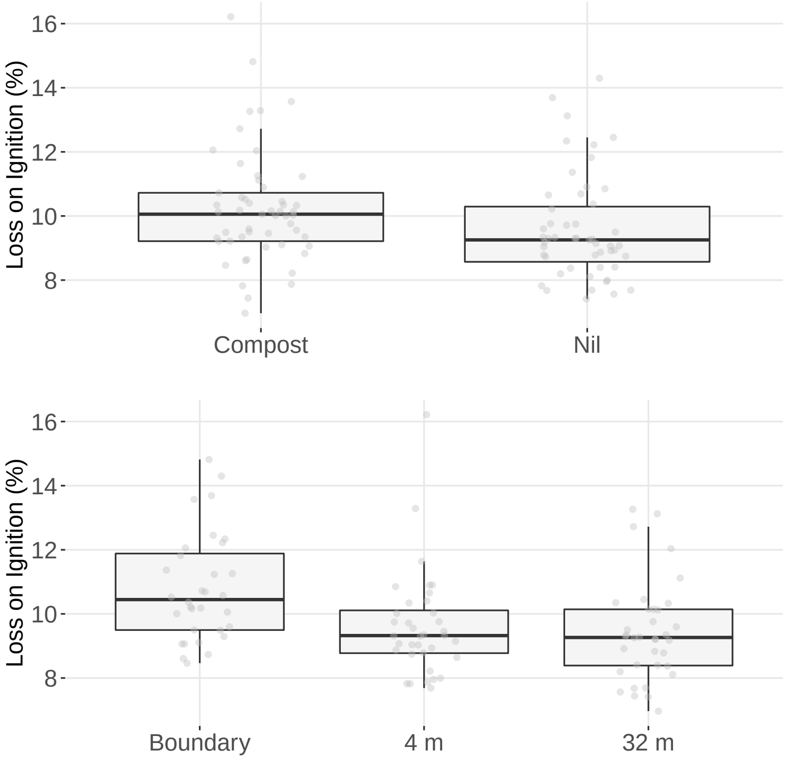


Figure 3.4: Organic matter content in plots with and without green waste compost amendment (*top*) and along transect sampling point (*bottom*).

There was no clear pattern of vegetation cover with soil treatments. A generalised linear mixed model with *sampling week* as random intercept factor, to account for crop growth in time, showed no significant effect of soil treatment in plots with and without cover crop (β = <-0.01, t_{141} = -0.23, p = 0.999) or compost amendment (β = -0.76, t_{141} = -1.46, p = 0.146). As the crops grew, the differences in vegetation cover in the same plot between the first and second round of sampling changed independently from the soil treatment, suggesting that patterns of crop growth were more strongly influence by local factors (Fig. 3.5). Vegetation cover changed significantly along the transects (Kruskal-Wallis $\chi^2 = 76.82$, $p < 0.001$), with Wilcoxon rank-signed tests showing field boundary as the most dense ($p < 0.001$). Vegetation cover at 32 m was consistently higher than at

4 m (Wilcoxon test $p < 0.001$, Fig. 3.6). Crops growth stages in each field are shown in Table 3.2

Table 3.2: Crop staging for Hope Farm fields according to Tottman et al. (1979). Ranks indicate the temporal succession of growth stages.

	Month	Cereal	Stage	Rank
Field 3	May	Spring barley	Stem Elongation	2
Field 4	May	Winter wheat	Stem Elongation	2
Field 5	May	Spring barley	Tillering	1
Field 3	June	Spring barley	Ear emerged	4
Field 4	June	Winter wheat	Flowering	5
Field 5	June	Spring barley	Booting	3

3.3.2 Invertebrate abundance

A total of 30,989 invertebrates were identified and counted, 18,950 from suction samples and 12,039 from pitfall traps. The composition of invertebrate community sampled with the two techniques was different, with Collembola, Acari, and Diptera dominating the suction samples, and Coleoptera and Araneae the pitfall traps. The relative proportion of the seven most abundant taxa for both sampling methods is shown in Table 3.3. Canopy dwelling invertebrates abundance was highest in the field boundary, and lowest at the field edge (Appendix Fig. B.2), while total pitfall catches decreased along the transects from the boundary towards the field centre (Fig. B.3). The overall catch of invertebrates increased between the May and June sampling, and invertebrates count was significantly higher in the field boundary than in the field (Fig.3.7).

A total of 5,273 individual adult ground beetles (*Carabidae*) representing 29 species were collected across all pitfall traps during the sampling. The overall assemblage was a representative mixture of common species associated with arable agricultural landscapes (Appendix Tab. B.1). Overall, six species represented 92.2% of the entire sample (Fig. 3.8). The assemblage was strongly dominated by *Harpalus rufipes* (64.6%), *Leistus rufomarginatus* (13.9%), and *Poecilus cupreus* (9.9%) during the May sampling, and by *Pterostichus madidus* (6.1.2%), *Harpalus rufipes* (16.9%), and *Pterostichus melanarius* (11.8%) during the second week of sampling.

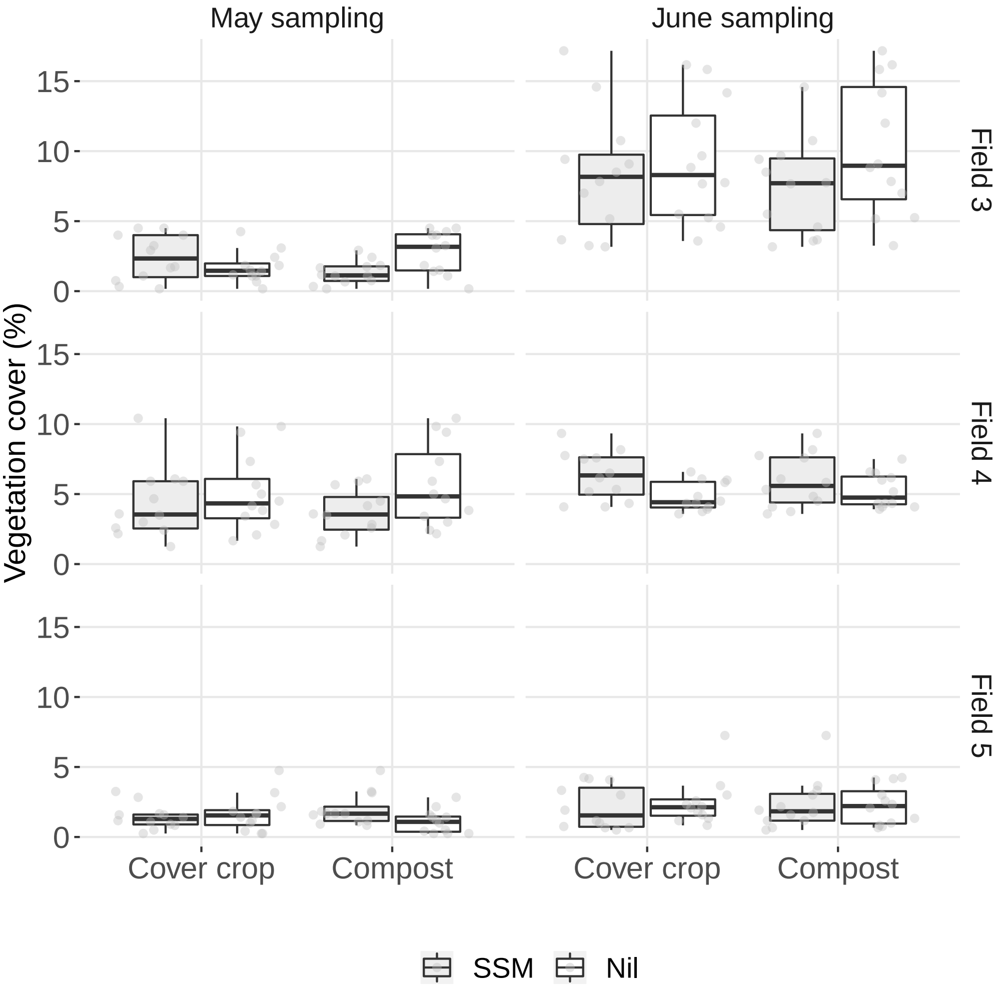


Figure 3.5: Vegetation cover in the three fields during sampling in May and June. Grey boxes (SMM) indicate plots with cover crop or compost amendment, while white ones (Nil) show plots without that soil treatment.

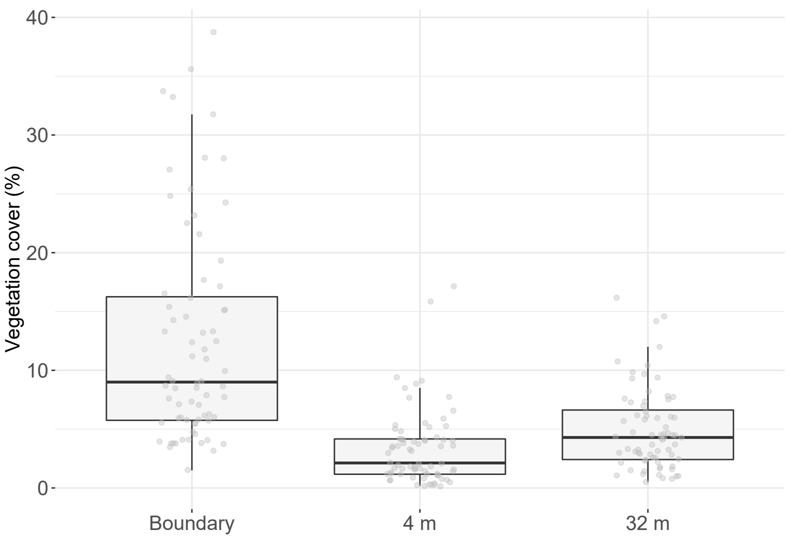


Figure 3.6: Vegetation cover measured with pin quadrats along transect sampling points. Cover was highest in the field boundary, and significantly higher further into the fields that close to the field edge.

Table 3.3: Relative abundance of the seven most common taxa collected with suction sampling and pitfall trapping.

Method	Rank	Taxa	% of Sample
Suction sampling	1	Collembola	39.4
	2	Acari	18.7
	3	Diptera	12.0
	4	Hemiptera	11.9
	5	Hymenoptera	6.8
	6	Coleoptera	4.2
	7	Araneae	2.9
Pitfall trapping	1	Coleoptera	59.0
	2	Araneae	23.6
	3	Opiliones	8.6
	4	Hymenoptera	5.4
	5	Isopoda	2.8
	6	Gastropoda	0.3
	7	Chilipoda	0.1

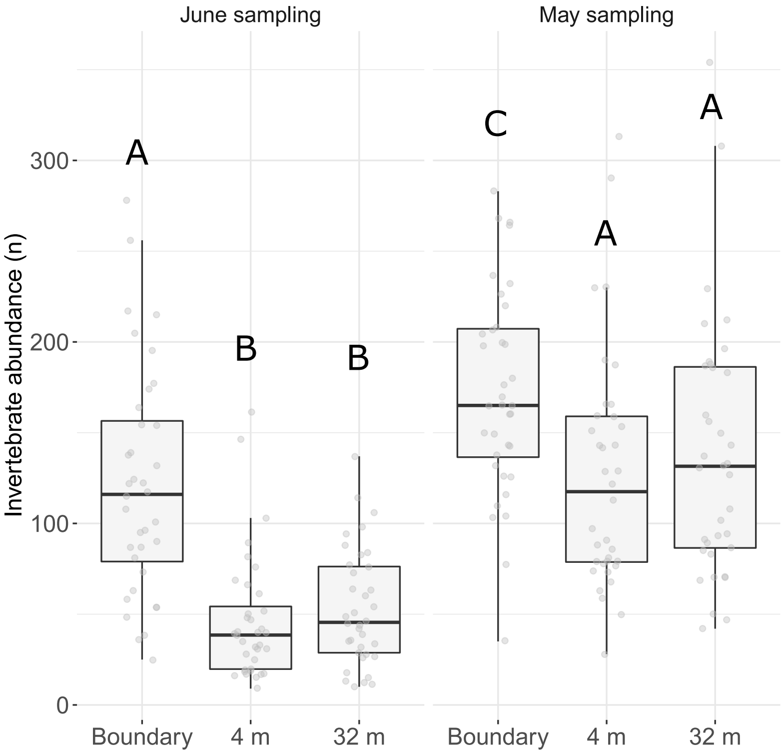


Figure 3.7: Total invertebrate abundance along the transects during May and June sampling. Different letters indicate significant difference according to pairwise Wilcoxon signed-rank test with Holm-Bonferroni multiple comparison adjustments: $p_{avsb} < 0.001$, $p_{avsc} < 0.04$.

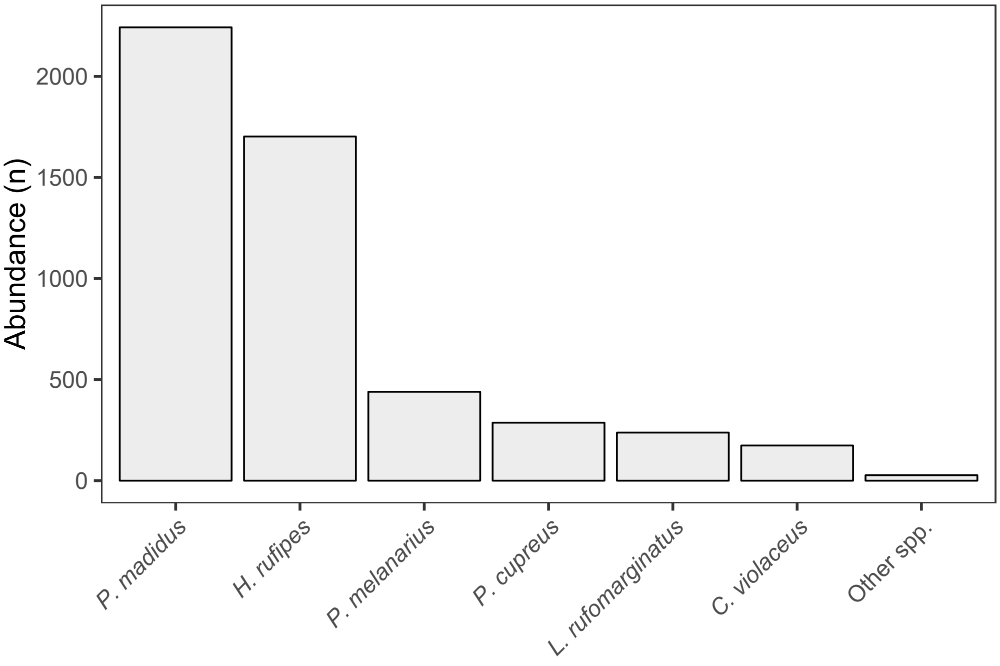


Figure 3.8: Abundance of the six most common species of ground beetles across both sampling rounds. P - *Pterostichus*/*Poecilus*; H - *Harpalus*; L - *Leistus*; C - *Carabus*.

3.3.3 Invertebrate community composition analysis

Multivariate analyses did not show significant changes in canopy and ground dwelling invertebrates assemblage among soil treatment main effects or interaction after restricting permutations for *field and sampling week.* The ground beetle community showed a significant composition change with treatment interaction. Assemblages of canopy dwelling invertebrates and ground beetles changed from the field edge towards the field centre, while ground dwelling invertebrates remained homogeneous along the transects.

Canopy dwelling invertebrates There was no significant effect of soil treatment on the community composition of canopy dwelling invertebrates according to PERMANOVA (Tab. 3.4) and ANOSIM (R = -0.012, P = 0.292) analyses. Instead, the taxa assemblage in the fields changed along the transects with sampling distance from the edge and with vegetation cover. Closer to the field boundary, the presence of phytophagous insects, such as aphids and some Heteroptera, increased, presumably in association to localised patches of higher vegetation cover (as this was generally higher at 32 m then at 4 m from the field edge, as shown in Fig. 3.6). Close to the field edge there also was a higher number of small canopy dwelling Coleoptera, such as phytophagous Curculionoidea and Chrysomelidae, detritivorous Cucujoidea, and predator Staphylinidae. These may depend on the resources availability of the field boundary. Springtails (Collembola), mites (Acari), and spiders (Araneae) were more dominant in the community 32 m into the fields than close to the edge. Collembola can reach densities up to 100,000 m^{-2} and they comprise a major source of prey food to predator mites and spiders in arable fields (Kuusk and Ekbom 2012). As arable fields contain high densities of spiders, that are frequently close to starvation (Bilde and Toft 1998), localised higher Collembola densities may attract active predators and, thus, increase the density of mites and spiders, as shown by Axelsen and Kristensen (2000). Figure 3.9 shows the nMDS ordination plot representing the canopy dwelling invertebrate community at each sampling point with soil treatments.

Sampling distance from the field boundary edge had a significant effect on invertebrate assemblage (pseudo $F_{1,\,143} = 2.46$, $P = 0.001$, Appendix Fig. B.2).Vegetation cover appeared to be the main driver in community composition variation among the quantitative habitat variables (pseudo $F_{1,\,143} = 14.53$, $P = 0.001$).

Ground dwelling invertebrates The overall ground dwelling invertebrate community did not differ among soil treatments in the PERMANOVA (Tab. 3.4) and ANOSIM (R= -0.001, P = 0.278), and did not change from the field edge towards the field centre. There was a significant effect of vegetation cover on assemblage, while the other habitat cov-

Table 3.4: Results of PERMANOVA and PERMDISP based on Bray-Curtis dissimilarities using $ln(x+1)$ transformed abundance data for canopy dwelling invertebrates collected with suction sampling (n = 144), and for ground dwelling invertebrates and ground beetles collected with pitfall trapping(n = 134). CD - Canopy dwelling; GD - Ground dwelling; GB - Ground beetles; Df - degrees of freedom; Sum Sq - sum of squares; Pseudo F - F value by permutation, boldface indicates statistical significance with $p < 0.05$, P-values based on 999 permutations.

			PERMANOVA				PERMDISP
	Source of variation	Df	Sum Sq	Pseudo F	p	Df	p
CD							
	Cover crop	1	0.096	0.721	0.221	1	0.926
	Compost	1	0.095	0.714	0.229	1	0.816
	Point	1	0.293	2.209	**0.001**	1	0.347
	Cover crop x Compost	1	0.077	0.582	0.644	3	0.939
	Vegetation cover	1	3.120	23.497	**0.004**		
	Penetrability	1	0.366	2.756	0.994		
	Organic matter	1	0.241	1.817	0.081		
	Residual	136	18.059	-	-		
	Total	143	22.348	-	-		
GD							
	Cover crop	1	0.018	0.336	0.710	1	0.949
	Compost	1	0.032	0.587	0.552	1	0.760
	Point	1	0.083	1.542	0.120	1	0.460
	Cover crop x Compost	1	0.103	1.904	0.115	3	0.955
	Vegetation cover	1	0.330	6.112	**0.043**		
	Penetrability	1	0.194	3.592	0.521		
	Organic matter	1	0.039	0.724	0.750		
	Residual	126	6.802	-	-		
	Total	133	7.600	-	-		
GB							
	Cover crop	1	0.124	0.794	0.315	1	0.180
	Compost	1	0.174	1.111	0.163	1	0.815
	Point	1	0.312	1.991	**0.017**	1	0.103
	Cover crop x Compost	1	0.372	2.370	**0.010**	3	**0.004**
	Vegetation cover	1	0.545	3.475	0.382		
	Penetrability	1	1.202	7.666	0.058		
	Organic matter	1	0.018	0.112	0.977		
	Residual	125	19.754	-	-		
	Total	133	22.500	-	-		

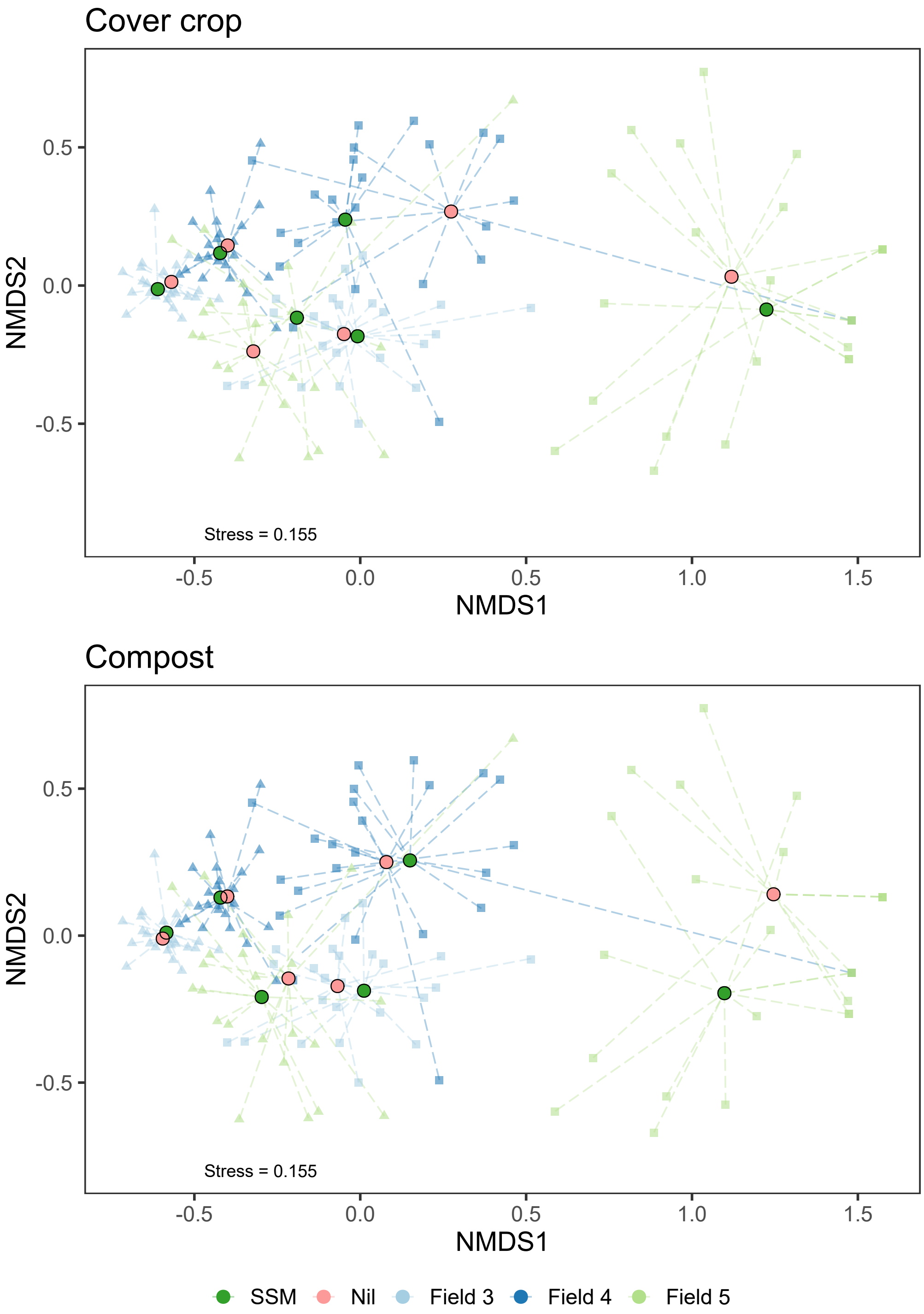


Figure 3.9: Non-metric multi dimensional scaling ordinations (nMDS plot) on the basis of the Bray-Curtis dissimilarity measures of $ln(x+1)$-transformed abundance of canopy dwelling invertebrates obtained in the three fields during the May (*squares*) and June(*triangles*) sampling weeks, showing rank-order relationship for *top*: plots with (SSM) and without (Nil) Cover crop; *bottom*: plots with (SSM) and without (Nil) green compost amendment.

ariates also showed no significant effect, suggesting that the ground dwelling invertebrate assemblage was fairly homogeneous within the plots, at least at the taxonomic level considered in the analysis. Figure 3.10 shows the nMDS ordination plot of the ground dwelling invertebrate community at each sampling point combinations of soil treatments.

Ground beetles The community composition of *Carabidae* showed no significant change with soil treatment main effects. There was, however, a significant effect of the interaction term attributed to the Cv and Co treatments (Tab.B.2, Fig. 3.13). This difference was driven by the very localised increase of two species of ground beetles. *Harpalus rufipes* increased by 1162% in the treatment Co (i.e. only green compost amendment) of Field 3 during the first week of sampling. Here, *H. rufipes* numbers rose to an average of 80.2 (SD ±35.6) from an average of 6.9 (SD±7.0) in plots undergoing the same treatment. A strong increase in *Pterostichus madididus* abundance was responsible, instead, of the significant difference in assemblage in the Cv treatment plot (i.e. cover crop alone) of Field 4 in June. The total count of individuals of *P. madidus* in treatment Cv of Field 4, raised to 676 from an average of 35.2 (SD ± 20.9) in plots with the same soil management. This 1920% increase significantly altered the overall community composition of the treatment. Figure 3.11 shows the nMDS ordination plots representing the ground beetle species assemblage at each sampling point in different soil treatments.

Ground beetles assemblage differed significantly with distance from the field edge according to species habitat preference. While *Harpalus rufipes*, *H. affinis*, *Poecilus cupreus*, and *Pterostichus melanarius* were more abundant the edge than at the field centre, *Demetria atricapillus*, and *Pterostichus madidus* were found at higher numbers at 32 m than at 4 m along the transects (App. Fig.B.4). *Pterostichus madidus* and *P. melanarius* are known to favour the field centre habitat, therefore, the increased abundance of *P. madidus* at 32 m was not unexpected. A higher abundance of *P. melanarius* at the field edge than at the field centre has also been shown in other studies (e.g.Holland et al. 2004a), and may be explained by the time of sampling, as in May/June most captured individuals are adults associated to the field boundaries in which they overwintered. *H. rufipes* is a seed predator species and, as such, tends to remain associated with hedgerows and field margins (C. F. G. Thomas et al. 1997). These results are also in accordance with the findings of Saska et al. (2007) in winter cereals fields in the Netherlands that showed firstly that, in respect to their habitat preference within an arable field, *Harpalus affinis* can be classified as "field-edge species" and *Demetrias atricapillus* as "field-interior species". Secondly, that species within one genus (e.g. *Pterostichus*) can differ in their response to the boundary, hence the important of ground beetles identification to species level.

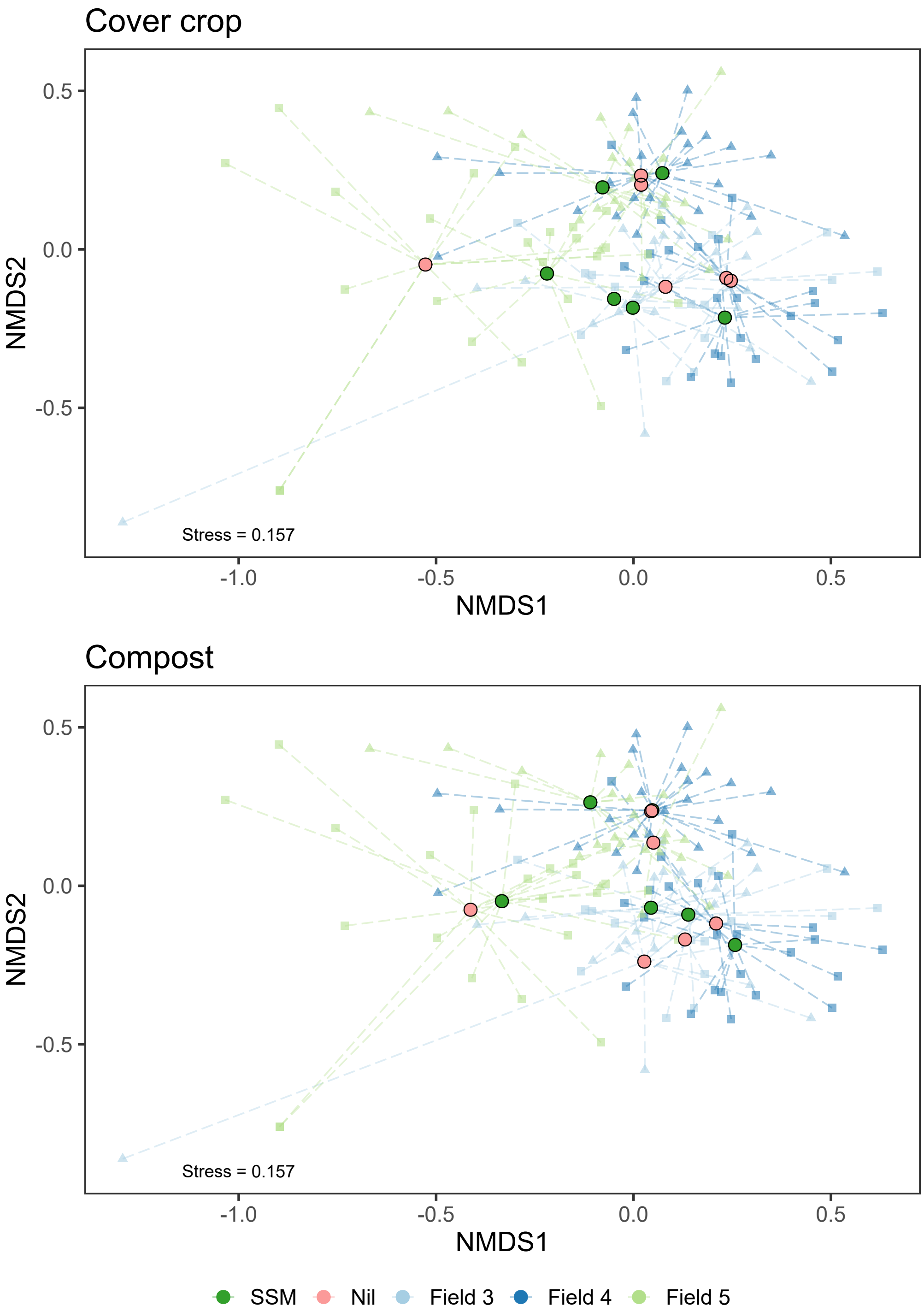


Figure 3.10: Non-metric multi dimensional scaling ordinations (nMDS plot) on the basis of the Bray-Curtis dissimilarity measures of $ln(x + 1)$-transformed abundance of ground dwelling invertebrates obtained in the three fields during the May (*squares*) and June (*triangles*) sampling weeks, showing rank-order relationship for *top*: plots with (SSM) and without (Nil) Cover crop; *bottom*: plots with (SSM) and without (Nil) green compost amendment.

The PERMANOVA found significant effect of the interaction term between the two soil treatments on community assemblage (pseudo $F_{1,\,133} = 2.61$, $P = 0.008$, Fig. 3.12, Tab. 3.4). *Post hoc* pairwise comparisons of the interaction term with Holm-Bonferroni multiple comparison p-value adjustment showed a significant difference in assemblage between the cover crop treatment with (CvCo) and without (Cv) compost amendment (pseudo $F_{1,\,66} = 2.55$, $P = 0.016$, Appendix Tab.B.2). PERMDISP analysis of heterogeneity of dispersion around group centroids was significant ($P = 0.006$), so that the significance of PERMANOVA may have been caused by both location and dispersion effects. ANOSIM showed a weak significant effect of soil treatment combination on species assemblage ($R = 0.06$, $P = 0.045$). However, ANOSIM tests are not robust in case of distribution heterogeneity around the group centroids (Anderson and Walsh 2013), therefore these results should be considered carefully. The sampling distance along the transect had a significant effect of in-field community composition (PERMANOVA pseudo $F_{1,\,133} = 2.17$, $P = 0.016$).

Multivariate analysis confirmed the change in ground beetle community composition between the first and second sampling (PERMANOVA: SSQ = 4.61, psuedo F = 41.82, P = 0.001). Figure 3.14 shows the difference in assemblage between the May and June samples, which were respectively dominated by smaller bodied species, such as *Harpalus rufipes* and *Leistus rufomarginatus*, and larger species, such as *Pterostichus* spp. and *Carabus violaceus*.

3.3.4 Effects of soil treatment and habitat covariates on in-field invertebrate abundance

The generalised linear mixed models showed a significant positive effect of cover crop on the total abundance of canopy and ground dwelling invertebrates. Ground dwelling invertebrates and ground beetles numbers were also significantly predicted by the cover crop $*$ compost interaction. This result appeared to be driven by a strong localised increase in *Pterostichus madidus*. Generally, marginal R^2_{GLMM} values were lower than conditional R^2_{GLMM}, indicating that the differences among fields at different growth stages were not negligible. Model validation indicated no problems.

Canopy dwelling invertebrates Cover crop had a significant positive effect on the abundance of canopy dwelling invertebrates ($t = 0.15$, $p = 0.036$, Tab. 3.5, Fig. 3.15) after removing the interaction term between soil treatments. The variance in the model explained by fixed factors alone was low (3.1% marginal R^2_{GLMM}), while the variance explained by fixed and random factors together was 98.9% (conditional R^2_{GLMM}).

Canopy dwelling invertebrates were more abundant further into the fields, and sampling

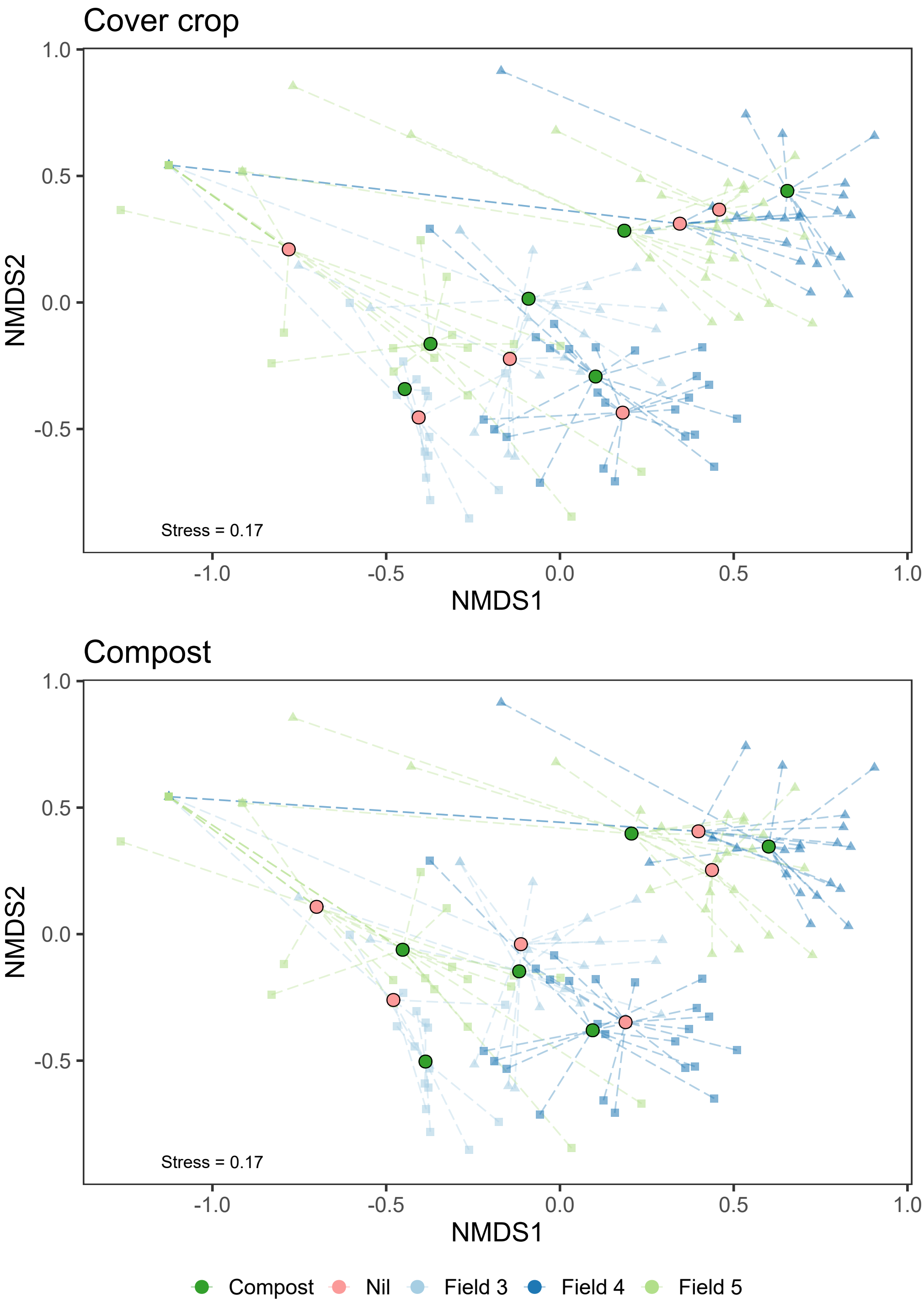


Figure 3.11: Non-metric multi dimensional scaling ordinations (nMDS plot) on the basis of the Bray-Curtis dissimilarity measures of $ln(x + 1)$-transformed abundance of ground beetles obtained in the three fields during the May (*squares*) and June (*triangles*) sampling weeks, showing rank-order relationship for *top*: plots with (SSM) and without (Nil) Cover crop; *bottom*: plots with (SSM) and without (Nil) green compost amendment.

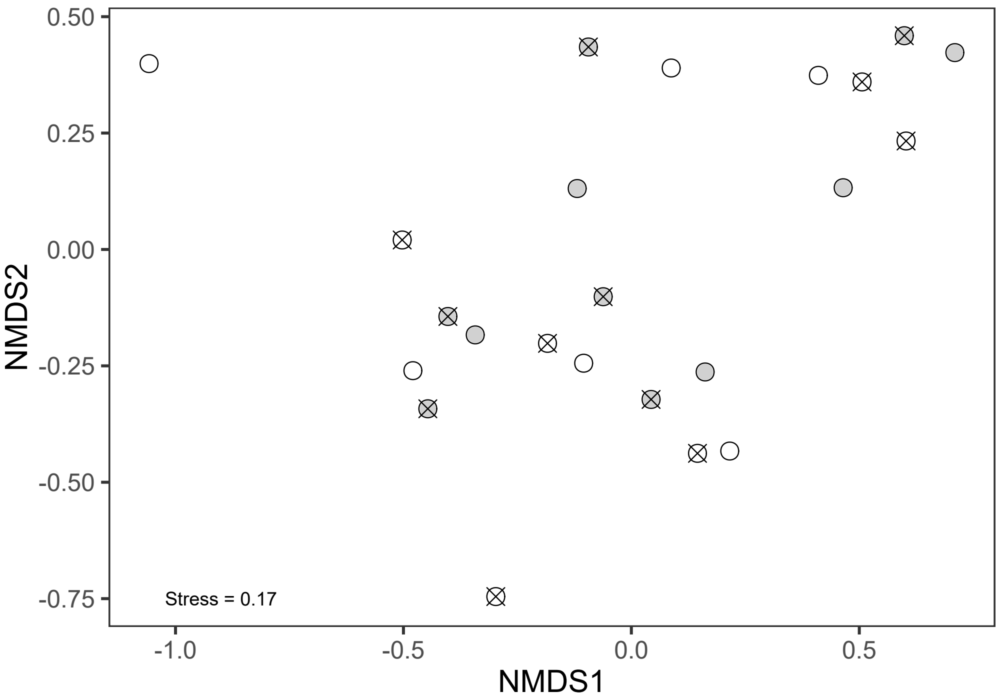


Figure 3.12: Non-metric multi dimensional scaling ordinations (nMDS plot) of distances among the centroids for plots with (*grey*) and without (*white*) cover crops, and with (*crossed*) and without (*empty*) compost amendment. The plot is based on Bray-Curtis dissimilarity measures of $\log(x + 1)$-transformed abundance of ground beetle species collected in n = 133 pitfall traps.

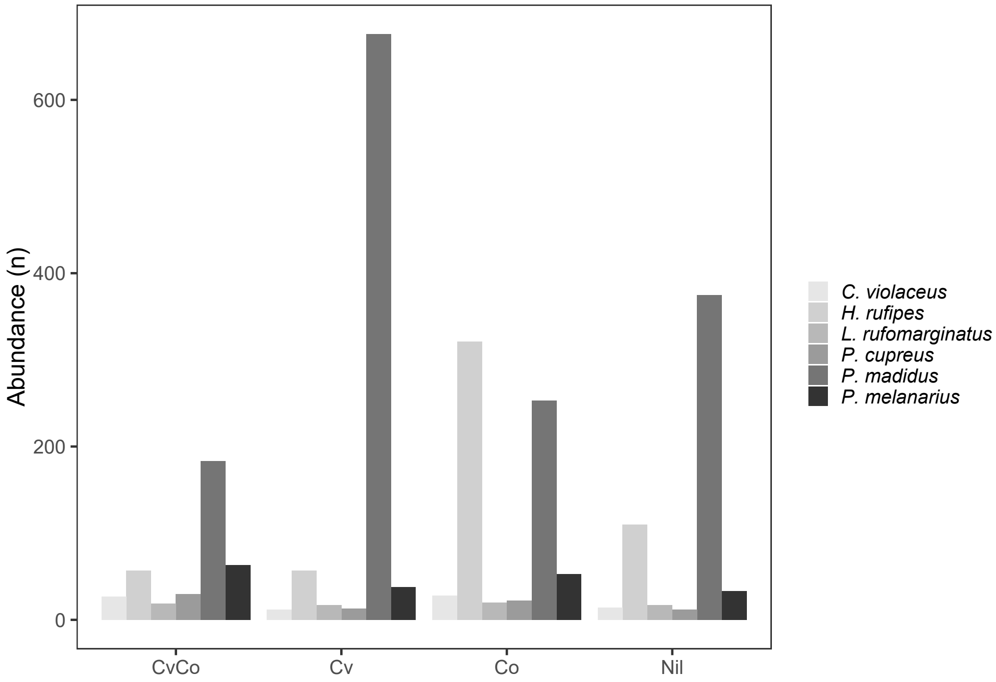


Figure 3.13: Abundance of the six most common ground beetle species in the four soil treatment combinations among May and June samples combined. *P. madidus* and *H. rufipes* were highest in the cover crop only (Cv) and in the compost only (Co) treatment, respectively. Cover crop and compost amendment (CvCo), cover crop (Cv), compost amendment (Co) and no sustainable soil management (Nil). P - *Pterostichus*/*Poecilus*; H - *Harpalus*; L - *Leistus*; C - *Carabus*.

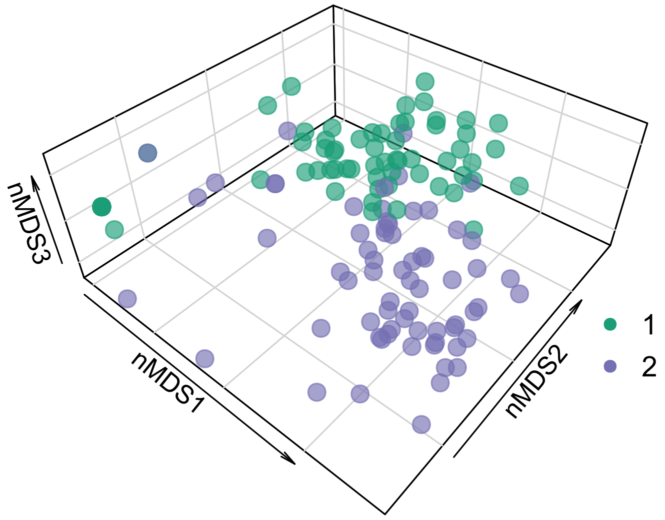


Figure 3.14: Non-metric multi dimensional scaling ordinations (nMDS plot) on the basis of the Bray-Curtis dissimilarity measures showing in-field ground beetles assemblage during the May (*purple*) and June (*green*) sampling.

points at 32m showed a significant increase in abundance compared to points at 4m (t = 4.32, $p < 0.001$, Appendix Fig. B.5), a pattern partly attributed to higher vegetation cover in the field centre (Fig. 3.6, 3.16). It would be expected that the colonisation of an arable field from the surrounding habitat would depend on the dispersal capacity of invertebrate organisms, and on their opportunity for reproduction and foraging within the cultivated crop. Previous studies have shown that the smaller and mobile species typically collected with suction sampling show very little association with their abundance in the field boundary (Holland et al. 2006; Holland et al. 2012). The distribution of these species is more strongly influenced by the composition of the surrounding landscape (e.g. as shown for spiders by Schmidt and Tscharntke 2005 and M. H. Schmidt et al. 2008), or by the local vegetation cover (e.g. as shown for phytophagous insects such as Heteroptera by Asteraki et al. 2004).

Localised higher measures of soil penetrability (i.e. cone resistance) were instead associated to lower canopy dwelling invertebrate abundance (t = -3.38, p = 0.001). Soil moisture, which is strongly associated with soil penetrability, has been shown to alter emergence of Diptera and small Coleoptera in the laboratory (Hulthen and Clarke 2006; Simelane 2014). It has also been shown to increase arthropod abundance during birds breeding season by favouring the emergence of aboveground arthropods in low intensity farming (Felici et al. 2019). Thus, lower emergence rate associated to local measures of soil hardness may have contributed to lower canopy dwelling invertebrate numbers.

Ground dwelling invertebrates The cover crop $*$ compost treatment interaction had a significant negative effect on ground dwelling invertebrates abundance (t = -2.39, p = 0.018, Tab. 3.6, Fig. 3.17). This was driven by a significant increase in abundance in the Cv treatment (t-ratio = 2.78, p = 0.026, Appendix Tab. B.3), following a strong increase in one ground beetle species, *Pterostichus madidus*, in the Cv treatment of Field 4 during the second field of sampling. All other treatment combinations showed no significant impact on pitfall invertebrates counts. Cover crop did not have a significant effect on abundance after removing the treatment interaction term (t = 0.377, p = 0.707). Abundance in the field boundary was also a significant predictor of ground dwelling invertebrates in the field (t = 3.37, p = 0.001, Fig. 3.18). The variance explained by fixed factors in the model was 27.5% (Marginal R^2_{GLMM}), while the variance explained by fixed and random factors together was 93.1% (Conditional R^2_{GLMM}).

Ground beetles The GLMM of ground beetle abundance showed similar results to those emerged from the analysis of overall ground dwelling community, as they were the

Table 3.5: Estimated regression parameters, standard errors, and t-values for the quasi-Poisson GLMM on canopy dwelling invertebrates. The interaction term between Cover crop and Compost amendament was removed for model simplification. Boldface indicates statistical significance with $p < 0.05$. Df = 131.

Predictor	Estimates	SE	t	p
Intercept	2.44	0.57	4.24	< 0.0001
Cover crop	0.15	0.07	2.12	**0.036**
Compost	-0.14	0.08	-1.81	0.073
Point	0.30	0.07	4.32	**<0.0001**
Abundance in boundary	0.02	0.04	0.50	0.617
Vegetation cover	0.05	0.04	1.27	0.205
Penetrability	-0.15	0.04	-3.38	**0.001**
Organic matter	0.02	0.04	0.46	0.646
Removed				
Cover crop : compost	0.06	0.17	0.33	0.739

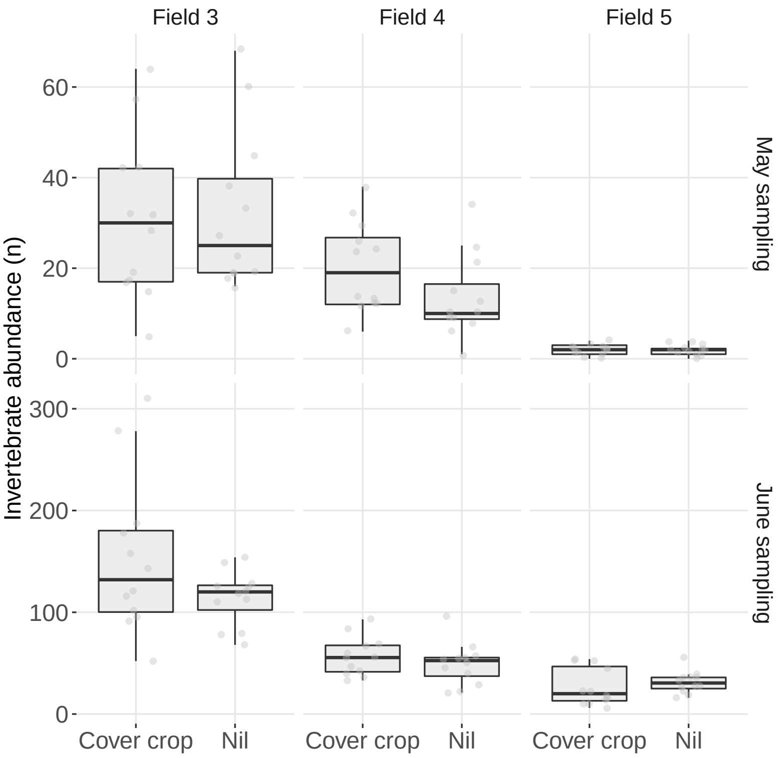


Figure 3.15: Abundance of canopy dwelling invertebrates collected with suction sampling in plots with and without cover crop during the May and June sampling weeks.

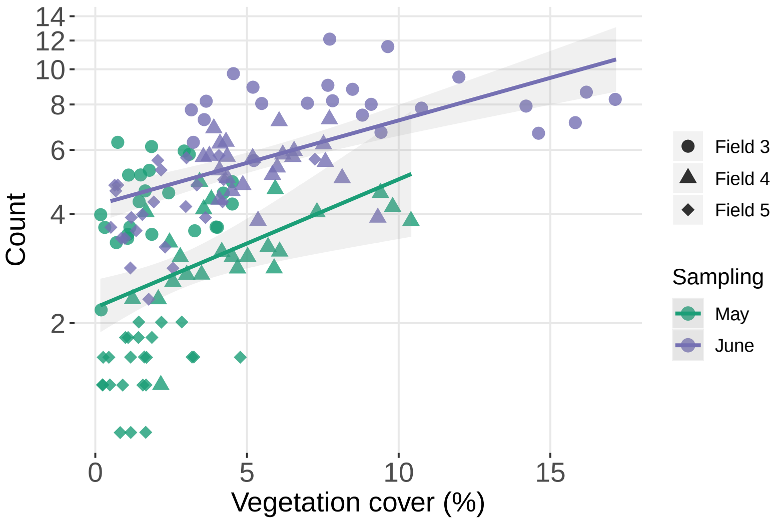


Figure 3.16: Abundance of canopy dwelling invertebrates and vegetation cover in the three study fields during the May (green) and June (purple) sampling. Mean estimates (solid line) and 95% confidence intervals (shaded area) of back-transformed data from log transformed data.

Table 3.6: Estimated regression parameters, standard errors, and *t*-values for the quasi-Poisson GLMM on ground dwelling invertebrates. Boldface indicates statistical significance with $p < 0.05$. Df = 120.

	Estimate	SE	t-value	P
Intercept	4.01	0.36	11.18	< 0.0001
Cover crop	0.37	0.15	2.39	**0.018**
Compost	0.19	0.15	1.27	0.205
Point	-0.21	0.11	-1.87	0.063
Abundance in boundary	0.17	0.05	3.37	**0.001**
Vegetation cover	0.10	0.09	1.13	0.261
Penetrability	0.01	0.10	0.10	0.918
Organic matter	-0.05	0.06	-0.88	0.379
Cover crop : compost	-0.65	0.22	-2.90	**0.004**

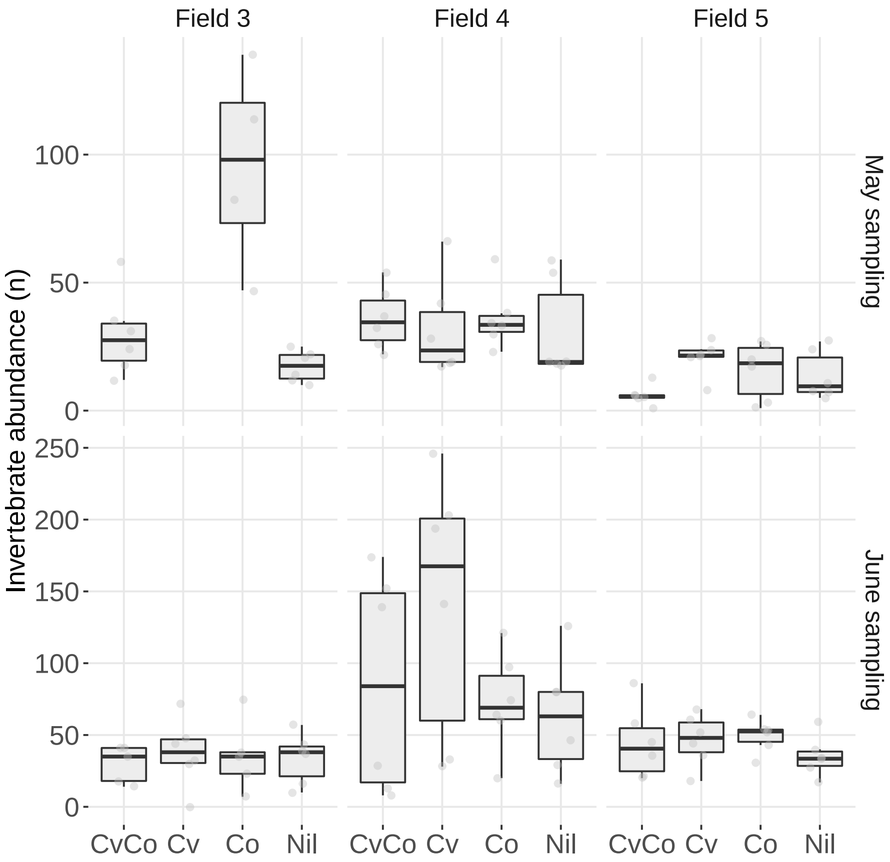


Figure 3.17: Abundance of ground dwelling invertebrates collected with pitfall trapping in each field in plots with and without cover crop during the May and June sampling weeks.

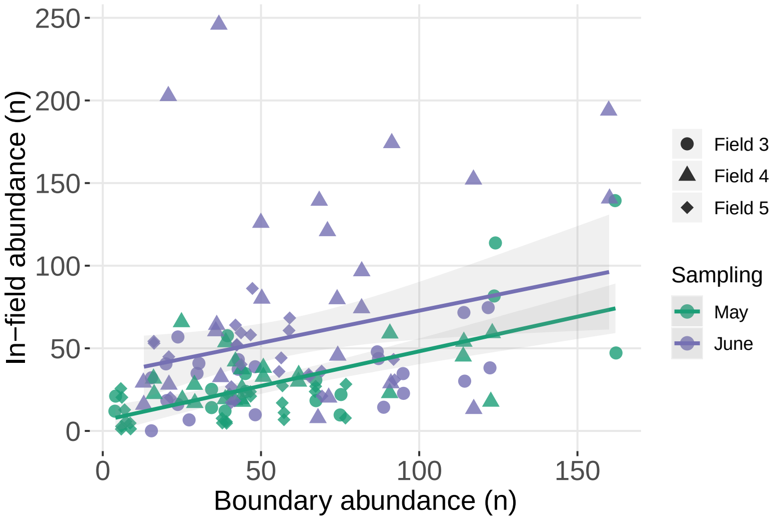


Figure 3.18: In-field and boundary abundance of ground dwelling invertebrates collected with pitfall trapping in the three study fields during the May (green) and June (purple) sampling. Mean estimates (solid line) and 95% confidence intervals (shaded area).

main component of ground dwelling invertebrates (Fig. 3.3). The cover crop $*$ compost interaction had a significant negative effect on *Carabidae* abundance (t = -2.56, p = 0.012, Tab. 3.7). *Post hoc* analysis showed that the Cv and Co treatments had significantly higher ground beetles abundance than the CvCo and Nil treatments (Cv p = 0.003, Co p = 0.046, Appendix Tab. B.4). These were, however, likely due to the localised increases in the population of *Harpalus rufipes* and *Pterostichus madidus* in Field 3 and Field 4, respectively (Fig.3.19).

Ground beetles decreased from the field edge towards the field centre. Larger invertebrate species, such as *Carabidae*, overwinter in the boundaries either as adults or larvae, and tend to remain more associated to them in the summer. Thus, abundance in the boundary was a significant predictor of overall ground dwelling invertebrates and of ground beetles number in the field. As seen in §3.3.3, ground beetle species present varying degrees of association to the field boundary. Some species rely on field margin habitats for resource availability (e.g. *Harpalus rufipes*, *Leistus rufomarginatus*), thus, their spillover in the field can be directly related to their abundance in the surrounding habitat. The opposite may occur for species that favour the field habitat, such as *Pterostichus* spp., and their abundance in the hunting ground of the arable field, may result in higher spillover into the field margin.

Distance from the field boundary had a negative effect on ground beetles abundance (t = -2.64, p = 0.009, Appendix Fig. B.6), and their abundance in the field boundary was a significant positive predictor of ground beetles in-field number (t = 4.22, $p < 0.001$, Fig. 3.20). Finally, there was a significant negative effect of soil organic matter content (t = -2.14, p = 0.034) The variance in the model explained by fixed factor was 26.1% (Marginal R^2_{GLMM}), while the variance explained by fixed and random factors was 95.2% (Conditional R^2_{GLMM}).

Table 3.7: Estimated regression parameters, standard errors, and *t*-values for the quasi-Poisson GLMM on ground beetles (*Carabidae*). Boldface indicates statistical significance with $p < 0.05$. Df = 120.

	Estimate	SE	t-value	P
Intercept	3.82	0.47	8.21	< 0.0001
Cover crop	0.47	0.18	2.56	**0.012**
Compost	0.01	0.19	0.03	0.972
Point	-0.36	0.14	-2.64	**0.009**
Abundance in boundary	0.27	0.06	4.22	**< 0.0001**
Vegetation cover	0.22	0.11	1.96	0.053
Penetrability	-0.07	0.13	-0.57	0.570
Organic matter	-0.16	0.08	-2.14	**0.034**
Cover crop : compost	-0.68	0.27	-2.56	**0.012**

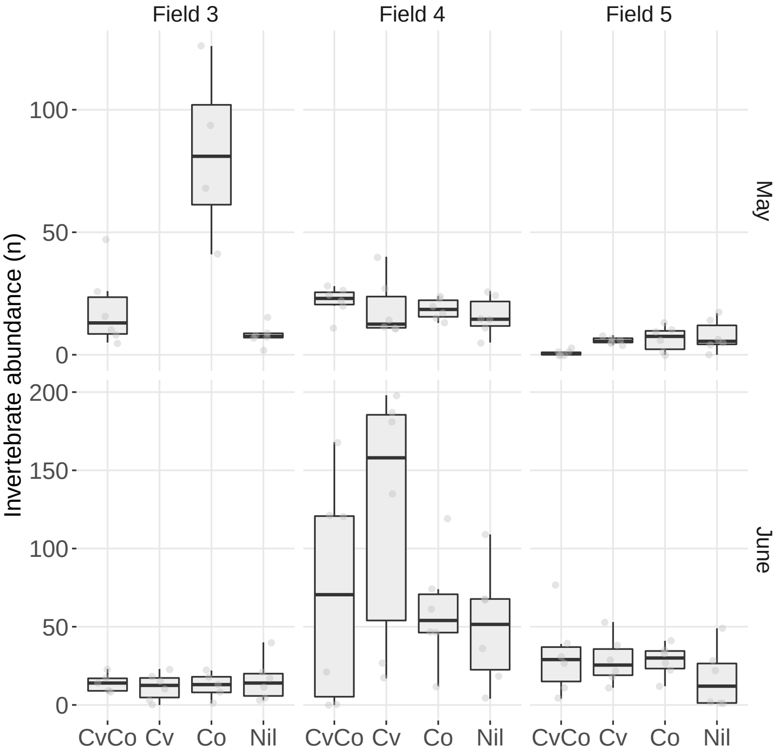


Figure 3.19: Abundance of ground beetles (*Carabidae*) collected with pitfall trapping in each field during the May and June sampling weeks. Cover crop and compost amendment (CvCo), cover crop (Cv), compost amendment (Co) and no sustainable soil management (Nil).

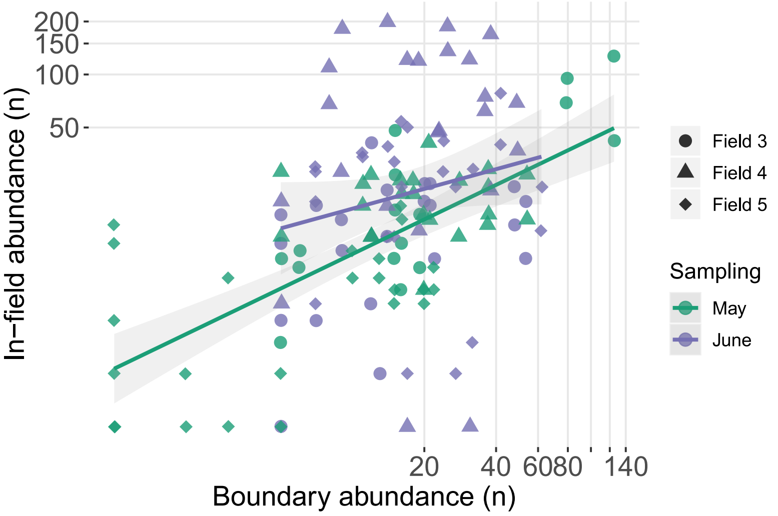


Figure 3.20: Abundance of ground beetles collected with pitfall trapping in the field and in the field boundary during the May (green) and June (purple) sampling. Mean estimates (solid line) and 95% confidence intervals (shaded area) of back-transformed from log transformed data.

3.4 Discussion

Soil health can be promoted in conventional arable systems by adopting soil management practices aimed at improving soil organic matter content, soil structure, and nutrient cycling (Garratt et al. 2018), with potential cascading effects for the aboveground biota (Bennett 2010). Although monitoring how invertebrate communities are affected by cover crops and green compost amendment provides an indication of their impact and sustainability, the potential of these practices to benefit invertebrate aboveground community are largely unexplored. This study investigated the assemblage and abundance of canopy dwelling and ground dwelling invertebrates in association with soil treatments and localised habitat characteristics of soil organic carbon, soil penetrability, and vegetation cover. While aboveground invertebrates exhibited differences among fields and sampling week, and along the sampling transects, little evidence was found of strong changes in aboveground invertebrates assemblage and abundance in cover crop and green compost amendment plots over traditionally managed ones. Cover crops, however, were shown to have a marginal positive association with canopy dwelling invertebrate numbers.

The findings of this study showed a positive association of cover crop treatment with the overall abundance of canopy dwelling invertebrates collected with suction sampling. There is some evidence that cover crops may increase the activity and survival of canopy dwelling invertebrates, such as parasitic wasps (Damien et al. 2017) or spiders (Hooks et al. 2011). Studies on canopy dwelling invertebrates and cover crops, however, have typically been conducted while the cover crop was present in the field (e.g. Hartwig and Ammon 2002; Mallinger et al. 2019), with modelled estimates (e.g. Schipanski et al. 2014; Prechsl et al. 2017), or with a primary focus on pest suppression effects (e.g. Teasdale et al. 2004; Damien et al. 2017). The results of this study suggest lasting effects of a cover crop mixture in the cash crop rotation, long after its termination.

While a positive legacy effect of cover crop was observed on the abundance of canopy dwelling invertebrates, their assemblage at the Order level remained homogeneous across soil treatments, indicating that cover crop did not favour any particular group at a broad taxonomic level within fields and weeks. Studies focusing on the pest control potential of cover crops have also shown that while cover crops can significantly increase the abundance of predators and predation rates, they do not alter their community composition, even when considering lower taxonomic levels, such as species or morphospecies (e.g. Lundgren and Fergen 2010, 2011).

Ground dwelling invertebrates abundance and assemblage did not show strong associations with the use of cover crop and green compost in the fields. Literature on cover crops and organic soil amendments effects on ground dwelling invertebrates –mostly surrounding ground beetles– is limited, and existing studies show contrasting results. Carmona and Landis (1999), for example, found no effect of cover crops on ground beetle across years in maize planted after a winter cover crop compared to plots with no cover crop treatment. Lundgren and Fergen (2010), instead, found a marginal effect of cover crop on ground beetles and spider abundance in maize plots with cover crop, and Groeneveld and Klein (2015) found higher herbivore – but not predator – ground beetle abundance and diversity in maize following a winter cover. Fewer studies investigate in-field changes in invertebrates associated with organic soil amendment. Zhang et al. (1998) found no effects of organic amendment (compost and manure) on *H. rufipes* distribution, while Idinger and Kromp (1997) found evidence of increased abundance of some Collembola and Diptera groups in plots fertilised with 80 t/ha of compost. Such high application rates, however, have also been associated with potential long term toxicity increase (Mbarki et al. 2008), as compost can cause an increase in heavy metal concentration, especially when sourced from industrial areas (Illera et al. 2000; Weber et al. 2007).

In this study, ground beetles, which represented 59% of pitfall catches and largely drove distribution patterns of ground dwelling invertebrates, were positively associated to the cover crop $*$ compost interaction. This result was caused by localised increases in the abundance of two species. Ground beetle often present highly aggregated distributions within arable habitats, which, due to their high mobility, can rapidly shift in a dynamic manner (C. F. G. Thomas et al. 2002). Here, a strong increase in *Pterostichus madidus* activity was recorded in the cover crop treatment of one field, during the June sampling. This species is a generalist predators very common in UK arable landscapes and known for exhibiting highly mobile and ephemeral clustering behaviour (Holland et al. 2004a), often following the distribution of cereal aphids (Winder et al. 2001; Holland et al. 2004b). *Harpalus rufipes*, a seed predator that has also been shown to be locally very active in the presence of suitable habitat conditions (Zhang et al. 1998), presented a spike in its distribution in the compost treatment of another field during the May sampling week. As these clusterings were temporary and localised, the significant effect of the interaction between of soil treatments was unlikely due to habitat characteristics emerging from the soil treatments, and rather caused by population dynamics of these species.

This study helps to document that the inclusion of cover crops in the cash crop rotation may have marginal positive legacy effects on the overall abundance of canopy dwelling invertebrates during bird breeding season, while green waste compost amendment was not associated with the aboveground invertebrate community abundance and composition.

3.5 Conclusions

Patterns of soil degradation in intensive arable systems have encouraged the development of sustainable soil management practices aimed at promoting soil health. Due to the links between belowground and aboveground biological communities, the introduction of SSM practices, such as cover crops or green waste compost amendment, has the potential to support greater abundance and diversity of aboveground invertebrates in conventional systems. This study investigated the assemblage and abundance of canopy and ground dwelling invertebrates in association with the use of cover crop and compost in a commercial arable farm during bird breeding season, when aboveground invertebrates represent an important dietary component for most farmland bird species. Cover crop was shown to be positively associated with overall canopy dwelling invertebrates abundance, suggesting that the presence of cover crops in the cash crop rotation may have beneficial legacy effects on aboveground invertebrates. Ground dwelling invertebrates, however, did not exhibit changes in abundance or assemblage among soil treatments.

Chapter 4

Effects of sustainable soil management on chick food availability

4.1 Introduction

Farmland birds have undergone extensive population declines throughout Europe (Donald et al. 2001c; Jerrentrup et al. 2017), and one of the hypotheses for the reason of this decline is the reduced invertebrate prey availability during breeding season as a result of intensive agricultural practices (Benton et al. 2002; Hart et al. 2006; Douglas et al. 2012; Holland et al. 2012; Schrauth and Wink 2018; B. M. Smith et al. 2020). Most farmland birds dietary requirements encompass a continuum between insectivore and granivore diets, i.e. their main source of nutrients derives from eating some combination of invertebrates and seeds. During the early stages of development, however, invertebrates represent a vital dietary component of most farmland species, as they supply the necessary protein needed to grow and resist chilling (Martin 1987; Southwood and Cross 2002). Poorly fed wild chicks have been shown to have a lower chance of survival in adverse weather conditions, and to be at higher risk of predation (A. D. Evans et al. 1997).

It is not, however, only the mere abundance of invertebrates that contributes towards chick survival, but also the availability of a diverse assemblage that allows for the selection of an optimal nutritional combination. A mixed diet, whether largely selected by the chicks (i.e. precocial birds) or exclusively supplied by the parents (i.e. altricial birds) is likely to provide a more balanced nutrient intake (Krebs and Avery 1984; Borg and Toft 2000). The diet of farmland birds has traditionally been studied with faecal or neck-collar analyses, due to the impracticality of feeding birds known diets. These dietary studies, however, can be biased against soft-bodied taxonomic groups, such as springtails or Hemiptera nymphs (Moreby 1988; Robinson et al. 2018); because of the high digestibility of these insects (G. Castro et al. 1989), their contribution in the diet could be underestimated (Moreby and Stoate 2001). Moreover, dietary studies may reflect the availability of invertebrates present in the study site. Finally, extensive dietary requirement studies are present for many, but not all birds commonly found in UK's lowland farmland (Wilson et al. 1999).

The available methodologies for estimating bird food availability in arable landscapes are somewhat limited, as they are exclusively based on selected dietary studies of a few number of species. Some Chick Food Indices have been proposed in the literature, most famously the grey partridge CFI (Potts and Aebischer 1991), which was developed when computational methods in ecology were still limited. Most commonly, agroecology studies estimating breeding season food availability measured it as the overall count of invertebrate taxa known to be eaten by birds, using more or less detailed taxonomic identification (e.g. Wood et al. 2013; Holland et al. 2014). This conservative practice can exclude taxa which may be under-represented in dietary studies. Moreover, when considering the availability of broad taxonomic groups that present a wide spectrum of morphological traits, such as Orders, prey selection by birds based on visual clues may be underestimated by this method. Visual clues are known to play a role in birds' choice of invertebrates prey (Stephens and Barnard 1981; Turner 1982; Moreby et al. 2006; Svádová et al. 2009), however, the only trait commonly used in chick food availability studies is prey size.

This chapter proposes a method to compute an invertebrate Chick Food Index that takes into account available dietary studies of farmland birds and includes taxa that may have been be under-represented in such studies, but that may still be available to birds in impoverished arable landscapes. It incorporates knowledge of breeding birds' dietary requirements (Gibbons and Dawson 2004) with prey palatability based on traits of colour, size, and movement speed of invertebrates (Moreby et al. 2006). The methodology can be applied to any number of bird species with available dietary requirements information, and takes into consideration different prey availability to altricial and precocial birds in terms of the size of ingestible prey items.

The invertebrate Chick Food Index was calculated for forty species commonly found in UK's lowland farmland, and used to investigate if combinations of SSM practices, namely cover crops and green compost amendment, were associated with changes in invertebrate chick food availability. This study was conducted at the RSPB-owned Hope Farm, a commercial arable farm on clay-rich soil, that since 2015 has managed three experimental fields under a split-plot management regime to assess the potential of SSM to benefit declining farmland bird populations.

4.2 Methods

4.2.1 Chick Food Index definition

The **Chick Food Index** definition presented in this chapter is calculated for a selected group of bird species. This method assigns a CFI score to each invertebrate taxon of a particular life stage with described colour, size, and speed traits based on:

- the dietary requirements of the bird species
- the palatability of physical traits of that taxon for the bird species.

The CFI score of a taxon is then multiplied by the number of individuals of that taxon in the invertebrate sample and summed across all taxa found in a sample to obtain the total CFI score of that invertebrate sample.

The Chick Food Index was computed using information on the importance of selected invertebrate taxa for the individual bird species considered, and by compiling a physical score for each remaining taxa weighing the palatability of their physical traits. Dietary information was extracted from the most complete RSPB literature review on breeding farmland birds diet available to date, an unpublished report compiled by Gibbons and Dawson (2004), while physical traits palatability scoring was based on a study of food preferences for grey partridge *Perdix perdix* chicks by Moreby et al. (2006). In order to score these physical traits, given the absence of quantitative information about the ranking of their categories, assumptions were drawn on the relationships between colour, size, and speed categories and their palatability.

4.2.1.1 Dietary requirement table

The dietary information review compiled by Gibbons and Dawson (2004) covered the breeding season diet of bird species that breed in UK's lowland farmland (Tab.4.1) and was built on previous publications by Wilson et al. (1996) and Wilson et al. (1999). These lowland species can be found at Hope Farm, where, between December 2018 and February 2019 a total of 54 species were observed during winter bird surveying (RSPB 2019). Dietary requirements of all forty species were included in the Chick Food Index computation. The review, which collated published and unpublished literature, was originally used to inform the Farm Scale Evaluation of genetically modified herbicide resistant crops on the availability of farmland bird resources (e.g. Gibbons et al. 2006). Eighteen of the forty species included are farmland specialists that are part of UK's Farmland Bird Indicator (FBI). The FBI is based on breeding season counts of nineteen common and widespread native farmland specialists, and is measured as the average trend in their relative abundance (DEFRA 2015). Only one FBI species, the kestrel *Falco tinnunculus*, was not included in the review. Although the FBI species are routinely used as representative

Table 4.1: List of bird species considered in the chick food index. All species are farmland specialist breeding in UK's lowland farms. Shaded rows highlight species included in the Farmland Bird Index.

Common name	Scientific name	Code	Passerine	Diet	Chicks
Blackbird	*Turdus merula*	B	P	I	A
Bullfinch	*Pyrrhula pyrrhula*	BF	P	H	A
Carrion crow	*Corvus corone*	C	P	O	A
Chaffinch	*Fringilla coelebs*	CH	P	H	A
Cirl bunting	*Emberiza cirlus*	CL	P	H	A
Collared dove	*Streptopelia decaocto*	CD	NP	H	A
Corn bunting	*Miliaria calandra*	CB	P	H	A
Dunnock	*Prunella modularis*	D	P	I	A
Goldfinch	*Carduelis carduelis*	GO	P	H	A
Greenfinch	*Carduelis chloris*	GR	P	H	A
Grey partridge	*Perdix perdix*	P	NP	H	P
House martin	*Delichon urbica*	HM	P	I	A
House sparrow	*Passer domesticus*	HS	P	H	A
Jackdaw	*Corvus monedula*	JD	P	O	A
Lapwing	*Vanellus vanellus*	L	NP	I	P
Linnet	*Carduelis cannabina*	LI	P	H	A
Magpie	*Pica pica*	MG	P	O	A
Meadow pipit	*Anthus pratensis*	MP	P	I	A
Mistle thrush	*Turdus viscivorus*	M	P	I	A
Pheasant	*Phasianus colchicus*	PH	NP	H	P
Pied wagtail	*Motacilla alba*	PW	P	I	A
Red-legged partridge	*Alectoris rufa*	RL	NP	H	P
Reed bunting	*Emberiza schoeniclus*	RB	P	H	A
Robin	*Erithacus rubecula*	*R*	P	I	A
Rook	*Corvus frugilegus*	*RO*	P	O	A
Sand martin	*Riparia riparia*	SM	P	I	A
Skylark	*Alauda arvensis*	S	P	H	A
Song thrush	*Turdus philomelos*	ST	P	I	A
Spotted flycatcher	*Muscicapa striata*	SF	P	I	A
Starling	*Sturnus vulgaris*	SG	P	I	A
Stock Dove	*Columba oenas*	SD	NP	H	A
Stone curlew	*Burhinus oedicnemus*	TN	NP	I	P
Swallow	*Hirundo rustica*	SL	P	I	A
Tree sparrow	*Passer montanus*	TS	P	H	A

Continued on next page

Table 4.1 – continued from previous page

Common name	Scientific name	Code	Passerine	Diet	Chicks
Turtle dove	*Streptopelia turtur*	TD	NP	H	A
Whitethroat	*Sylvia communis*	WH	P	I	A
Woodpigeon	*Columba palumbus*	WP	NP	H	A
Wren	*Troglodytes troglodytes*	WR	P	I	A
Yellow wagtail	*Motacilla flava*	YW	P	I	A
Yellowhammer	*Emberiza citrinella*	Y	P	H	A

Order: Passeriformes (P) and others (NP); adult diet: herbivore (H), insectivore (I), omnivore (O); Chicks development type: altricial (A), precocial (P).

of the breeding farmland bird community in the UK, their inclusion in the indicator is due not only to their farmland habitat requirement for breeding, but also because they historically present sufficient distribution data throughout the UK (DEFRA 2015). Aside from the FBI species, the review by Gibbons and Dawson (2004) used in the compilation of the CFI included a further twenty-two species that are not exclusively farmland breeders, but are nonetheless typical species found in British lowland farmland. Of the forty species in total, thirty-five have an altricial development, and five are precocial.

Chicks development type is a spectrum, but we can broadly identify:

- **Altricial** chicks that hatch in an almost embryo-like state and need to be fed in the nest by the parents, which select all of their food items until fledging;
- **Precocial** chicks that are relatively mature and mobile from the moment of hatching and do not rely on food provisioning by the parents, which can instead point them towards food sources during excursions.

The breeding diet literature reviewed any evidence found of the consumption of an invertebrate taxon by each of the bird species considered. In the dietary requirement table invertebrate taxa identified to different taxonomic levels were marked as either ‘important’ or ‘present’ (C.2, C.3). An invertebrate taxon was deemed important for a particular bird species in the review if it was reported in the literature as comprising at least 5% of the species’ diet, or if particular attention was drawn to it in the literature as being significant in some way. The latter could include, for example, the importance of aphids for grey partridge, as self-selection experiments have shown that the consumption of small quantities of aphids carries valuable nutritional properties for the chicks (Borg and Toft 2000).

‘Important’ taxa were taken into account in the Chick Food Index computation (see details in section 4.2.1.3), while information on the ‘presence’ of a taxon in the diet of a bird was disregarded during the computation. This was due to the nature of the literature synthesis of the dietary information table, which did not allow to draw a distinction between the absence of information and the evidence of presence of invertebrate taxa in the diet of birds. Invertebrate taxa names from the original table were updated to adhere to current nomenclature.

4.2.1.2 Palatability scoring of physical traits

If an invertebrate taxon was not classified as 'important' for a particular bird species in the dietary requirements table, it was assigned a physical palatability score based on its colour, size, and movement speed. The scoring was formulated to consider all invertebrates as potential prey items, while taking into account their individual attractiveness to birds as prey based on their physical characteristics. The main reference point for the development of the physical scoring system was the publication by Moreby et al. (2006), who conducted an extensive feeding trial using the thoroughly studied grey partridge *Perdix perdix* chicks as a model predator species, and insects with different physical characteristics of colour, size, and movement speed as prey items in paired choice experiments. Grey partridge is a precocial bird, but given the absence of a comparable study on feeding choice of altricial birds, the same ranking of physical characteristics was assumed for both precocial and altricial species for colour and speed. Size was instead scored differently based on physical limitations of the prey items that are available to precocial chicks.

Within each physical trait, trait categories were scored based on their palatability. Curves were drawn based on assumptions made on the relationship between physical traits and their palatability. Each curve displays the graph of a function with the physical trait on the x axis going from 0 to 1. The palatability (y axis) was also set to range from 0 to 1, with 1 indicating the highest level of palatability, and 0 the lowest level. Physical traits categories were placed uniformly along the x axis. For example, for the trait colour, red was assigned the interval from 0 to 0.2, blue from 0.2 to 0.4, and so on. The physical palatability score of each category was calculated as the average of the curve in the interval assigned to that category. The parameters of each curve were chosen such that the area under the curve from 0 to 1 totals 0.5. Because the score assigned to each category is the average of the curve on that interval, the average of all scores on all intervals is also 0.5. This ensures that the scores assigned to colour, size, and speed of each invertebrate have comparable magnitude. The curve underlying the scoring of each category are shown in Fig.4.1, while their functions and exact scores are shown in Fig.C.2 and Tab.C.1.

The physical scores of prey items for altricial (`Phys.A`) and precocial birds (`Phys.P`) were computed as the geometric mean of the score for colour, size, and speed of the prey (eq.4.1). Thus, if one of the three physical scores was zero (i.e. the invertebrate was not a suitable prey due to one of its characteristics), the final physical score was also zero. Moreover, the use of geometric mean ensured that the three scores contributed equally to the final physical palatability score.

$$\begin{aligned}\texttt{Phys.A} &= \sqrt[3]{\mathtt{S}_{\text{colour}} \times \mathtt{S}_{\text{size(altricial)}} \times \mathtt{S}_{\text{speed}}} \\ \texttt{Phys.P} &= \sqrt[3]{\mathtt{S}_{\text{colour}} \times \mathtt{S}_{\text{size(precocial)}} \times \mathtt{S}_{\text{speed}}}\end{aligned} \tag{4.1}$$

Colour Moreby et al. (2006) ranked the palatability of seven colours with paired selection feeding trials the palatability. Colour preference was shown to be greatest for green invertebrates, followed by yellow, buff, brown, black, blue, and, finally, red ones. This ranking reflected familiar patterns of colour preference in insectivorous birds, which are commonly known to find particularly palatable green insects, such as Heteoptera, Orthoptera, or Lepidoptera larvae, while they are often averse to red coloured insects, such as *Coccinellidae*. It is worth noting that the observed aversion towards red insects is not reflected by an avoidance of red fruit, which is usually very attractive to birds, and that colour choice for fruit is generally quite different to that for invertebrates (Willson and Comet 1993; Schmidt and Schaefer 2004). Invertebrates, thus, were classified based on the colour categories defined by Moreby et al. (2006) based on their most prominent colouration. One category, metallic green, was added to the ones ranked by Moreby et al. in the computation of the Chick Food Index to distinguish dark coloured beetles with green metallic reflection from monochrome ones. The scoring of colours was based on the assumption of a linear relationship between colours and palatability.

Size Birds are known to select preferentially –but not exclusively– larger prey items (Turner 1982; Quinney and Ankney 1985; Whitmore et al. 1986; Moreby et al. 2006). Size palatability was scored differently when considering altricial and precocial birds, as altricial birds have the possibility of food preparation by parents, which allows chicks to have access to large prey items. For example, large adult ground beetles have been observed in the diet of small passerine nestlings such as yellowhammers *Emberiza citrinella*, corn buntings *E. calandra*, and yellow wagtail *Motacilla flava* (Brickle and Harper 1999; Moreby and Stoate 2001; Gilroy et al. 2009). These large invertebrates, instead, are physically inaccessible to precocial chicks that swallow food intact (Moreby et al. 2006; Potts 2012). On these bases, two different scoring systems were developed for precocial and altricial birds. Invertebrates were classified as very small ($\leq$ 1mm), small (1–3 mm), medium (3–6 mm), large (6–9 mm), and very large ($>$ 9mm). Very small invertebrates were deemed disregarded by both precocial and altricial birds (unless otherwise specified by the dietary information table, see §4.2.1.3), while very large invertebrates were estimated to be inaccessible only to precocial chicks.

The relationship between prey size and palatability for precocial species was assumed to be a parabola, on which medium sized insect were the highest scored category. The size of ingestible prey items is likely to increase with the age –and thus body size– of the precocial chick, so that medium sized insects are likely to be accessible for the longest period of chick growth.

The relationship between prey size and palatability for altricial species, instead, was assumed to be linear between small and large species, and plateauing for very large prey items. This was due to the fact that it is likely that there is an –unknown– maximum size of prey items that can be successfully caught by the bird species considered.

Speed Invertebrates can be broadly categorised as either fast or slow moving. For example, agile flyers, most predators, and invertebrates that have the ability to propel by jumping can be defined as 'fast'. Invertebrate taxa were here categorised as either 'slow' (e.g. all larvae, Heteroptera, *Curculionidae*, Opiliones) or 'fast' (e.g. *Carabidae*, *Staphylinidae*, adult Diptera, *Delphacidae*, Araneae). Moreby et al. (2006) found that slower moving prey items were preferred to faster moving ones. The relationship between palatability and speed was based on the assumption that very slow moving insects would be very accessible to birds, very fast moving insects may be hard to catch, and that most invertebrates, however, would fall between these two extremes. In order not to estimate a great disparity in the effort required to catch 'fast' and 'slow' moving invertebrates, given the broad categorisation of this trait, the relationship between movement speed and palatability was assumed to have a sigmoid distribution, rather than a linear one.

4.2.1.3 Taxon CFI computation method

The information in the breeding season dietary requirements table and the physical scores were combined using the algorithm shown in Fig.4.2. The algorithm integrates known breeding dietary requirements of selected bird species, and palatability of all invertebrate prey items available to them in the habitat considered. Different prey size requirements for precocial and altricial species are also considered. Given (1) any invertebrate field dataset for which each row corresponds to an invertebrate taxon with described colour, size, and speed traits, (2) the dietary requirements table for each of the species considered, and (3) the list of bird species, the algorithm computes the Chick Food Index score of that particular invertebrate taxon for the bird species considered.

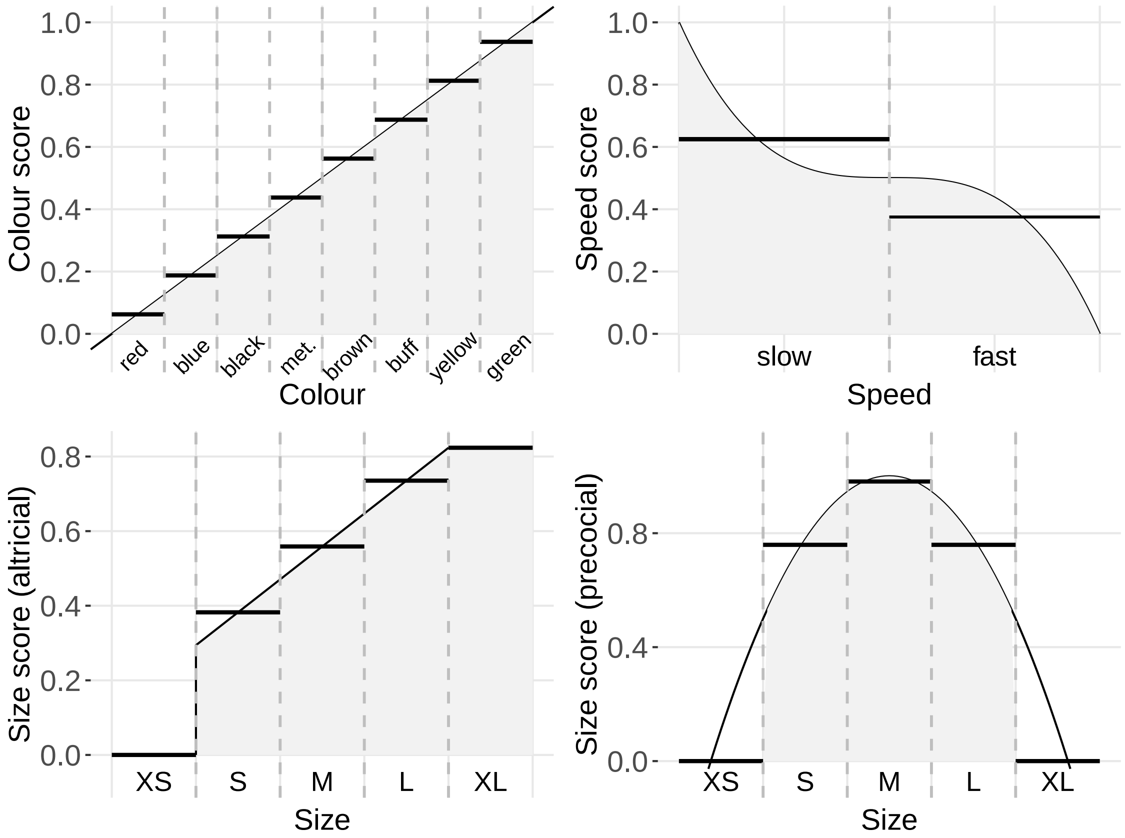
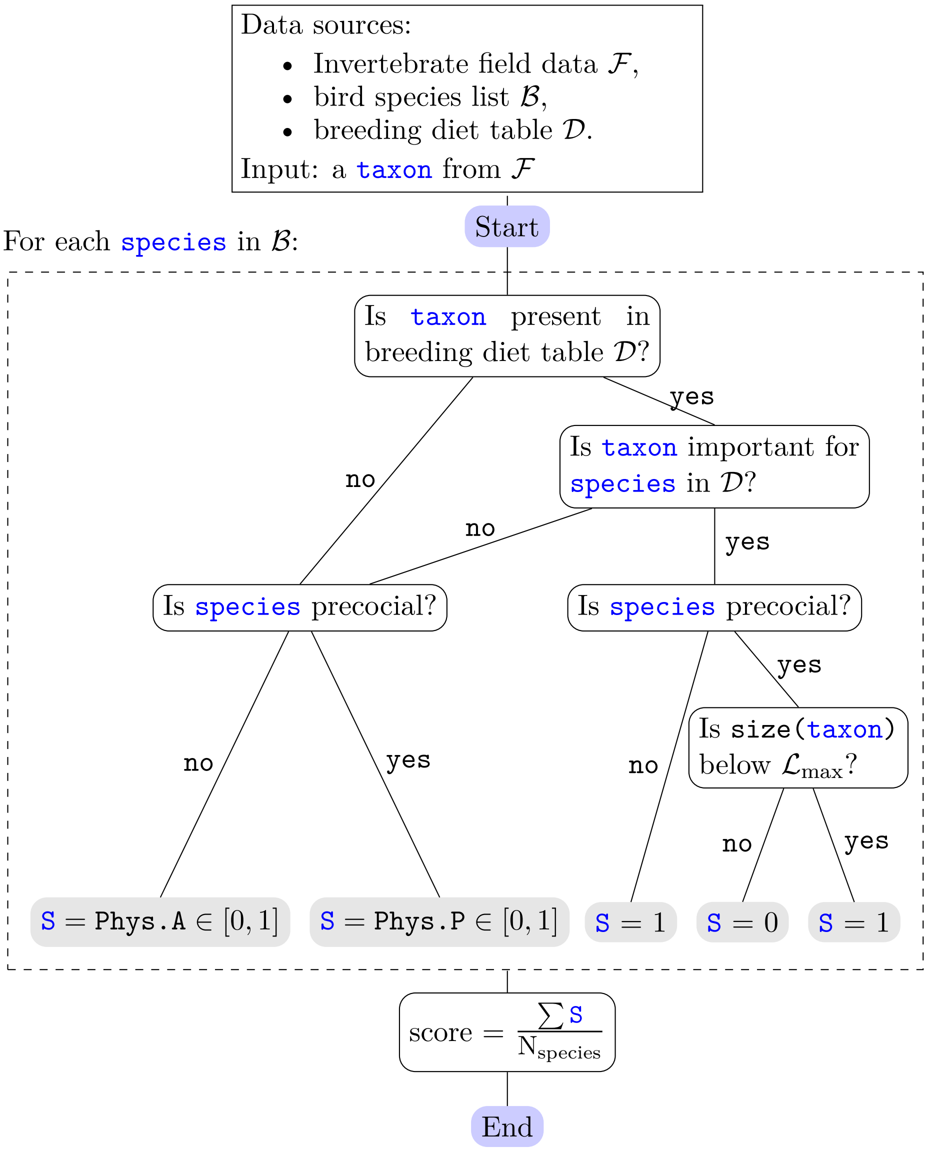


Figure 4.1: Curves used to assign scores to invertebrate physical characteristics of colour, size, and speed. The characterises were selected based on the selection study conducted by Moreby et al. (2006). Shaded area totals 0.5 for all curves.



`taxon`: a particular invertebrate taxonomic level and life stage,
`species`: a species in the list of bird species considered,
`Phys.A, Phys.P`: physical palatability score of `taxon` for altricial and precocial bird species (4.1),
$\mathcal{L}_{\max}$: maximum length of prey that can be ingested by `species`,
`S`: the physical palatability score of `taxon` for `species`.

Figure 4.2: Algorithm for the computation of the total palatability score of any invertebrate taxon across the forty farmland bird species considered in the analysis. Given an invertebrate with a set of physical characteristics (`taxon`) and a bird species (`species`), the computation calculates the invertebrate's palatability score for that bird (`S`). The invertebrate's score is calculated for each bird species, and averaged to obtain the total palatability score of that invertebrate. The algorithm accounts for the invertebrates' physical traits, its importance according to the dietary information table compiled by Gibbons and Dawson (2004), and the chick development type of the bird considered, as precocial birds present physical limits for the size of prey that they can ingest.

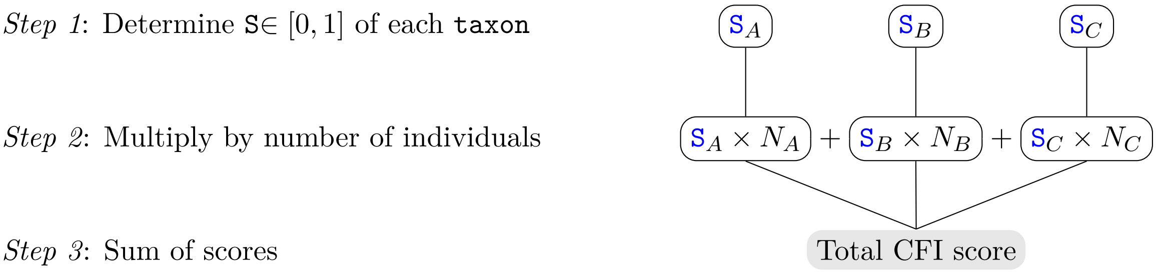


Figure 4.3: Steps to obtain total CFI score of an invertebrate sample from individual taxon CFI scores. If taxa A, B, and C were found in the field, their CFI scores (S) can be obtained with the algorithm previously shown in Fig.4.2. These scores are then multiplicated by the number of individual of that taxon found in the sample. Finally, all of these are summed to obtain the total CFI of that sample.

For each of the bird species considered in the CFI, each invertebrate taxon in the field dataset is searched within the diet table.

- If the taxon is absent in the diet table, or if it is present but it is not classified as 'important' for that bird species, the taxon physical score is computed as shown in eq. 4.1. This score ranges between 0 and 1 and is obtained using the information on the taxon colour, size, and speed. In the case that the bird species considered is precocial, the taxon physicals score is computed considering the chicks physical limitations for very large prey ingestion (`Phys.P`), while if the bird is altricial, the score reflects the ability of these species to access very large prey items (`Phys.A`).
- If the taxon is present in the diet table and it has been marked as 'important' for that bird species, it is given a score of 1 unless the bird species considered is precocial and the taxon is 'very large'. In this case, as the invertebrate is not accessible to the chicks of the species, it gets assigned a score of 0.

For each taxon, the scores obtained for all bird species considered are summed and averaged, to obtain the Chick Food Index score of that taxon.

4.2.1.4 From taxon CFI to total sample CFI

Once the Chick Food Index scores have been computed for each of the invertebrate taxa found in the dataset, the total CFI score of an invertebrate sample can be calculated. This total score is obtained by multiplying a taxon CFI score by the number of individuals of that taxon and, finally, by summing the CFI scores of all taxa found in the sample (Fig. 4.3).

Table 4.2: Selected examples of scores attributed to invertebrates with different characteristics and importance in breeding bird diet.

Class/Subclass	Order	Suborder	Family	Species	Life stage	Length (mm)	Colour	Speed	Score
Arachnida	Araneae				adult	8	brown	fast	0.75
Collembola					adult	2	buff	fast	0.53
Collembola					adult	1	buff	fast	0.10
Insecta	Coleoptera		Carabidae	*B. biguttatum*	adult	5	black	fast	0.61
Insecta	Coleoptera		Carabidae	*P. melanarius*	adult	23	black	fast	0.54
Insecta	Coleoptera		Chrysomelidae		adult	5	red	slow	0.43
Insecta	Coleoptera		Coccinellidae		adult	8	red	slow	0.34
Insecta	Coleoptera	Staphylinoidea	Ptiliidae		adult	1	black	slow	0
Insecta	Diptera		Tipulidae		adult	20	yellow	slow	0.71
Insecta	Diptera				adult	3	black	fast	0.37
Insecta	Hemiptera	Heteroptera	Miridae		adult	6	green	slow	0.74
Insecta	Hemiptera	Heteroptera			nymph	3	green	slow	0.67
Insecta	Hemiptera	Sternorrhyncha	Aphididae		adult	2	black	slow	0.70
Insecta	Hymenoptera	Chalcidoidea			adult	2	black	fast	0.37
Insecta	Hymenoptera		Formicidae		adult	2	black	slow	0.64
Insecta	Hymenoptera	Symphyta			larvae	3	green	slow	0.70
Insecta	Lepidoptera				larvae	8	green	slow	0.89

4.2.2 Taxa CFI scores for the selected bird species

The methodology illustrated in §4.2.1.3 can be adopted for any number of farmland bird species for which breeding diet requirements information is available. The Chick Food Index that was calculated for Hope Farm and used to investigate the potential association of SSM with bird food availability was based on all forty lowland farmland bird species shown in table 4.1, as reviewed by Gibbons and Dawson (2004). While this method can be applied to a small number of species of interest by reducing the dietary input data in the bird species list, for the scope of this analysis all bird species were selected, and the resulting CFI scores are calculated based on prey palatability estimates for all English farmland bird species.

The palatability scoring method captured the varied morphology of invertebrates in the fields, and successfully integrated physical trait scores with the dietary requirement table information. Table 4.2 shows an array of common invertebrates found in the fields, with different physical traits and importance according to the breeding diet requirement table (C.2, C.2) and their relative palatability score. Known favourite prey items, such as Lepidoptera larvae that featured as important for the diet of twenty-two of the forty considered, were assigned a high score. Other larvae with similar morphology, such as Symphyta larvae, also obtained a high palatability score due to their desirable physical traits.

Highly non-palatable traits were also well reflected by the scoring system, with, for instance, adults of the miniscule featherwing beetle family (*Ptiilidae*) receiving a score of 0. For other less desirable prey traits, such as red colouring, the importance outlined by the breeding diet table was determinant in their final score ranking. For example, while a large adult ladybird (*Coccinellidae*) was assigned a score of 0.34 as it was important to only two bird species, a similarily coloured and sized leaf beetle (*Chrysomelidae*) was given a score of 0.43, as its family was of importance to eight species instead. A taxon dietary importance also strongly emerged when presenting non-palatable traits. For instance, small and dark coloured aphids, that provide important nutrients to a wide range of species (nineteen out of forty) were given a much higher score than the morphologically similar, albeit not important chalcid wasps (*Chalcidoidea*).

Within the same taxon, physical traits were successful in differentiating the palatability of individual prey items. In particular, ground beetles featured as important in the diet of fourteen species in the list, and, especially, of grey partridge, lapwing *Vanellus vanellus*, stone curlew *Burhinus oedicnemus*, and pheasant *Phasianus colchicus*, four of the five precocial species considered. At parity of colour and speed, an individual of very large adult ground beetle, such as *Pterostichus melanarius* was given a lower score than a medium-sized beetle, such as *Bembidon biguttatum*, as it was inaccesible to precocial birds due to its size. The inclusion of the dietary importance in the scoring system also alowed to have very small individuals of important taxa to count towards the total Chick Food Index score. For example, Collembola were important in the diet of four species out of forty, and contributed to the CFI even when very small. For this reason, their score increased considerably with their size (e.g. 0.1 for 1 mm, and 0.53 for 2 mm).

4.2.3 Analysis

Invertebrate data and habitat covariates The invertebrate abundance dataset described in Chapter 3 was used to compute the Chick Food Index scores analysed in this chapter. Aboveground invertebrates were sampled once in May 2018 and once in June 2018 across the three experimental fields at Hope Farm undergoing different soil management in a split-plot design, with combinations of cover crop (present or absent) and green waste compost amendment (present or absent), as shown in Fig.3.2. Within each plot, sampling occurred along three randomly placed transects at least 30 m apart and projecting from the field boundary towards the field centre (§2.2.2). Canopy dwelling invertebrates were collected using suction sampling and ground dwelling invertebrates were collected with pitfall traps in the field boundary and at 4 m and 32 m in the fields (§3.2.4). Measures of soil organic content, soil penetrability and vegetation cover were collected at each sampling point (§2.2.2).

Each invertebrate identified in the dataset was recorded with its colour and length measurement accurate to the nearest millilitre, and was then associated to the movement speed category ‘fast’ or ‘slow’. The Chick Food Index score of each invertebrate taxon was computed using the algorithm shown in §4.2.1.3, using the invertebrate dataset collected at Hope Farm, the breeding dietary requirement table reviewed by Gibbons and Dawson (2004), and the list of forty lowland farmland bird species considered in the review as inputs. The CFI computation output was the palatability score of each sampled taxon.

The CFI score of each taxon were multiplied by the number of invertebrates of that taxon found in one sample. CFI scores were then summed across sampling points (boundary, 4 m, and 32 m) to obtain the CFI score of each sampling point. Canopy dwelling and ground dwelling invertebrates were scored separately to reflect differences in their sampling technique.

Statistical analysis Linear Mixed Models (LMMs) were used in the analysis, with infield Chick Food Index obtained from canopy dwelling and ground dwelling invertebrate samples at 4m and 32m from the field boundary towards the field centre as response variables. Cover crop, green compost amendment, and their interaction were Sustainable Soil Management treatment predictors. The interaction between treatments was excluded from the model when non-significant. Distance from the field boundary along the sampling transect (4 m or 32 m), vegetation cover, soil penetrability, and soil organic matter were also included as covariates. *Field and sampling week* was used as a random effect to account for differences among fields and time of sampling that would not be otherwise included in the model. Models were fitted using `lmer` in the `lme4` package (Bates et al. 2014) and were defines as follows:

```
m = lmer (CFI ~
cover crop * compost +
point +
vegetation cover +
penetrability +
organic matter +
(1|field and sampling week)
).
```

4.3 Results

The average palatability score across all taxa observed was 0.50 ± 0.20 for canopy dwelling invertebrates collected with suction sampling, and of 0.59 ± 0.11 for ground dwelling invertebrates collected in pitfall traps. Overall, the average palatability score of taxa observed across both sampling methods was 0.54 ± 0.17.

Chick Food Index and Sustainable Soil Management

At Hope Farm, the average CFI score of along transects for canopy dwelling invertebrates collected with suction sampling was of 33.3 ± 17.9 in the field boundary, and 18.4 ± 19.7, and 24.8 ± 29.7 at 4m and 32m from the field edge towards the field centre, respectively. For ground dwelling invertebrates, the average CFI recorded along the transects were of 37.5 ± 25.0 in the boundary, 28.0 ± 25.7 at 4m, and 25.0 ± 20.8 at 32m from the field edge (Fig.C.1).

The model output showed no significant effect of cover crop or green compost amendament on the Chick Food Index of canopy dwelling invertebrate samples (Tab.4.3, Fig.4.4). As for the total invertebrate abundance model shown in chapter 3, distance along the transect and soil penetrability were significant predictors; sampling points further into the fields had higher CFI than points closer to the field edge, while higher soil penetrability measures, i.e. more compact soil, were associated with lower CFI scores.

Chick Food Index values for ground dwelling invertebrates, instead, showed a significant effect of the interaction term between the two Sustainable Soil Management treatments. Post-hoc pairwise comparisons with Bonferroni correction showed no significant difference between treatment combinations. It is likely that the significant interaction term was due to the localised strong increase of *Pterostichus madidus* during the second period of sampling in Field 4 (Fig. 4.4). All other predictors were not significant.

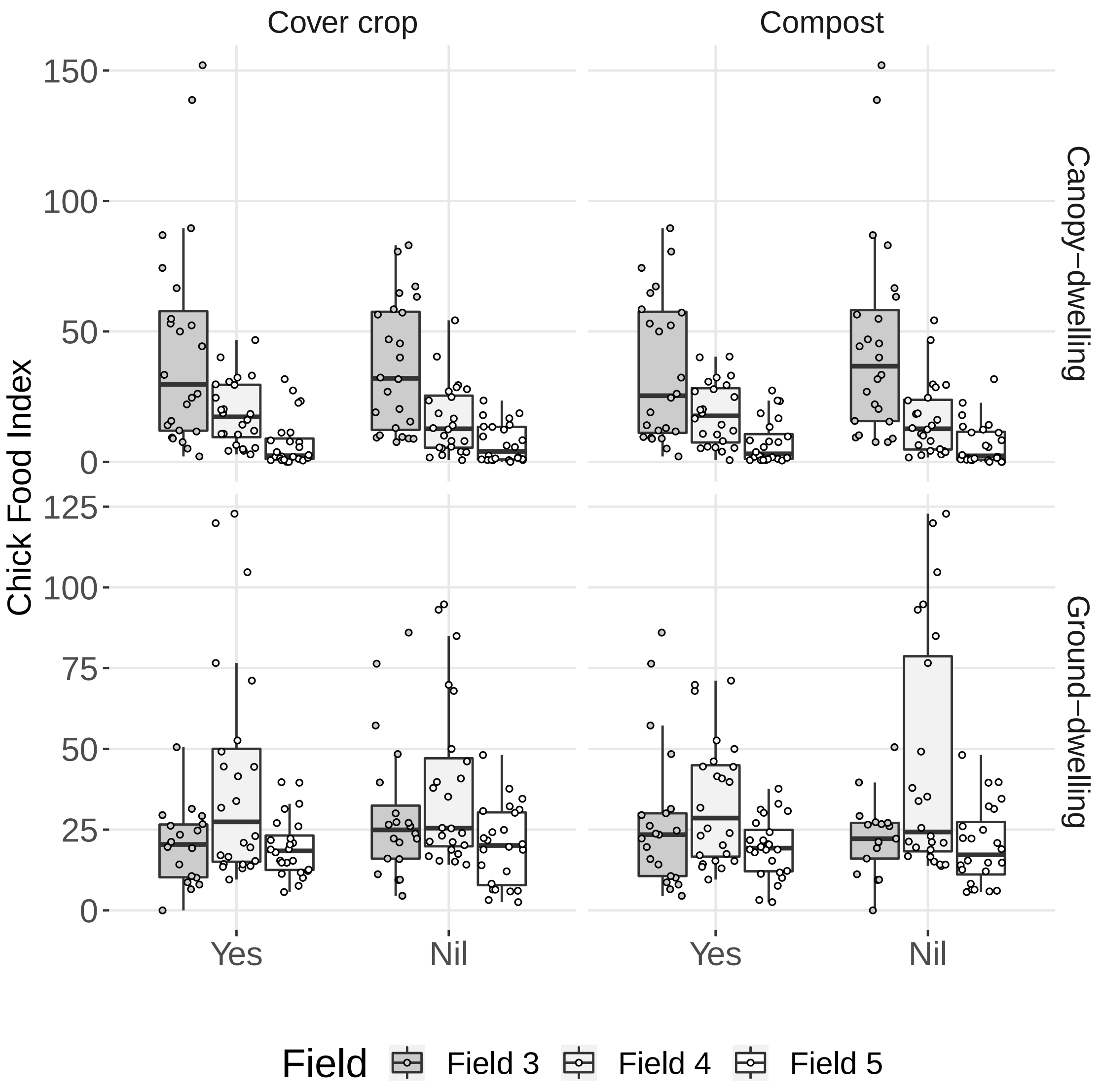


Figure 4.4: Chick Food Index scores recorded in the three arable fields at Hope Farm undergoing Sustainable Soil Management treatments.

Table 4.3: Estimated regression parameters, standard errors, and *t*-values for the LMM on Chick Food Index value obtained from canopy dwelling invertebrate samples. The interaction term between cover crop and green compost amendment was removed for model simplification. Boldface indicates statistical significance with $p < 0.05$.

	Estimate	SE	DF	t-value	P
Intercept	12.124	12.729	131	0.95	0.343
Cover crop	2.879	2.162	131	1.33	0.185
Compost	-2.972	2.328	131	-1.27	0.204
Point	5.175	2.319	131	2.23	**0.027**
Vegetation cover	6.936	4.566	131	1.52	0.131
Penetrability	-5.049	1.760	131	-2.87	**0.005**
Organic matter	0.093	1.501	131	0.06	0.950
Removed					
Cover crop : compost	-2.280	4.510	131	-0.51	0.614

Table 4.4: Estimated regression parameters, standard errors, and *t*-values for the LMM on Chick Food Index value obtained from ground dwelling invertebrate samples. Boldface indicates statistical significance with $p < 0.05$.

	Estimate	SE	DF	t-value	P
Intercept	46.84	11.92	121	3.93	**< 0.001**
Cover crop	8.02	4.85	121	1.65	0.101
Compost	6.61	4.78	121	1.38	0.169
Point	-6.27	3.42	121	-1.83	0.069
Vegetation cover	10.67	6.45	121	1.65	0.101
Penetrability	-1.04	2.73	121	-0.38	0.704
Organic matter	-2.86	2.21	121	-1.29	0.198
Cover crop : compost	-18.89	6.79	121	-2.78	**0.006**

4.4 Discussion

This chapter proposed a methodology for the computation of invertebrate Chick Food Index based on breeding season dietary requirements of forty farmland species commonly found in UK's arable lowland, as reviewed by Wilson et al. (1999) and Gibbons and Dawson (2004). The dietary information was combined with the computation of physical palatability scores based on invertebrates morphology traits of colour, size, and movement speed studied by Moreby et al. (2006). Moreover, the size of invertebrates was taken into account when considering precocial and altricial birds, as the two development types present different physical limits for the size of preys available to chicks.

The abundance of invertebrates in arable fields is not the only element contributing towards the availability of food to farmland chicks, as birds require not just an abundance of preys, but also a diverse assemblage in order to optimise their nutrient intake (Borg and Toft 2000; Liukkonen-Anttila et al. 2002). Most approaches found in the literature to assess food availability rely on abundance or biomass estimates based on dietary studies of selected farmland species (e.g. Wood et al. 2013; Holland et al. 2014; McHugh et al. 2018; Schrauth and Wink 2018). This approach is limited, as arable fields are environments mostly impoverished of invertebrates (Conrad et al. 2006; K. L. Evans et al. 2007; Shortall et al. 2009). Thus, it is likely that birds will have adopt an opportunistic feeding strategy, while still relying on visual clues to assess prey palatability. This chapter is an attempt at integrating the existing knowledge on dietary requirement of breeding farmland birds, with the palatability of all potential invertebrates preys found in the fields.

The Chick Food Index was computed for the invertebrate data collected at Hope Farm (chapter 3), to assess if the Sustainable Soil Management treatment plots were associated with different food availability for breeding farmland birds. Although CFI scores differed across fields and along transects, there was no evidence of strong patterns of increased chick food availability among soil treatments at Hope Farm. Cover crop and green compost amendment did not show any significant association with the availability of canopy dwelling invertebrate chick food obtained from suction sampling. There was a significant effect of the interaction between the two SSM treatments and Chick Food Index scores obtained from pitfall traps, however, post-hoc comparisons showed no difference among treatment combinations. The significant result was likely driven by the temporary and localised spike in abundance of *Pterostichus madidus* observed in one of the fields during the second week of sampling (§3.3.3). Although there is some evidence that the presence of a cover crop may attract birds when the cover is present in the field (Golawski et al. 2013; Wilcoxen et al. 2018), this increased use of arable fields with cover crops has not yet been linked with increased patterns of bird food availability. The findings of this study suggest that SSM did not play a role in increasing invertebrate food availability in arable fields to farmland bird chicks .

The CFI proposed here was successful in capturing the diversity of invertebrate assemblages and their palatability to birds. One of the main advantages of the index is that it can be used for any number or group of species for which some breeding dietary information is available. The index could, thus, be restricted for a smaller number of target species of interest. The index, however, could be improved further with the addition of quantitative information on the palatability of different morphology traits. In particular, the CFI assigns scores to different prey morphological traits based on –albeit conservative– assumptions on the relationship between these characteristics and their palatability. This could lead to over or under-scoring of traits. While Moreby et al. (2006) offers a qualitative ranking of traits on which the curves used in this CFI were based, there is no quantitative information available in the literature on the ranking of different morphological traits. While we know that that birds do not select exclusively based on colour, size, or speed, it is likely that prey selection is driven by a large degree by combination of these factors. This approach allows to estimate the palatability of invertebrate groups of unknown importance to breeding birds diet, and integrate it with established dietary requirements of farmland birds.

4.5 Conclusions

This chapter proposes a methodology to compute an invertebrate Chick Food Index integrating the existing knowledge on dietary requirement of breeding farmland birds, with the palatability of invertebrates preys found in the fields. Palatability was scored based on morphological traits of colour, size, and speed of preys. Moreover, the CFI takes into account the different limits to size prey accessibility of precocial and altricial birds. Breeding dietary requirements of forty species commonly found in UK's lowland farmland were computed for canopy dwelling and ground dwelling invertebrates collected along transects in arable fields undergoing combinations of SSM (i.e. cover crop and green compost amendment). This study found no strong evidence of an effect of SSM on invertebrate food availability in arable fields to farmland bird chicks.

Chapter 5

Effects of sustainable soil management on Skylark territory distribution

5.1 Introduction

Sustainable Soil Management (SSM) practices are being adopted increasingly in intensive arable landscapes to improve soil health and to, ultimately, increase soil fertility and productivity while decreasing the amount of chemical inputs applied within fields (Powlson et al. 2011). SSM has been introduced to European agri-environment schemes (Turpin et al. 2017), and is often promoted as beneficial for farmland avifauna, which has undergone extensive population declines throughout Europe (Donald et al. 2001c; Jerrentrup et al. 2017). As there some is evidence of an association between soil degradation and decreased abundance of farmland birds (Gilroy et al. 2008; Johnston et al. 2014; Eglington 2014; Felici et al. 2019), SSM may have advantageous repercussions on bird populations. These, however, remain poorly understood.

The effect of cover crops (i.e. break crops planted after cash crop harvest and terminated in early spring) and green waste compost (i.e. soil amendment from botanical cuttings), on farmland birds are largely unexplored (Hepperly et al. 2009; Kaspar and Singer 2011). No studies have yet been conducted linking compost amendments to farmland birds, while the few existing studies of cover crops and farmland birds are specific to seed-rich cover crops for game birds (e.g. Chris Stoate et al. 2003; Perkins et al. 2007), or to different farming system (e.g. J. Castro et al. 2014 for olive groves). Moreover, these studies have taken all place in winter or early spring before the start of birds' breeding season, when the cover crop is in the field (Golawski et al. 2013; Wilcoxen et al. 2018).

In this study, Skylarks were selected as model species to assess the effects of SSM on breeding bird territory settlement. The Skylark *Aulada arvensis* is one of the many farmland bird species that have suffered population declines in association with patterns of agricultural intensification, and its UK conservation status is red-listed (Eaton et al. 2015). While most other farmland passerine species nest in semi-natural habitat fragments and use fields for foraging, Skylarks are ground nesting and feeding birds. Thus, their behaviour can be directly associated to in-field management practices (e.g. Morris et al. 2004; Josefsson et al. 2013; J.-U. Schmidt et al. 2017), particularly in the light of the extensive literature on this species ecology (Donald 2004). Moreover, in spite of their decline, Skylarks are still widely distributed on farmland, and their small territory size makes them more suitable for field-scale analysis than other ground nesting farmland species with larger home ranges, such as Grey partridge *Perdix perdix*.

Hope Farm provides an exceptional insight into farmland bird responses to the adoption of sustainable agricultural practices, due to the longevity and consistency of species territory mapping, which has been taking place on the farm since 2000 (Morris et al. 2010), and the adoption of a SSM trial on the combination of cover crops and green compost amendment since 2015. During breeding season, passerine birds mark their territories by singing conspicuously, thus allowing for the mapping of their territories over the course of a number of field visits. Territory mapping requires high quality outline maps and is considerably more time consuming than other survey methods, such as line or point transects (Bibby et al. 2000). Therefore, the continuous Skylark monitoring at Hope Farm provides an unique, fine-scale insight of soil management practices effects on breeding birds.

In this chapter, a five year observational dataset of breeding Skylarks was analysed in the light of patterns of Skylark territory settlement on the farm for the previous seven years, with the aim to determine whether Skylark territory numbers were influenced by the adoption of two SSM practices (i.e. cover crop and green waste compost amendment) in arable fields at Hope Farm. Hope Farm is an RSPB-owned lowland commercial arable farm located in Cambridgeshire (UK) that since 2015 has been undergoing a SSM split-plot field trial on the potential benefits of cover cropping and green compost amendments on declining farmland bird populations.

One of the difficulties of territory mapping is that result comparison across studies is hard, unless common standards of territory analysis can be applied (R. D. Gregory et al. 2004). This study proposes a method for the automation of territory delimitation, to apply consistently mapping rules across datasets, removing interpretation error in time and space. Moreover, when investigating the effect of SSM at the field scale, it is essential to consider pre-existing drivers of territory settlement in the fields. Thus, the modelling of Skylark territories accounted for the suitable area of fields, a measured derived from the distribution of disturbance elements in the landscape (i.e. *nearness to human-made structures* and *nearness to tall vegetation*), and for fine-scale landscape quality features (i.e. crop sowing time and skylark plots). The methodology of this study, which illustrates automated territory delimitation from digitised survey maps, and habitat suitability optimisation based on species ecology, can be applied to future studies of sustainable practices on farmland birds in agricultural landscapes.

5.2 Methods

Breeding season skylark observations from eleven fields at Hope Farm between 2008 and 2019 were clustered into territories with an automated process following *territory mapping* rules (Bibby et al. 2000). The resulting dataset was split between a calibration dataset from 2008 to 2014, and an evaluation dataset from 2015 to 2019. The former was analysed to determine the best *Suitable area* predictor of in-field territory numbers at Hope Farm, through an optimisation process accounting for Skylark territory settlement drivers (Donald 2004). The evaluation dataset, presenting territory number by fields and by SSM treatment plots and accounting for *Suitable area*, was analysed to asses if SSM could be associated with the distribution of Skylark territories among soil treatment plots.

5.2.1 Skylark monitoring data

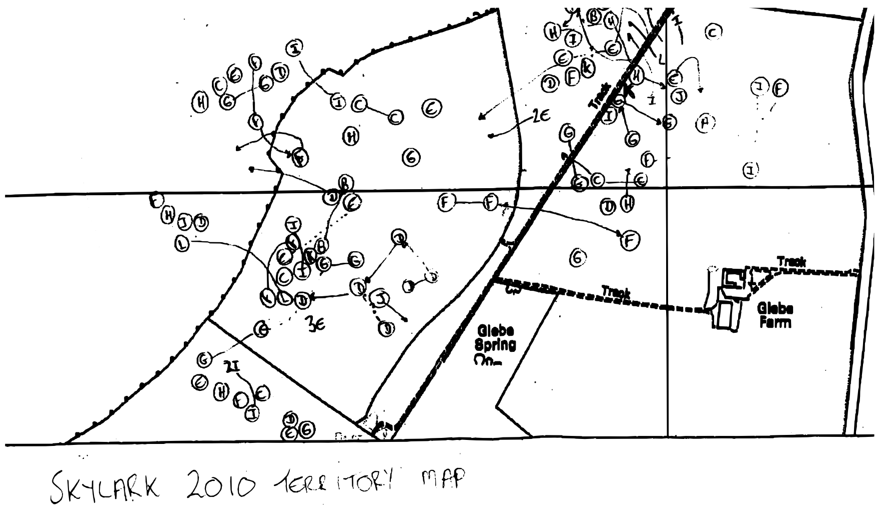


Figure 5.1: Example of a Skylark survey map at Hope Farm in 2010. Letters represent observations from different visits, and dotted lines join simultaneous observations of two singing males that are always attributed to different territories during territory mapping.

Breeding season skylark observation maps were obtained from Hope Farm (§1.4) archives for the twelve year period between 2008 and 2019. These maps were generated following the methodology for *territory mapping* (Bibby et al. 2000; Jablonski et al. 2010), with bird surveys carried out throughout the farm every year between April and July. In this census approach, surveys were carried out between 0600 and 1200 hours GMT, with a minimum of 10 days between surveys and avoiding suboptimal weather conditions, such as heavy rain, low visibility, and strong wind, as these tend to reduce skylark activity (Marchant 1990). During each survey, all contacts with birds, either by sight or sound, are recorded on an outline map (Fig.5.1). After all observations are recorded over the course of one season of surveys, the main aim of this method is to draw non-overlapping boundaries around cluster of observations, which are assumed to refer to one pair of breeding Skylarks. For this reason, an essential component of this survey method is to record the activity of individual birds on the survey map, particularly when simultaneous observations of singing males occur, as territorial males cannot coexist within the same territory, thus providing useful information about the approximate location of the territory boundaries (Bibby et al. 2000; R. D. Gregory et al. 2004).

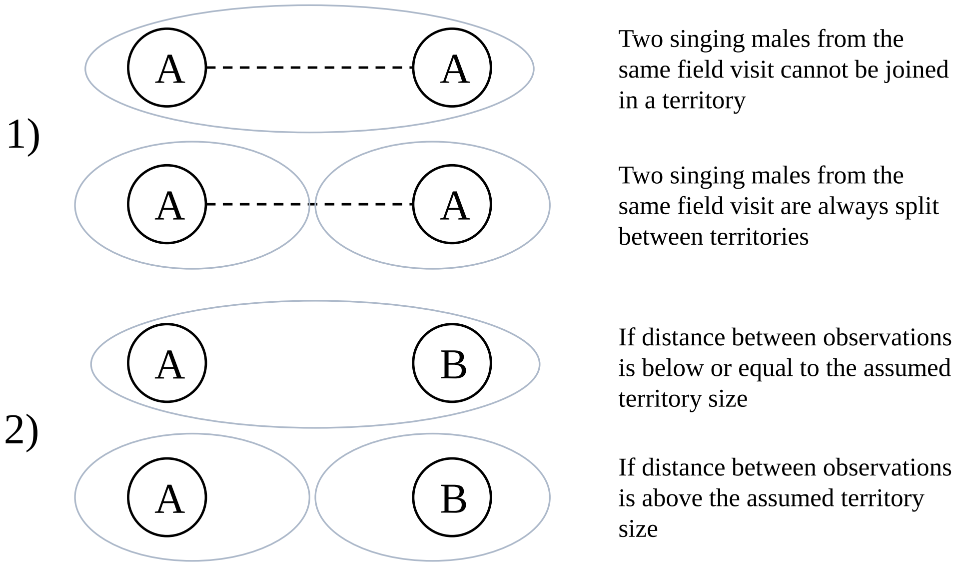


Figure 5.2: Visual representation of basic territory mapping rules, with circles recording observations from field visit, represented by letters, and ellipsoids representing territories. (1) Two singing males from the same field visit cannot coexist in the same territory and have to be attributed to different ones. (2) Two observations from any field visit can be attributed to the same territory if their distance is below the accepted territory size, which is determined based on species ecology. In the Skylark case, maximum accepted distance among points of the same cluster was set to be 200m (§5.2.1.2.)

5.2.1.1 Error in territory mapping

Territory mapping allows for fine-scale habitat associations to be studied within the study area, providing a high level of both identification and recording skills of the observer. Its monitoring method is accurate for territorial and monogamous species, and produces a detailed map of breeding birds distribution, however, this type of mapping can be subject to two main sources of error:

1. *Observational error* during surveys linked to individual differences among observers, weather conditions, habitat conditions, and distinctiveness of the species observed (MacKenzie et al. 2003).

2. *Interpretation error* when assigning individual observations to territories, resulting in different delimitations depending on individual judgment (Best 1975).

Issues arising from imperfect detection of organisms during surveys have been addressed in the literature with suggestions on how to analyse problematic datasets (e.g. Archaux et al. 2012; Kellner and Swihart 2014). Given the uniformity of observers' preparation in ornithological field data collection at RSPB Hope Farm, the relatively short time-frame between within-year surveys, and the dataset spanning multiple years, observational error was considered here to be negligible.

Interpretational error has long been known to be an issue with territory mapping, because of analysts' difficulty in maintaining consistent standards when interpreting distribution maps, a consistency that has to be maintained also over time (Bibby et al. 2000). This type of error has been historically controlled by training, and by requiring a minimum of three analysts to check the estimated territories (e.g. Wesołowski et al. 2010), a time consuming, and arguably impractical, solution.

Automating the process of assigning individual bird observations to clusters (i.e. territories) allows for the consistent application of clustering rules in time and space, however, there have been very few attempts to digitalise territory mapping. The main examples of automation applications are the `Autocluster` software used for breeding birds monitoring in the Netherlands –which has been criticised for the elevated number of rules implemented and has very limited English literature (van Dijk et al. 2013), a master thesis analysing the simplified bird monitoring system used in Switzerland (Schmid 2017; Wechsler 2018), and a publication specific to QGIS software (Kotowska et al. 2019). In this study, interpretational error was avoided by applying a generalised approach to automate the clustering of observations into territories.

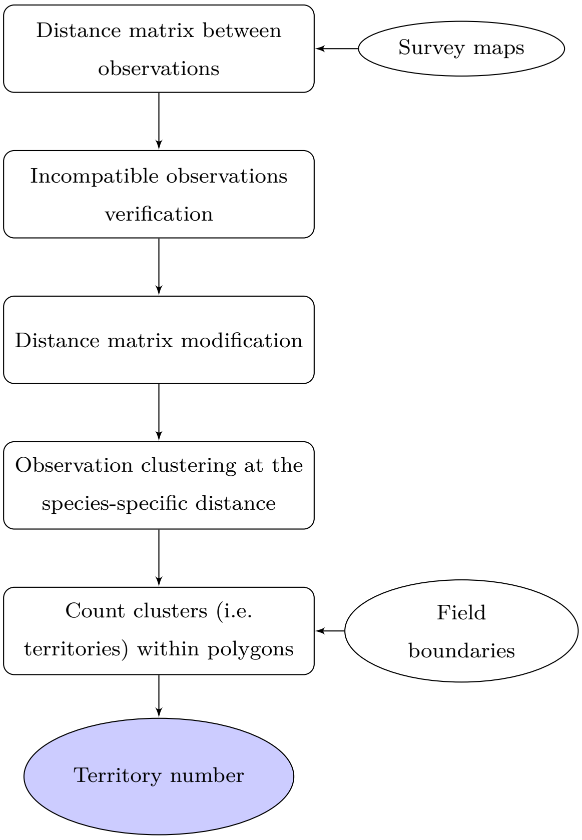


Figure 5.3: Flowchart showing the steps employed to obtain the number of territories from digitised maps of individual observations.

5.2.1.2 Territory number: automated territory delimitation

The **automatic territory delimitation** methodology presented in this chapter

1. calculates the distance matrix among observations;
2. alters the distance between males singing during the same visit, so that in the matrix they appear far more distant than they are on paper;
3. clusters observations that in the distance matrix are below a maximum territory size threshold;
4. counts the territories so obtained.

The objective of territory mapping is to obtain the number and location of breeding birds territories in an area by grouping individual observations from survey maps into not overlapping clusters of at least two observations, without multiple records of singing territorial males from the same visit (Bibby et al. 2000). An automated process to calculate territory number was developed to apply mapping rules consistently across each of the twelve survey map. The sequence of steps involved in this process are shown in Fig. 5.3.

Survey maps from 2008 to 2019 were individually digitised and all observations from each year were stored as spatial point dataframes (`.shp`). In this study, the digitisation of maps was conducted using the ArcGIS Desktop (Esri 2018), but any similar software package would suffice. Information regarding simultaneous sightings of territorial singing males were also included as attributes to be used to infer incompatibilities among clusters of observations, as two singing territorial males cannot be attributed to the same territory (Tab.5.1).

The first step in the automated process was to calculate an Euclidean distance matrix in meters between any two points based on easting and northing coordinates. The compatibility of individual observations was checked, and the Euclidean distances between incompatible observations (i.e simultaneous singing males) were modified accordingly. The modification ensures that points that cannot coexist in the same territory appear father apart than their true distance, and, thus, cannot be clustered in the same territory. In practice, this means that any singing male observation from the same day appears in the modified distance matrix farther apart than it was observed to be, so that any two singing male observations recorded on the same day may no be part of the same cluster (i.e. territory). The modified Euclidean distance between incompatible observations was set arbitrarily to be $10e10$ meters.

Table 5.1: Example of survey data storage for automated territory delimitation and respective compatibility matrix. Singing males are denoted with a bivariate variable (0=no, 1=yes). Letters represent different field visits. Singing males from the same visit (e.g. 3 and 4) are incompatible, while observations from different visits (e.g. 4 and 5) are *per se* compatible, as indicated by the compatibility matrix.

Bird observation ID	Singing	Visit
1	0	A
2	0	B
3	1	B
4	1	B
5	1	C

	Compatibility				
	1	2	3	4	5
1	–	✓	✓	✓	✓
2		–	✓	✓	✓
3			–	×	✓
4				–	✓
5					–

The clustering of observations was performed using complete linkage hierarchical clustering using `hclust` (Müllner 2013). This method, commonly applied when the number of clusters is not known *a priori*, defines the cluster distance between two clusters to be the maximum distance between individual components, and produces compact clusters (Romesburg 2004). In complete linkage clustering, each observation initially represents an individual cluster, and at each iteration of the clustering process the two closest clusters are merged. The process is repeated until all observations are clustered as one group

(Fig.5.4).

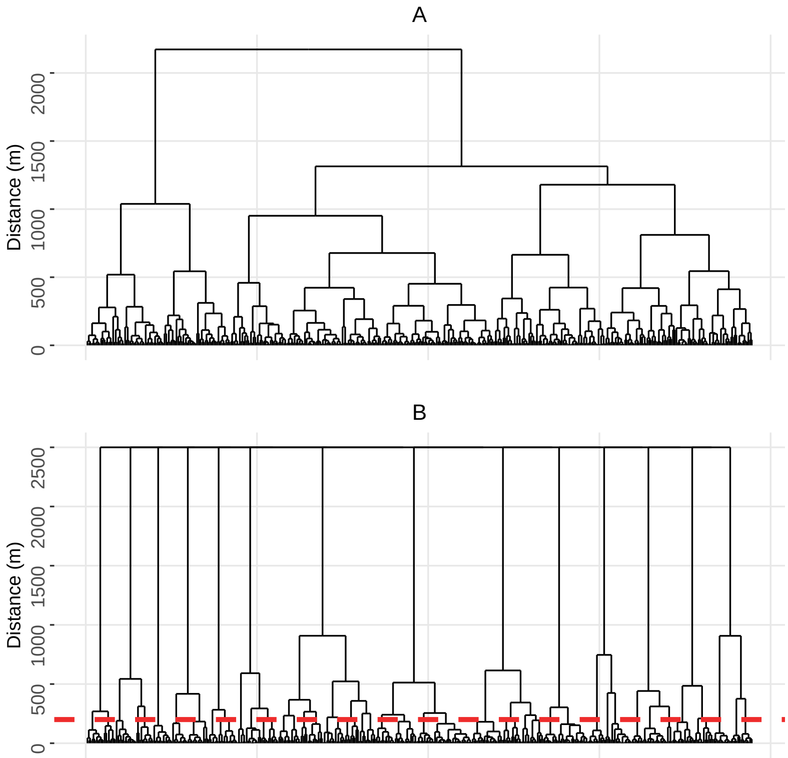


Figure 5.4: Dendrograms of original (A) and modified (B) distance matrix for the 2014 dataset obtained through complete linkage hierarchical clustering. Nodes at the lowest level represent individual observations of Skylarks at Hope Farm in 2014. The distance between incompatible observations is set here at 2500 for graph readability. The dashed line indicates the cutting distance, set at 200m. Observations linked below the cutting line belong to the same cluster (i.e. territory)

Following the clustering operation, the resulting dendrogram was cut at 200 m of height, and bird observations that remained grouped after the cut were assigned to the same territory. The distance of 200 m, which results in a maximum cluster size of 3.1 ha, was chosen based on the average proposed size for Skylark territories of 2-4 ha (Eraud and Boutin 2002; Donald 2004; Rahman et al. 2012), known provisioning trips distances for this species (Poulsen 1996), and for comparability with studies on other farmland passerine birds (e.g. Burgess et al. 2015). Clusters of single observations were discarded to satisfy the minimum requirements for territory mapping (Bibby et al. 2000).

The position of territory centroids was then calculated by averaging the easting and northing coordinates of the points of each cluster. This allowed to attribute each territories to a field or field subplot (i.e. soil treatment plot). Finally, the number of territories in each field was obtained by counting the number of territory centroids present in that field.

The following Python-style pseudocode shows in an artificial and informal text language the programming design of the algorithm implementing the automated territory count. This pseudocode is independent to the programming language used to implement the algorithm.

```
# data is a list of datasets indexed by year.
data :  list(dataframe)
# years is the index of data.
years : list(int)
# fields are the field boundary polygons
field_polygons : list(polygon)

# set_incompatible(df, distance_matrix) computes a modified distance matrix m, such that for observations i and j, if both i and j are singing at the same observation, m[i,j] == 10e10.
def set_incompatible(df, distance_matrix) :
    m  = distance_matrix
    df = filter(df, singing == 1)
    for i, j in df :
        if df$visit[i] == df$visit[j] :
            m[i,j] = 10e10
    return m

def count_territories(df, field_polygons) :

    # Compute the modified distance matrix
    m  = distm(df, method="euclidean")
    m = set_incompatible(df, m)

    # Compute the hierarchical clustering and cut the dendrogram at the established distance_treshold.
    hc = hclust(m, method="complete")
    distance_threshold = 200m
    df$clust = cutree(hc, h = distance_threshold)

    # Exclude clusters of n=1 observations and compute the centroids of each cluster
    for group in group_by(df, cluster)
        size = length(group)
        if size == 1 :
            df = drop(df, df$cluster == group)
        else :
            df$centroid.lat = mean(df$lat)
            df$centroid.lon = mean(df$lon)
    # Count territories within polygons, where each field is a polygon
```

```
    for field in field_polygons :
        count[field] = length(intersect(field, df$centroid))

    return (df, count)

for year in years :
    df = data[year]
    data[year], count[year] = count_territories(df, field_polygons)
```

5.2.1.3 Territory number: in practice

Skylark territory number and location were determined across eleven fields of Hope Farm between the years of 2008 and 2019. The dataset was split in two groups (Fig.5.6):

1. Habitat suitability calibration dataset spanning from 2008 to 2014, prior to the beginning of the SSM trial, with a total territory count by field and by year (n = 77).

2. Evaluation dataset from 2015-2019, with three fields subdivided into plots undergoing different combinations of soil management (i.e. cover crop, green waste compost, both cover crop and compost, and no sustainable soil management, see §1.4). The dataset presented a total territory count by field and by year for fields with no SSM, and a territory count by SSM subplot and by year for fields in the SSM trial (n = 100).

5.2.2 Components of breeding habitat suitability

Relevant landscape features in determining Skylark habitat suitability were identified in the literature, with the aim of analysing underlying patters of territory distribution at Hope Farm. The potential habitat requirement were split into two groups, one concerning disturbances that may cause active avoidance of particular areas, and one related to land use, which may influence habitat preference for nesting and feeding (Fig.5.5). Two main sources of disturbance in the establishment of territory were considered in the modelling:

- *Nearness to human-made structures.* As for most farmland bird species, human settlements (i.e. built-up areas, paved, and even unpaved roads), have been shown to decrease Skylark occurrence in agricultural landscapes (Benítez-López et al. 2010; Loretto et al. 2019). The degree to which these structures can impact birds number is variable. For example, busy roads may impact Skylarks for up to 1.5 km, while on quieter roads occurrences are depressed up to 100 m (Reijnen et al. 1996). Human

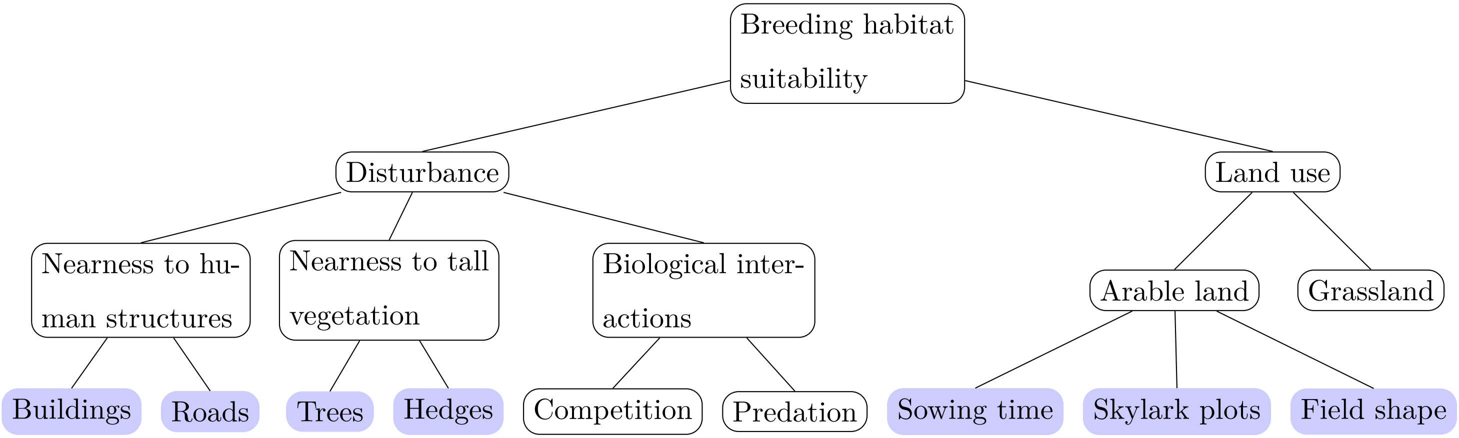


Figure 5.5: Drivers of Skylark breeding habitat suitability, with coloured blocks indicating variables considered in this study. Components of grassland habitat suitability were not considered, as Hope Farm is a conventional arable farm.

influence on the territory distribution at Hope Farm was defined as *nearness to settlements* and *nearness to roads*.

- *Nearness to tall vegetation*. Skylarks actively avoid nesting in proximity of trees or tall hedges, as these can be used by potential predators to locate nests on the ground (Donald et al. 2001a; Donald et al. 2001b). Skylarks suffer a very high rate of nest predation, and tall vegetation avoidance is so strong that ”even the best habitats may be completely unused by Skylarks if they occur too close to trees” (Donald 2004). The undesirable aspect of tall vegetation was decomposed in the model as *nearness to trees* and *nearness to hedges*.

- *Boundary index*, a hybrid disturbance variable of human structures and tall vegetation, was also calculated. This index was developed by Wilson et al. (1997) to describe the type of boundaries placed to delimit fields. It is calculated as:

  $$\text{Boundary index} = B_0 + B_1 + 2 \times B_2 + 3 \times B_3 + 4 \times B_4$$

  Where B_0 is the proportion of field perimeter with no vertical structures, B_1 the proportion of fence, B_2 the proportion of hedge $<$2 m, B_3 the proportion of hedge $>$2 m or with isolated trees, and B_4 the proportion of field perimeter comprising woodland.

Aside from the indirect effect of proximity to tall vegetation, the direct effect of predator avoidance on territory distribution was not considered in the model, as this would have required territory maps of mammal and bird predators of Skylarks (e.g. kestrels), which were not available for Hope Farm. Similarly, while intraspecific disturbance in the location of territories was accounted for in the clustering (5.2.1.2), interspecific competition from birds within the same feeding guilds was not taken into account.

Within agricultural landscapes, Skylarks in cropland are present at higher densities than grassland (Browne et al. 2000). As Hope Farm is primarily a conventional arable farm, different preferences within grassland were not considered in this study. In arable landscapes, Skylarks have been shown to exhibit some crop preferences, which, however, can mostly been attributed to different sward height in time rather than crop characteristics *per se* (Hoffmann et al. 2018).

- *Sowing time* has been shown to have an impact on Skylark distribution in arable landscapes, a difference attributed to sward height differences between autumn sown and spring sown crops (Donald et al. 2001b; Hiron et al. 2012). Some preferential behaviour among specific crop types has also been shown, however, this emerges when looking at detailed temporal variations in Skylark densities throughout the course of breeding season (Chamberlain et al. 1999; Donald et al. 2001a; Hoffmann et al. 2016). As territories here were counted for each year, only sowing time was considered to have a potential effect on Skylark territory distribution.

- *Skylark plots* are small (16-24 m^2) unsown areas within winter cereals crop fields, with a recommended minimum density of two plots per hectare. This agri-environment practice was originally developed by the RSPB at Hope Farm to enhance Skylark breeding habitat suitability in autumn sown cereals, which are believed to be too dense for effective foraging by early summer (Morris et al. 2004; Morris 2006; Gruar et al. 2010; RSPB 2012). Morris et al. (2004) showed that Skylark plots attracted higher densities of Skylarks, particularly later in the breeding season, and that they increased chick production by 50% per breeding attempt.

- *Field shape* describes the shape of a particular field, a historical result of land use. It was developed by Donald et al. (2001b) to measure the length of boundaries per hectare of field, and is calculated as the ratio of field perimeter to the circumference of a circle with the same area as the field:

$$\text{Field shape index} = \frac{\text{Field perimeter}}{2\pi\sqrt{\frac{\text{Field area}}{\pi}}}$$

  The relationship between Skylarks and field shape is not as clear as with other land use related variables, as some studies have shown a preference for fields with a high boundary to area ratio (e.g. Donald et al. 2001b), while others found a preference for ratios ~0 (e.g. Whittingham et al. 2003), and some found no association (e.g. Boatman et al. 2010). Field shape index was included in the modelling, as the eleven fields at Hope Farm present different shapes.

5.2.3 Analysis

The analysis occurred in three steps, outlined in Fig. 5.6.

1. An initial, literature-based model following the analysis of Donald et al. (2001b), aiming to confirm the reliability of literature-guided predictors and determine the shape of the relationship between territory density and continuous variables at Hope Farm from 2008 and 2014.

2. A suitability model specific to Hope Farm, building on the previous model and optimised by *Nearness to human structures* and *Nearness to tall vegetation*, with the aim of developing a *Suitable area* variable for each field which best described the territory distribution between 2008 and 2014.

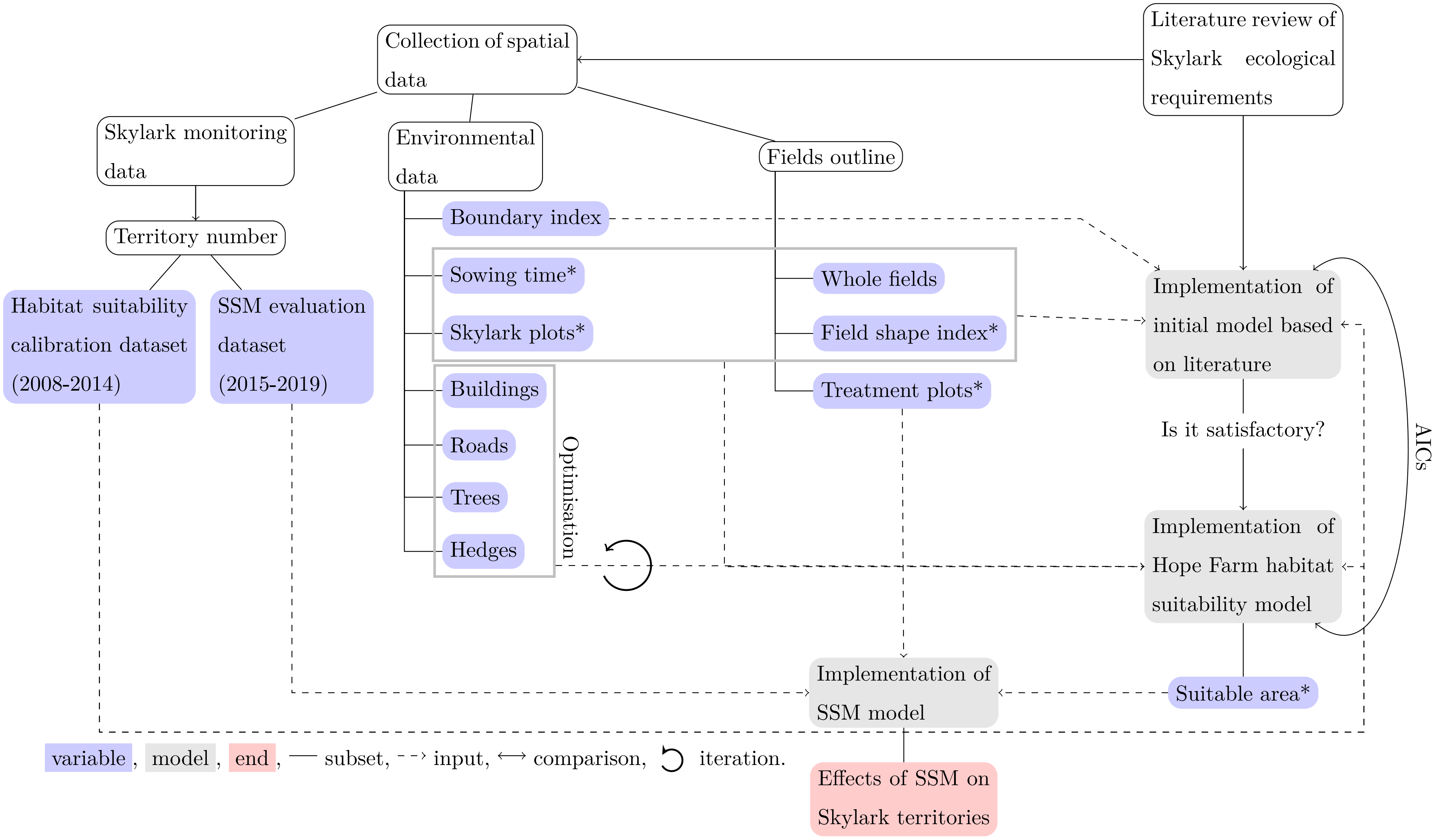


Figure 5.6: Framework of analysis of Skylark territory number at Hope Farm, showing a simplified model conceptual stage, and model quantification and application. Variables followed by an asterisks were included as predictors in the final sustainable soil management model.

3. The final model investigating the marginal effect of sustainable soil management on the territory distribution of Skylarks between 2015 and 2019, accounting for the *Suitable area* of each field and treatment plot.

Analyses were performed in R (R Core Team 2019), and all tests of null hypotheses were evaluated against a standard of $\alpha = 0.05$.

5.2.3.1 Literature based model

The starting point of the modelling was represented by a simplified version of an established approach to model Skylark territory distribution in UK arable lowland, rooted in literature ranging from the late 90s to the early 00s. Wilson et al. (1997), Chamberlain et al. (1999), Donald et al. (2001b) and Donald et al. (2001a), and Whittingham et al. (2003) investigated Skylark habitat selection in arable landscapes. These models analysed Skylark densities by month throughout the breeding season, in association with crop type and canopy structure, field area, shape, and boundary index.

A Generalised Linear Mixed Model (GLMM) was used to identify significant predictors of Skylark territory distribution at Hope Farm. Territory count in each field was the response variable, with field area, shape index, boundary index, crop sowing time (autumn or spring), and skylark plots (presence or absence) as predictors. Year was included in the model as a random factor, to account for fluctuations in population among years due to, for example, weather conditions. Quadratic terms were fitted in addition to linear terms for continuous variables, to determine the shape of their relationship with the response variable. The GLMM with Poisson error distribution and *log*link function was fitted using `glmer` in the `lme4` package (Bates et al. 2014).

```
literature based model  = glmer(territories ~
                  poly(field shape) +
                  poly(boundary index) +
                  poly(field area) +
                  skylark plots presence +
                  sowing time +
                  (1|Year)
                )
```

5.2.3.2 Hope Farm habitat suitability model

Once established if Skylark territory distribution at Hope Farm reflected patterns explained by field size and shape, management practices, and tall vegetation avoidance –as indicated in the literature, the second step of the analysis was to develop a habitat suitability model that best described the distribution of Skylarks among the fields. This GLMM included elements of disturbance in the landscape (*Nearness to human settlements* and *Nearness to tall vegetation*, see section 5.2.2) incorporated as a field area predictor by subtracting from the total field area the area subject to expected avoidance behaviour.

The **suitable area optimisation** methodology presented in this chapter

1. draws buffers of set widths around known disturbance elements of terriotry establishment in the landscape;
2. subtracts these buffers from the crop fields area;
3. inputs the remaning suitable area for territory establishment as one of the predictors in a GLMM model.

These three steps are repeated for all desired combinations of buffer widths around the disturbance elements. The AICs of the models so obtained are compared to find the best *Suitable area* predictor of bird abundance.

Suitable area definition GLMMs were used to determine the strongest *Suitable area* predictor at Hope Farm through an iterative optimisation. During this process, buffers of varying width were drawn around disturbance elements for Skylark territory settlement (i.e. buildings, roads, trees, and hedgerows) and subtracted from the total area of the fields. The resulting area was defined as suitable area and entered as one of the predictors in the GLMM. Buffers represented the distance from disturbance features at which territory presence is depressed within the fields. While buffers around different types of disturbance elements were allowed to vary independently, the distance at which each of them affected Skylarks was assumed to be uniform across the farm. For example, while buffers around trees were allowed to vary, the same buffer size was maintained at each iteration of the optimisation, meaning that a tree located in a particular hedge was assumed to have the same disturbance buffer width of a tree located at a different area of the farm. Buffer widths changed in steps of 10 m, and a maximum width was set for each disturbance element, to reduce computational requirements and to avoid the complete obliteration of fields by the buffers, as every field in the calibration dataset presented at least one territory between 2008 and 2015. The minimum width was set at 10 m for all features, while the maximum width was 170 m for buildings, 100 m for roads, 110 m for trees, and 50 m for hedgerows, for a total of $17 \times 10 \times 11 \times 5 = 9350$ GLMMs. The calculation of suitable area was an iterative process, during which the disturbance elements that were optimised for in the modelling were revised in the light of the optimisation results.

The area of each field resulting from the subtraction of buffers from the total area was defined as suitable area, and the best fitting GLMM output by the optimisation (i.e.

model with the lowest AIC) was deemed to present the best *Suitable area* predictor of territory number at Hope Farm. *Suitable area* is a theoretical artefact, as the width of buffers used in its calculation, although based on previous research findings (section 5.2.2), were not observed in the field and should not be interpreted as a true minimum territory settlement distance from disturbance elements. It was, however, the best area predictor in describing the distribution of Skylark territories in time at Hope Farm, allowing to adjust for in-field disparities when investigating marginal changes associated with sustainable soil management.

Model specification 9350 GLMMs were used to identify significant predictors of Skylark territory distribution at Hope Farm between 2008 and 2015. Territory count in each field was the response variable, with suitable area, field shape index, crop sowing time (autumn or spring), and skylark plots (present or absent) as predictors, and year as a random factor. Suitable area was log transformed in light of the findings of the literature based model. GLMMs with Poisson error distribution and *log*link were fitted using `glmer` in the `lme4` package (Bates et al. 2014). In the optimisation, models differed in the amount of suitable area calculated at each iteration, and the model with lowest Akaike's Information Criterion (AIC) was chosen as best fitting model. Best model residuals were checked using diagnostics plotting functions in the `DHARMa` package (Hartig 2019).

```
suitable area model = glmer (territories ~
                    log(suitable area)+
                    shape index +
                    skylark plots presence +
                    sowing time +
                    (1|Year)
                   )
```

5.2.3.3 Sustainable soil management model

A final model was formulated to test for responses of Skylark territory count data on sustainable soil management practices (i.e. cover crop and green waste compost amendment) adopted in three of the eleven fields at Hope Farm (§2.2.2), using the SSM evaluation dataset from 2015 to 2019 (Fig.5.6). The *Suitable area* predictor obtained from the habitat suitability model (§5.2.3.2) was used to control for pre-existing patterns of territory distribution at Hope Farm. The suitable area of each of the four SSM treatment plots in fields 3, 4, and 5 was obtained by overlaying the landscape disturbance elements buffers on the split-plot polygons, rather than the fields polygons, as done for the habitat suitability model. As the factor Year presented a very small estimated variance as a random effect, the effects of SSM on Skylark territory density were investigated using a Generalised Linear Model (GLM), with territory count as response variable, and cover crop, green waste compost, suitable area, field shape, skylark plots (presence or absence), sowing time (autumn or spring), and year as predictors. Territories between 2015 and 2019 from all eleven fields were used as response, with territory count calculated for each soil treatment subplots in the three SSM fields (n = 100) . The fit GLM had a Poisson error distribution with *log*link, and residuals were tested using `DHARMa` plots (Hartig 2019).

```
SSM model = glm (territories ~
            cover crop * compost +
            log(suitable area)+
            shape index +
            skylark plots presence +
            sowing time +
            year
          )
```

A linear model with territory density in the three SSM trial fields, calculated as number per hectare of suitable area, as response variable, and cover crop and green compost as predictors was also fitted.

```
SSM linear model = lm (territory density ~ cover crop + compost)
```

5.3 Results

5.3.1 Skylark territories at Hope Farm

The automated territory delimitation process identified 521 Skylark territories at Hope farm between 2008 and 2019, with an average of 43.4 territories per year (SD ±10.2) (Fig.5.7). The average territory density in the fields was 0.24±0.13 territories per hectare (4.2 ha per territory), and 0.99±1.78 territories per hectare of suitable area. The habitat suitability dataset between 2008 and 2014 showed an average count of 44 ± 13.3 territories in each field, and the SSM evaluation dataset 42.6 ± 4.67. Territory number was lowest in 2008 (n=21) and peaked in 2009 (n=64).

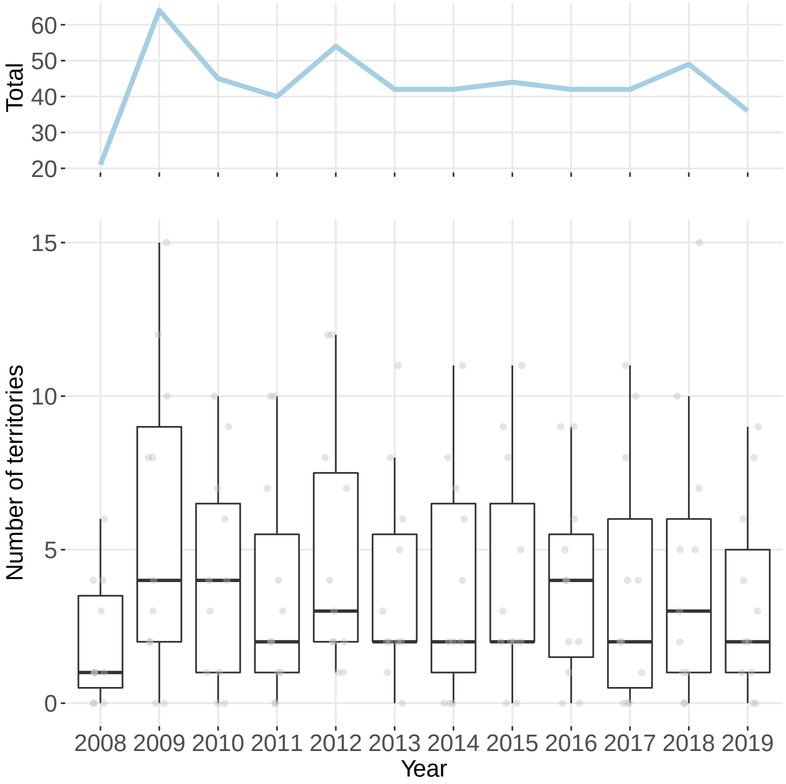


Figure 5.7: Number of Skylark territories at Hope Farm during each year of the study (2008-2019). The number of territories in the eleven fields was obtained with an automated territory delimitation process following rules of territory mapping. Continuous line (*top*) indicates total number of territories, points (*bottom*) indicate counts in individual fields.

Table 5.2: Estimated regression parameters, standard errors, and z-values for the Poisson GLMM of Skylark territory numbers in eleven fields at Hope Farm between 2008 and 2014, following the analysis of Donald et al. (2001b). Model AIC = 280.1, significant values are in bold.

	Estimate	SE	z-value	P
Intercept	0.83	0.17	4.88	**< 0.0001**
Field shape index	3.11	1.34	2.32	**0.020**
Field shape index^2	-1.33	0.99	-1.35	0.177
Boundary index	-3.81	0.70	-5.47	**< 0.0001**
Boundary index^2	1.53	1.19	1.29	0.198
Field area	8.23	0.99	8.33	**< 0.0001**
Field area^2	-4.76	0.87	-5.50	**< 0.0001**
Skylark plots	-0.07	0.19	-0.40	0.692
Autumn sowing	0.08	0.15	0.56	0.578

5.3.2 Literature based model

The results of the literature based model defined in §5.2.3.1 were in accordance with literature findings, indicating that Skylark territory distribution at Hope Farm exhibited patterns of tall vegetation avoidance, and preference for larger, compact, fields. Sowing time and skylark plots presence, instead, did not show a significant effect on the number of territories. Field area and boundary index emerged from this initial analysis as the main predictors of territory number (Tab.5.2). The parameter estimate of the field area predictor was always positive, while its quadratic term was negative, suggesting an asymptotic relationship with territories, conforming with the findings of Donald et al. (2001b). Boundary index showed a significant negative linear effect on territory number, in accordance with the findings of other studies using this variable to account for tall vegetation avoidance (Donald 2004; Copland et al. 2012). Field shape, instead, had a significant positive linear effect, suggesting a preference towards fields with a higher ratio of perimeter to area.

5.3.3 Suitable area for breeding Skylarks at Hope Farm

The habitat suitability GLMM defined in §5.2.3.2 showed that the best *Suitable area* predictor of Skylark territory number at Hope farm between 2008 and 2014 was obtained by subtracting from the fields buffers of 140m, 110m, and 50m around buildings, trees, and roads, respectively.

Optimisation performance The suitable area optimisation identified *nearness to trees* as the strongest predictors of Skylark territory numbers at Hope Farm, while *nearness to hedgerows* emerged as the weakest one and was excluded from the calculation. Fig.5.8 shows the components of the final suitable area optimisation, ordered by AIC score of the resulting GLMM. The 95% CI drawn around the smooth curves of buffer width for each feature indicate the variation in buffer size among models with similar AICs.

Generally, suitable area measurements derived from wider building and trees buffers, and narrow road buffers resulted in better fitting models than suitable areas calculated from narrow building and tree buffers, and wide road buffers. However, the degree of variation in buffer size around disturbance elements in the landscape varied, with deviations in tree buffer size (i.e. *nearness to trees*) having stronger effects on AIC scores than those around buildings and roads (i.e. *nearness to human-made structures*). The optimisation, thus, indicated a strong association between Skylark territory number in the fields and trees distribution around them.

Human settlements were associated with the widest disturbance buffer in the models, in accordance with the detrimental effect of urbanised areas on Skylark abundance (Loretto et al. 2019), while the low level of impact on territory number attributed to *nearness to roads* can be explained with the limited and mostly farm-related traffic occurring on local and unpaved roads in the area. Hedgerows were excluded from the suitable area calculation, as their uniform distribution and overlap with other landscape disturbance elements across Hope Farm (Fig.D.1), resulted in the lack of a measurable effect of their presence on territory number during the optimisation, as shown in Fig.D.3. It is worth reminding that the effect of both *nearness to tall vegetation* and *nearness to human structures* were restricted by a maximum buffer width to avoid the obliteration of the small fields located in the East part of the farm (Fig.5.11). Territory settlement may be depressed, at least to a certain extent, further than the buffer considered.

Habitat suitability model output The best fitting model emerging from the optimisation showed a significant positive association of fields' territory number and suitable area (z=11.22, P $<$ 0.0001, Tab.5.3). Field shape index did not show a significant effect on territory number after adjusting for in-field habitat suitability. As for the literature based model, the timing of sowing and the presence of Skylark plots did not have a significant effect on territory number. Overall, the habitat suitability model, with an AIC score of 270.0, showed a better fit then the literature based model (AIC=280.1), indicating that *Suitable area* was the best available predictor of Skylark territory number at Hope farm between 2008 and 2014.

There was a positive correlation between the size of the field and its suitable area (Kendall $\tau = 0.89$), however, the suitable area in each field varied, ranging from 54.3% of the total area, to 0.9% (Fig.5.9, Tab.D.1). The average territory density between 2008 and 2019 was of 1.13±2.28 territories per hectare of suitable area, compared to 0.25±0.18 per hectare of total field area.

Table 5.3: Estimated regression parameters, standard errors, and z-values for the Poisson GLMM of Skylark territory numbers in eleven fields at Hope Farm between 2008 and 2014. Model AIC = 270.0, significant values are in bold.

	Estimate	SE	z-value	P
Intercept	-2.03	1.25	-1.62	0.105
Field shape index	1.72	0.97	1.77	0.077
Suitable area	0.82	0.07	11.22	**< 0.0001**
Skylark plots	-0.14	0.18	-0.76	0.444
Autumn sowing	0.13	0.14	0.92	0.359

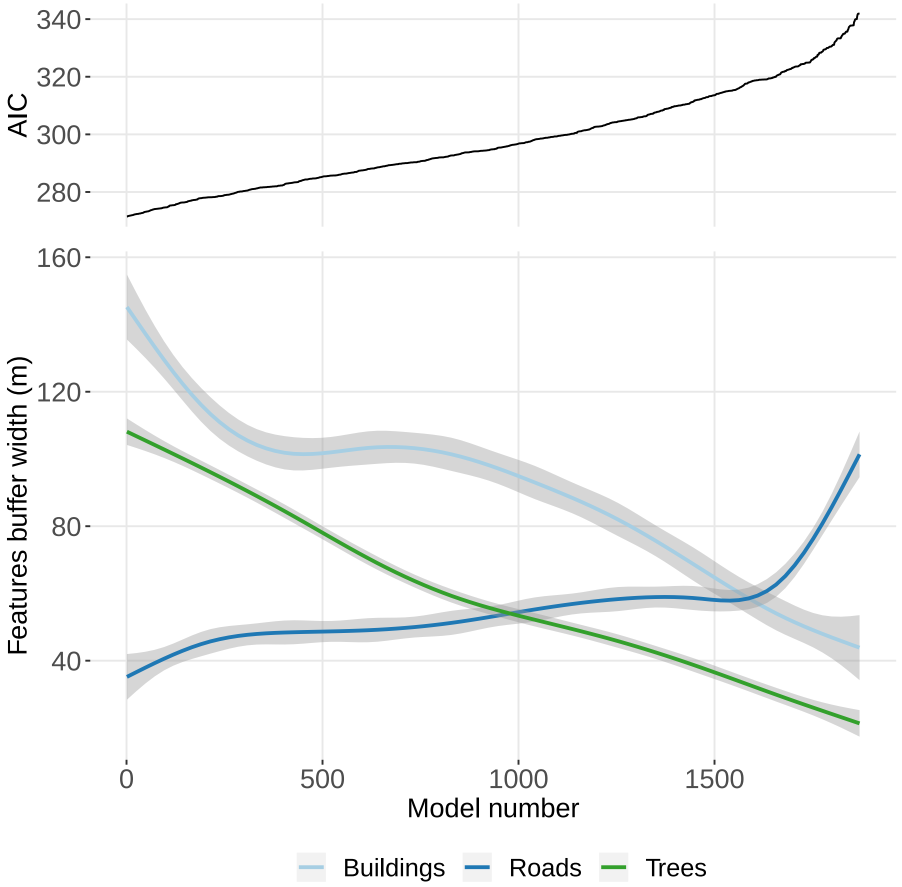


Figure 5.8: Smooth curve of buffer width around disturbance features during the optimisation, and AIC score of the respective model. Buffers were drawn around elements of disturbance for Skylark territory settlement and subtracted from the total field area to obtain a measure of suitable area for each field. The resulting GLMMs were ordered on goodness of fit to select the best *Suitable area* predictor for Hope Farm. Features lines indicate estimated values, and shaded areas the 95% CI, which represent the degree of variation in buffer width among neighbouring GLMMs (i.e. GLMMs with very similar AIC scores).

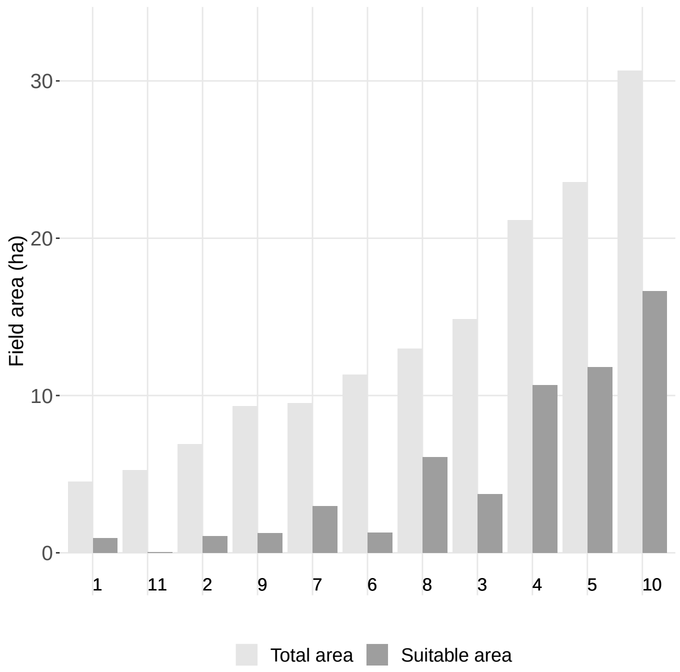


Figure 5.9: Total and suitable area of each of the eleven fields of Hope Farm, ordered by increasing total area (see MapD.2). Fields 3, 4, and 5 were part of the SSM trial starting in 2015.

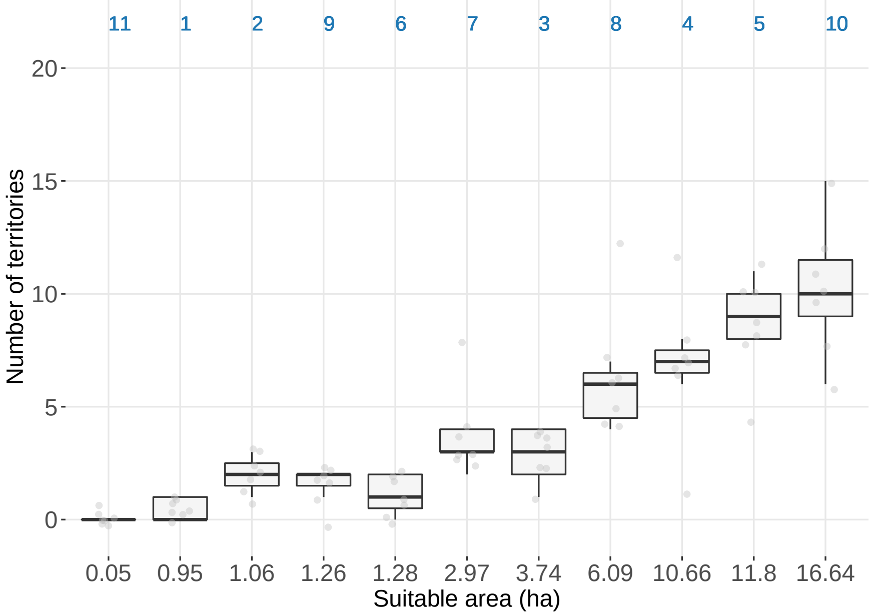


Figure 5.10: Number of Skylark territories between 2008 and 2014 and suitable area of each field calculated during the optimisation. Numbers at the top represent field identification numbers (see Map D.2).

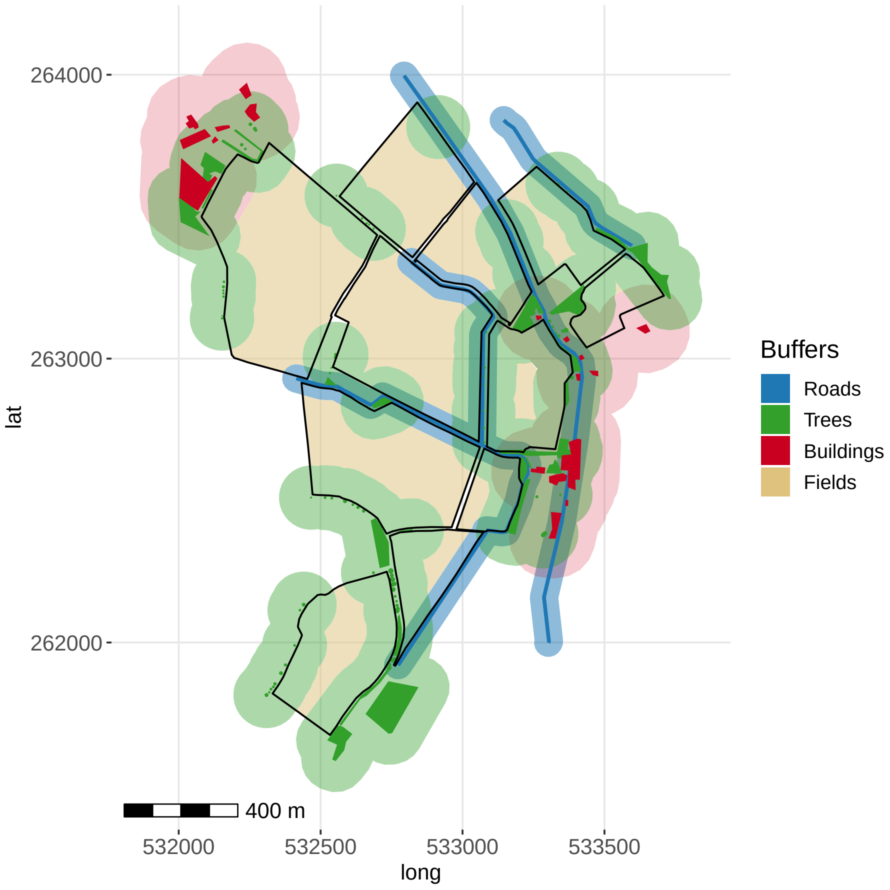


Figure 5.11: Width of buffers used for the suitable area calculation. Buffer width was 140 m, 110 m , and 50 m for buildings, trees, and roads, respectively. Hedges were excluded from the optimisation, as their effect was negligible on the resulting GLMMs.

5.3.4 Sustainable soil management effects on Skylark territories

The GLM defined in §5.2.3.3 of sustainable soil management and Skylark territory number between 2015 and 2019 showed no significant effect of either cover crop or green waste compost amendment on territories, as shown in Tab.5.4. Suitable area was again a significant positive predictor of territory number (z=10.48, P<0.0001), supporting the validity of this measure in describing Skylark territory distribution at Hope Farm. Field shape index, Skylark plots presence, and crop sowing time did not show a significant effect on the number of Skylark territories across the farm and treatment plots.

Similarly, there was no significant difference in Skylark territory density, adjusted by split plot suitable area, between cover crop and green compost treatment plots in the three SSM fields (cover crop t=1.19, P = 0.281; compost t=-1.37, P = 0.177), indicating that SSM did not have an effect on Skylark territory distribution within the fields.

The *Suitable area* measurement obtained from the habitat suitability model was highly heterogeneous among the sustainable soil management treatments. The suitable area of SSM trial treatment plots ranged from 0.56 ha (17.2%) in the traditional soil management plot of Field 3, to 3.76 ha (72.4%) of the cover crop treatment in Field 4 (Fig.5.12). The average suitable area coverage of total plot area, not accounting for variation in plot size among treatments and fields, was of 42.1%(±16.2).

Field 3, in particular, had a smaller suitable area per plot (average 3.72ha±0.93, 25% of plot area) compared to fields 4 and 5 (average 5.7 ha±2.73, 50.3% of plot area), resulting in higher variance in territory density in this field (Fig.5.13).

Table 5.4: Estimated regression parameters, standard errors, and z-values for the Poisson GLMM of Skylark territory numbers between 2015 and 2019 at Hope Farm. Model AIC = 287.6, significant values are in bold.

	Estimate	SE	z-value	P
Intercept	32.24	98.16	0.33	0.743
Cover crop	0.11	0.17	0.65	0.519
Compost	-0.17	0.18	-0.97	0.333
Suitable area	0.87	0.08	10.48	**< 0.0001**
Field shape index	1.32	1.31	1.01	0.314
Skylark plots	-0.17	0.20	-0.83	0.406
Autumn sowing	0.12	0.15	0.81	0.420
Year	-0.02	0.05	-0.35	0.729
Removed				
Cover crop : compost	0.52	0.37	1.42	0.156

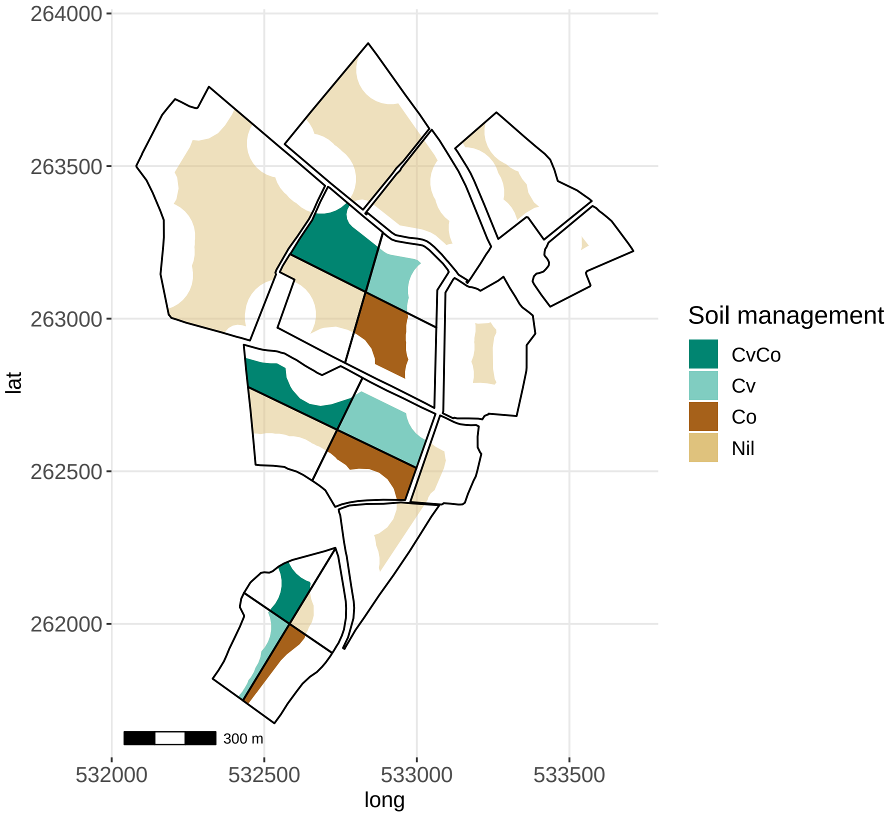


Figure 5.12: Suitable area by soil management treatment at Hope Farm. The SSM trial fields, dividend into four plots with different soil management combinations, are field number 5,4, and 3 from the top. Cv = cover crop, Co = compost, Nil = no SSM.

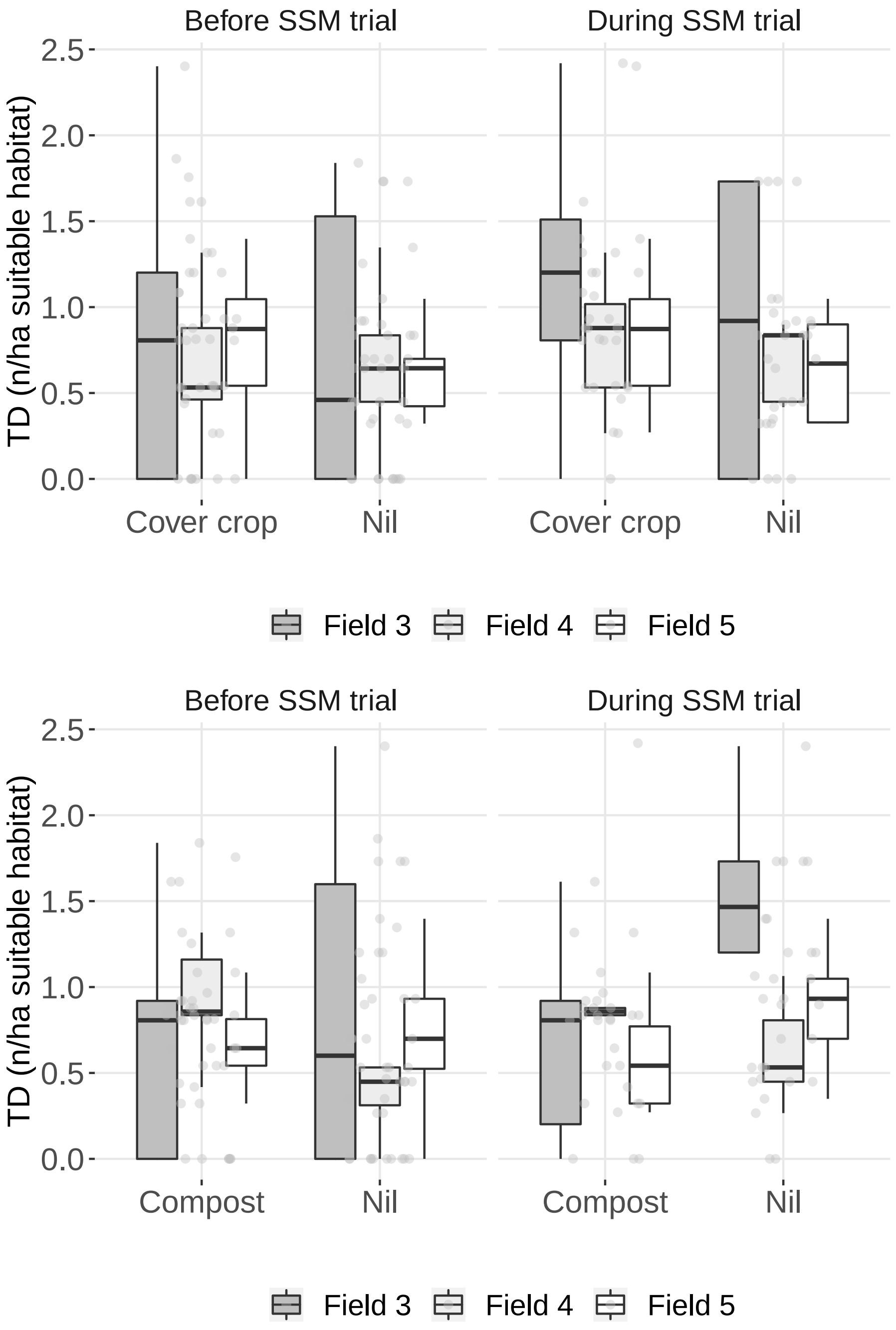


Figure 5.13: Skylark territory density per hectare of suitable area in plots with and without cover crop (top) and green waste compost amendment (bottom). Left-hand graphs show territory densities in the location of the SSM plots, before the trial was put in place (2008-2014), and right-hand graphs show modelled data (2015-2019).

5.4 Discussion

Cover crops and organic soil amendments are adopted increasingly in conventional arable landscapes to maximise crop yield while decreasing anthropogenic inputs (Garratt et al. 2018). Given the potential impact of soil health on farmland birds (Gilroy et al. 2008; Felici et al. 2019), in this study, five years of breeding Skylark observational data in SSM treatment plots on a conventional lowland arable farm were analysed to assess if Skylarks showed a preference in territory settlement among soil management treatments. To account for known drivers of territory placement in the landscape (i.e. vicinity to human settlements and to tall vegetation), territory number was adjusted by considering the amount of suitable area of fields and treatment plots.

There was no evidence of increased patterns of territory settlement in cover crop or green waste compost treatment plots over traditionally managed areas during the five years of this study, suggesting that SSM did not play a role in Skylarks selection of breeding habitat. While there is some evidence of an increased use of cover crop fields by birds in winter (Golawski et al. 2013), and by some migratory species in early spring (Wilcoxen et al. 2018), winter and summer requirements of farmland passerine species can vary substantially (Atkinson et al. 2002; D. E. Anderson 2014), and the findings of this study indicate that there is no legacy effect of cover crops on Skylarks. Although SSM may be successful in improving soil chemical, physical, and biological properties (Garratt et al. 2018), green waste compost and cover crops did not have an effect on Skylark breeding habitat selection.

One of the disadvantages of the split-plot design used in this study is the size of treatment plots, which may have affected the discernibility of the influence of SSM on territory distribution in Field 3. In this field there was a larger discrepancy between field area and suitable area, highlighted by a high variance in territory density. Field 5 and 4, however, showed a high ratio of suitable to total area, and a high territory count, thus, proving suitable breeding habitat, and showing a consistent pattern in territory number. The split-plot design offers many advantages, as it assures comparability of treatments over whole-field study design, due to the shared history of treatment plots (e.g. same input management, same rotation, etc), which is particularly important when considering soil management practices (Andrew S Gregory et al. 2009; Carlesso et al. 2019). Finally, while some have suggested an up to seven years latency in farmland birds responses to changes in agricultural practices (Ambrosini et al. 2002), the length of 5 years of this study should account for most latency effects, as the average lifespan of a Skylark is of 2 years (Donald 2004).

No evidence was found that the presence of Skylark plots – patches of lower vegetation suitable for foraging established in winter cereal crops – influenced the number of territories. This was surprising, and contrary to findings from the same farm (Morris et al. 2004; Morris et al. 2010). Other studies with similar results have attributed the absence of an effect on bird densities and reproductive success to low in-field vegetation density in the fields (Berg and Kvarnbaeck 2011). In this case, however, this was likely due to a temporal effect, as the research conducted by Morris et al. (2010) analysed Skylark territories during the first period of Hope Farm's RSPB ownership, characterised by a strong rise in bird population associated to the implementation of multiple AES trials. Skylark plots may have an initial positive effect on Skylark density and reproductive success, however, as the Skylark population plateaued, as shown in Fig.5.7, the effect of Skylark plots may have strongly diminished. Crop sowing time also did not have an effect on the number of territories. This was probably as a result of analysing annual, rater than monthly, territory number, as crop height preference has been shown to be act as a dynamic driver of Skylark densities during the course of the breeding season (Eggers et al. 2011; Hoffmann et al. 2016). Findings from this study suggest that fields with and without Skylark plots, and autumn sown and spring sown crops were equally suitable for annual Skylark territory settlement.

The approach to automated territory delimitation in this study offers several advantages over the traditional manual method based on individual expert judgment, as the automation always allows to cluster observations consistently based on their distance, on rules of territory mapping, and independently of their density. Although this method may result in a different overall territory number to what manual delimitation would establish – and without taking into account the discrepancy in territory numbers estimated by different analysis (Best 1975; Morozov 1995) – its main strength is its consistency in applying rules uniformly in time and space, making its result effectively comparable across split-plots, fields, study sites, regions, and years, eliminating the comparability issue among different studies, as long as they present similar levels of observational error. Moreover, as hierarchical clustering is independent of point density, this approach allows to apply territory mapping rules consistently in areas densely populated by observations. Here, interpretational error might prevail, as analysts tend to delineate homogeneous territories, whereas individuals of the same species can present territories of highly variable size (Adams 2001) and often clustered together (Muller et al. 1997). For example, it is likely that a Skylark territory comprises of a small, actively defended area of varying size depending on the density of competitors, and a larger surrounding area within which other individuals may be tolerated (Donald 2004). It should be noted that, as highlighted by

Bibby et al. (2000), territory mapping is only appropriate for territorial, socially monogamous songbird species with a limited home range. Therefore, automatic delimitation based on rules of territory mapping should also only be used for such species.

Finally, the optimisation method used to calculate the amount of suitable area of fields and split plots may also be applied in similar fine-scale studies, as long as the main drivers of territory settlement of the species considered are known. Suitable area was the best area predictor of Skylark territory distribution at Hope Farm in the seven years preceding the SSM trial (2008-2014), and it was calculated by subtracting from total field and treatment plot area buffers of varying widths around disturbance elements in the landscape. Without adjusting for these, the number of Skylark territories in the SSM plots (2015-2019) would have been biased towards cover crop treatments (D.5). The result of the optimisation process was very consistent with other studies' findings on drivers of territory settlement, presenting tree avoidance as the main driver fine-scale of Skylark territory number (Wilson et al. 1997; Donald 2004; Gayer et al. 2019), and indicating human settlements as far-reaching agents of settlment depression in the landscape (Gillings 2019; Loretto et al. 2019). The optimisation method proposed here may take into account any number of drivers, but, evidently, the buffer widths identified in each study are specific to its particular setting. Similarly, while in this study hedgerows were excluded from the suitable area calculation because of their distribution on the farm, they are known to generally affect Skylark territory settlement (Donald et al. 2001b) and should not be excluded when analysing territory distribution in different locations.

5.5 Conclusions

Cover crops and organic soil amendments are adopted increasingly in conventional arable landscapes to improve soil health, maximise crop yield, and decrease anthropogenic inputs. Although SSM may be successful in improving soil chemical, physical, and biological properties, this study found no evidence of an effect of green waste compost and cover crops on Skylark territory number, suggesting that these practices do not play a role in the selection of breeding areas. This study proposes a method for automatic territory delimitation following territory mapping rules for territorial, socially monogamous birds, and an approach to calculate in-field breeding suitable area based on known drivers of territory settlement that may be used in further studies of fine scale effects of sustainable management practices on farmland birds.

Chapter 6

Conclusion

Declines in farmland bird populations associated with agricultural management have been experienced throughout Europe and the UK (Donald et al. 2001c; Jerrentrup et al. 2017), and one of the reasons that have been hypothesised for this decline is the reduced prey availability during breeding season (Douglas et al. 2012; Schrauth and Wink 2018; Hallmann et al. 2017). The use of sustainable soil management (SSM) has been introduced in conventional arable agriculture to improve soil health while decreasing the application of anthropogenic inputs (Powlson et al. 2011; Bommarco et al. 2013; Garratt et al. 2018). SSM practices have been introduced to European agri-environmental policies (Turpin et al. 2017) and are often promoted as beneficial to birds, however, the effects of specific practices, such as cover cropping and green waste compost amendment, on farmland biodiversity remain largely unexplored.

In the previous chapters, I have attempted to answer a wide range of questions that relate the use of cover crops and green waste compost amendment in conventional arable fields to breeding farmland birds. The thesis investigated SSM associations with belowground invertebrate abundance and soil biological quality (Chapter 2), aboveground invertebrate abundance and assemblage (Chapter 3), chick food availability (Chapter 4), and breeding bird territory distribution (Chapter 5).

6.1 Is sustainable soil management good for farmland birds?

In this thesis, cover crops were shown to have some beneficial legacy effects on the invertebrate community that, however, were not mirrored by patterns of increased chick food availability or by evidence of preferential territory settlement in SSM plots over traditionally managed ones. Thus, **the results of this thesis suggest that SSM practices did not benefit breeding farmland birds at Hope Farm**.

Aboveground invertebrate food availability was calculated for forty species commonly found in UK's arable landscapes with a newly developed Chick Food Index, which considered all invertebrates ($\leq$ 2mm) sampled in the field as potential prey items. Nonetheless, food availability was shown not to exhibit strong differences among soil management treatment plots. It is possible, that the overall success of Hope Farm at enhancing wildlife may have hindered the detection of local effects of SSM in the fields, and that these would be heightened in highly degraded arable landscapes that present little semi-natural habitats. Hope Farm has employed agri-environment measures for the past two decades, resulting in semi-natural habitat provision throughout its area and a steep increase in breeding and overwintering birds, and of charismatic invertebrate species (Morris et al. 2010; RSPB 2018). Aboveground invertebrate availability thus may have been overall higher than in the surrounding landscape, perhaps rendering all soil management plots suitable for chick food provision. The abundance of soil invertebrates, however, was shown in Chapter 2 to be of the same magnitude as that of commercial farms not managed explicitly for wildlife trade-offs (Field et al. 2016). Moreover, patterns of variation in food availability were found along the sampling transects and in different fields, indicating that local variations in micro-habitat and habitat conditions did in fact significantly affect the invertebrate community, and, in turn, chick food availability.

In Chapter 5, the Skylark *Auleda arvensis* was used as a model species due to its ground nesting and feeding behaviour, and widespread abundance at Hope Farm, to investigate if there were patterns in territory distribution within the experimental plots. While foraging habitat choice is driven by prey abundance (Morris et al. 2005; Kuiper et al. 2013; McHugh et al. 2016), breeding birds are known not to settle territories exclusively based on food availability, but also based on other –often unknown– habitat conditions (Gilroy et al. 2009). For example, in-field Skylarks territory settlement have been been associated with localised vegetation density or soil conditions (Donald 2004). The results of Chapter 5 suggest that SSM did not strongly affect micro-habitat conditions in the field, and that the surrounding landscape features drove the location of territories. It is possible that other species may react differently; the less common Lapwing *Vanellus vanellus*, for example, can nest in spring-grown crops but not in winter-sown crops and, thus, may benefit from the rise in cover crop use. As the historical switch to winter cereals has been shown to have had deleterious effects for many species that were once widespread in agricultural landscapes (Stoate et al. 2001; Robinson and Sutherland 2002; Eggers et al. 2011, 2011), there may be further benefits that could be attributed to this potential increase in spring sown crops in the UK.

The focus of this thesis was birds' breeding season, and the association of SSM to winter season requirements could be different. Birds exhibit different behaviour in summer and in winter, and their requirements can be season-specific (Atkinson et al. 2002; D. E. Anderson 2014). The decline of farmland bird populations in arable landscapes has also been associated to resources availability in winter due to, for example, loss of fallow land and its seed and invertebrate resources, and changing temperatures (Siriwardena et al. 2008; Geiger et al. 2010b). Therefore, relating SSM to in-field summer conditions is just one side of the coin, and further research should investigate the potential of SSM to aid the decline. While extensive literature is available on birds' preference for stubble (Moorcroft et al. 2002; McKenzie et al. 2011) and AES seeding winter covers (Vickery et al. 2002; Perkins et al. 2007), until now, little attention has been given to their use of SSM fields over winter. The existing studies suggest a possible positive association; in particular, Wilcoxen et al. (2018) has shown that cover crops may favour certain migratory species in the Midwest USA, while Golawski et al. (2013) found a partially positive association with overwintering birds.

Finally, it should be noted that food availability was highest in the field boundaries, an unsurprising result given their role in supporting invertebrate and vertebrate populations in arable landscapes (Smart et al. 2002; Benton et al. 2003; Hof and Bright 2010; MacDonald et al. 2012). Although AES that support the creation or retention of semi-natural habitats do not have an undisputed success, as they, for instance, can vary depending on the quality and placement of habitat provided (Kleijn and Sutherland 2003; Henderson et al. 2012; Kuiper et al. 2013), the results of this thesis reiterate the importance of establishing buffer strips and field margins to support breeding farmland birds.

6.2 Sustainable soil management and invertebrates

Sustainable soil management can improve soil conditions and soil habitat quality, favouring invertebrate emergence and overall abundance and diversity. Chapter 2 and Chapter 3 of this thesis focused on the association between SSM and belowground and aboveground invertebrates abundance and assemblage. The terrestrial invertebrate community is integral to the full functioning of arable systems through numerous roles, such as decomposition, nutrient cycling and redistribution (Yang and Gratton 2014), pollination (Klein et al. 2007), and pest control (Hawlena and Schmitz 2010; Honek et al. 2013). Moreover, invertebrates are a vital source of proteins to birds during breeding season, when most species need to consume invertebrates in order to grow and increase their chance of survival (Potts 1986; Douglas et al. 2012; Hallmann et al. 2014). Due to the application of soil management to increase pest predator populations (Settle et al. 1996; Riggi and Bommarco 2019), SSM has the potential to support declining invertebrate populations

(Hallmann et al. 2014).

6.2.1 Green waste compost and invertebrates

The SSM practices investigated in this thesis use different approaches at improving soil health. Green waste compost is an organic soil amendment, and, as such, it aims at directly increasing SOM (Kremen and Miles 2012). As SOM availability is key in the soil trophic web (Moore et al. 2004), compost may be expected to have a positive cascading effects on the soil biological community. The results of this thesis, however, showed that compost did not have an effect on invertebrate abundance or assemblage, even though it significantly increased soil organic carbon content.

The implementation of SSM within arable systems is often driven by the necessity of reducing chemical inputs over a short period of time, as fertiliser application has been linked, for example, to crop pest populations increase (Garratt et al. 2011; Z.-H. Zhao et al. 2015). Organic amendments are likely to provide fertilisation benefits more quickly than cover crops, however, few studies have examined the repercussions of their application in relation to in-field invertebrates. A recent study by Riggi and Bommarco (2019) found a positive effect on the abundance and size of aboveground predators emerging in plots fertilised with manure compared to traditionally managed ones when the crop was present in the field. While it is likely that amendments such as manure or sludge may provide more readily available organic matter than the lignin-rich compost, there is little knowledge of specific organic amendments heterogeneity, toxicity, and effect on invertebrates (Renaud et al. 2017; Azim et al. 2018), and future research should focus on the different long term effects of their application.

6.2.2 Cover crops and invertebrates

The relationship of cover crops with SOM is not as direct as for organic soil amendments. Unless cover crops are directly drilled in the soil (Kautz et al. 2006), they may not have a strong effect on SOM as a whole, and this effect on has been shown to vary depending on crop rotation (Kaspar et al. 2006), tillage (Olson et al. 2010), and species used (Kuo et al. 1997a). Here, this practice did not increase SOM content, but it was shown to have some beneficial effects on the belowground and aboveground invertebrate community. The absence of an effect on SOM may be partly attributed to the fact that cover crops can contribute differently to specific soil organic fractions (Duval et al. 2016).

This thesis found some beneficial legacy effects of cover crops on in-field invertebrate community long after the cover crop termination, suggesting that the addition of cover crops in the cash crop rotation may aid invertebrate populations. In particular, cover crops were shown to have a positive beneficial effect on belowground invertebrates abundance and soil biological quality (Chapter 2), and on the abundance of canopy dwelling invertebrates (Chapter 3). Further research, however, is needed on the long-term effects of cover crops use in the field and, in particular, on their termination. In fact, while the success of cover crops is largely attributed to their introduced plant diversity in otherwise homogeneous plant communities (Crotty et al. 2015; Tiemann et al. 2015), from a commercial prospective, cover crops can become weeds, and thus have to be terminated successfully. Future research should investigate the long-term effects of the most common form of cover crop termination in the field, which is glyphosate application. Glyphosate-based herbicides are widely used, however, very little is known about their effect on the non-target biota present in the fields. Although some studies have shown little impact on invertebrate communities (Sullivan and Sullivan 2003; Santos et al. 2012), more recent ones have challenged these findings (e.g. Domínguez et al. 2016; Niemeyer et al. 2018). It is possible that long-term increased use of glyphosate related to cover crops may hinder their positive associations with belowground and aboveground invertebrates.

6.3 Final remarks

Conventional arable systems produce food at high levels of efficiency, but usually require large amounts of energy and chemical inputs, which have been shown to have negative impacts on the environment (Foley et al. 2005; Newbold et al. 2015). Thus, developing arable systems that are less dependent on these inputs is a key challenge in developing sustainable intensive landscapes for agricultural production (Garnett et al. 2013; Garratt et al. 2018). The growing recognition of soil health importance in maintaining agroecosystems functions is reflected by the development of SSM practices, some of which are also offered as part of agri-environmental policies (Turpin et al. 2017).

Farmers' uptake of selected agri-environmental management options is voluntary, and the potential impact of specific in-field management practice must be evaluated and its impact attributed to targeted goals. SSM practices, whether part of AES or not, have the potential to benefit both soil health and biodiversity, however, the effect of individual practices may be more limited. For example, while there is evidence that targeted agri-environment schemes and management options can benefit farmland birds by halting their declines (Bright et al. 2015; Colhoun et al. 2017; Daskalova et al. 2019), broader interventions designed to benefit species across multiple trophic levels have shown mixed results (D. J. Baker et al. 2012). In particular, in-field management practices have been shown to be less successful at enhancing species diversity than field margins ones (Batáry et al. 2015).

This thesis recorded some positive cover crop legacy effects on belowground and canopy dwelling invertebrates, which, however, did not translate into an increased food availability or territory settlement preference for birds at Hope Farm. Green waste compost amendment was shown to not be associated with variations in invertebrate abundance or breeding territory settlement. The effect of these practices may differ for birds' winter food and habitat requirements, however, the results of this thesis suggest that the expectation that these practices will aid farmland bird populations should be considered carefully.

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Appendix A

Effects of sustainable soil management on belowground invertebrates

Table A.1: Description of the study sites. Fields A–D were three experimental fields located at Hope Farm (HF), of which one was sampled twice, once in 2017 and once in 2018. Fields E–H were located in commercial farms not owned by the RSPB.

Field	HF field	Treatments	Cover crop mix	Compost (t/ha)	Plots	Samples	Year	Crop	Texture
A	Field 3	Cover, Compost	Pachelia, radish, vetch	30	4	36	2017	SB	Clayey, some loamy
B	Field 3	Cover, Compost	Pachelia, radish, vetch	30	4	36	2018	SB	Clayey, some loamy
C	Field 4	Cover, Compost	Pachelia, radish, vetch	30	4	36	2018	WW	Clayey, some loamy
D	Field 5	Cover, Compost	Pachelia, radish, vetch	30	4	36	2018	SB	Clayey, some loamy
E		Cover	Phacelia, radish, black oat		2	18	2018	SB	Loamy
F		Cover	Phacelia, black oat, mustard		2	18	2018	SB	Clayey, some loamy
G		Compost			2	18	2018	SB	Loamy
H		Nil		30	2	18	2018	WW	Clayey and loamy

Cover crop: Black oat (*Avena strigosa*), radish (*Raphanus sativus*), phacelia (*Phacelia tanacetifolia*), vetch (*Vicia sativa*), mustard (*Brassica* sp.); Crop: WW = winter wheat, SB = spring barley.

Table A.2: Spearman correlation coefficients of the soil habitat covariates used in the study.

	Penetrability	Water content	Bulk density	SOC
Penetrability	1.00	-0.31	0.04	0.13
Water content	-0.31	1.00	0.10	0.09
Bulk density	0.04	0.10	1.00	-0.37
SOC	0.13	0.09	-0.37	1.00

Table A.3: Models set explaining total abundance of belowground invertebrates palatable to birds. Models are ranked by their AICc and weight. K is the number of parameters in the model, see below for definitions of model terms.

	Model	K	AICc	Delta AICc	AICc weight
1	Null	3	615.79	0.00	0.51
2	C+S	6	618.62	2.83	0.12
5	C+S+P	7	618.62	2.83	0.12
6	C+S+P+BY	8	619.83	4.04	0.07
7	C+S+W+CR	8	620.12	4.34	0.06
4	C+S+CT	7	620.37	4.58	0.05
3	C*S	7	620.80	5.01	0.04
8	C+S+P+CR+BD	9	622.57	6.79	0.02
9	C+S+CR+BD+LOI	9	623.32	7.53	0.01
10	C+S+P+CT+BY+W+CR+BD+LOI	15	630.11	14.32	0.00

C = cover crop; S = compost amendment; P = sampling point; CT = crop type; BD = bulk density; BY = abundance in the boundary; CR = soil penetrability; LOI = soil organic content; W = gravimetric water content; Null = intercept-only model.

Table A.4: Estimated regression parameters, standard errors, and z-values for the Poisson model of in-field abundance of belowground invertebrates palatable to birds with soil treatments, year, and distance along the transect. The models were not deemed appropriate in describing the response variable, likely due to the low presence of these invertebrates in the samples.

	Estimate	SE	z-value	P
Intercept	1.64	1.06	1.54	0.123
Cover crop	0.18	0.17	1.04	0.298
Compost	-0.07	0.18	-0.39	0.693
Year	-0.81	0.52	-1.57	0.117
Point	0.23	0.15	1.52	0.127

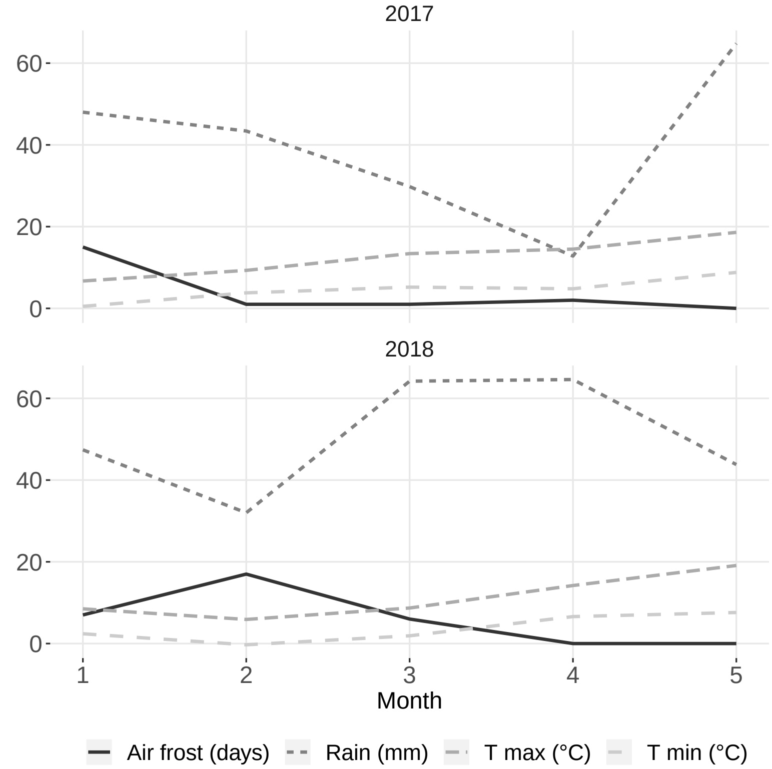

Figure A.1: Monthly average weather conditions in Cambridgeshire (United Kingdom) between January and May 2017 and January and May 2018. One study field was sampled twice during this period and was denoted as A (May 2017) and B (May 2018). Data from Met Office (metoffice.gov.uk).

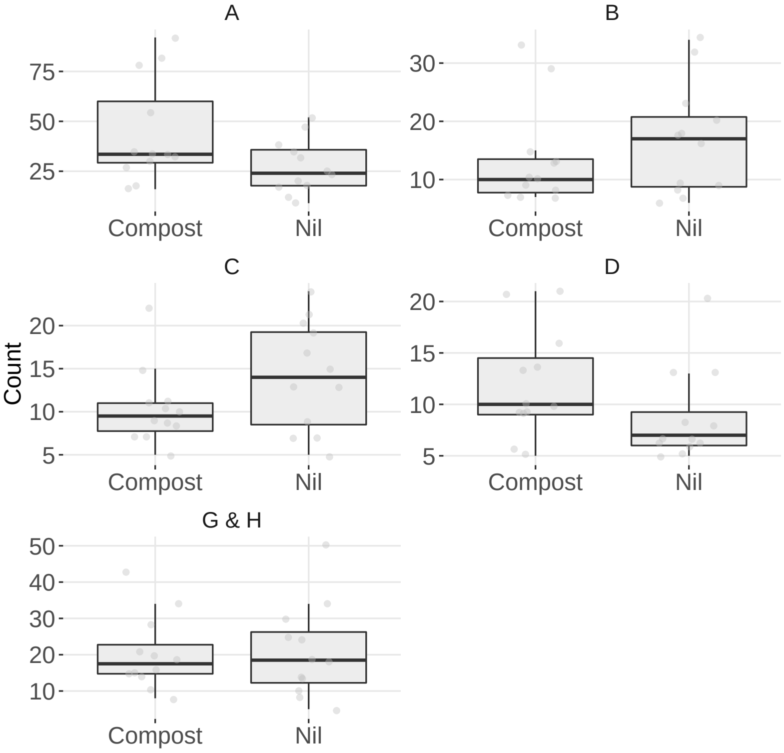

Figure A.2: Total invertebrate abundance with and without compost amendment.

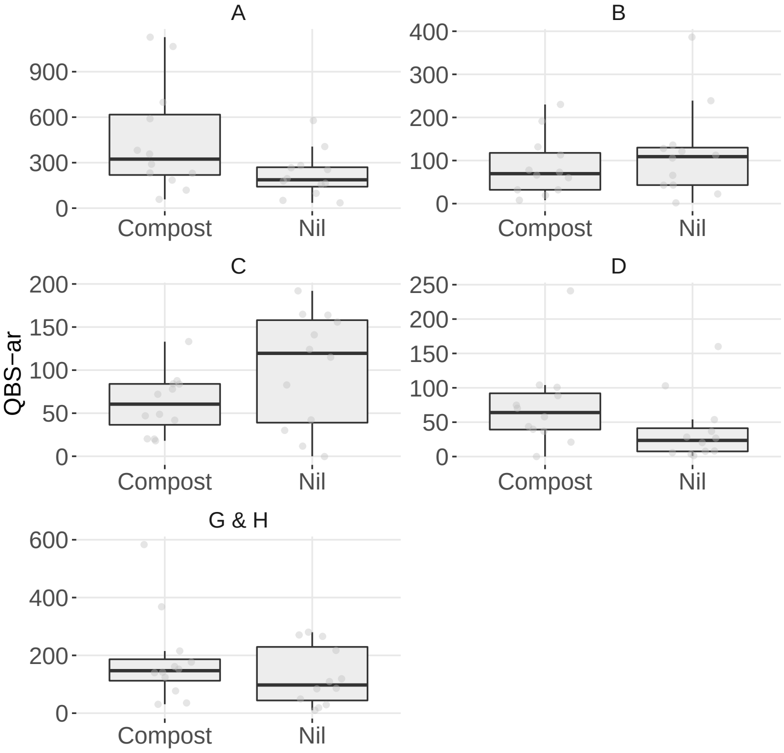

Figure A.3: QBS-ar scores in plots with and without compost amendment.

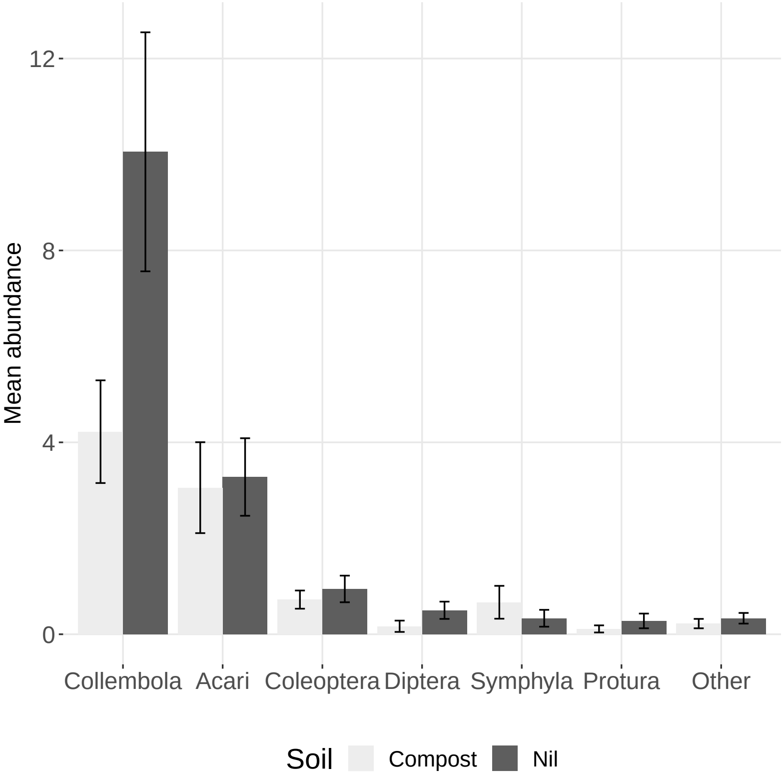

Figure A.4: Abundance of the most common taxa in plots with and without green compost amendment in field C. Collembola (springtail) showed a strong difference in abundance, potentially suggesting a local negative impact of the compost amendment in this field.

Appendix B

Effects of sustainable soil management on aboveground invertebrates

Table B.1: List of ground beetle *Carabidae* species found at Hope Farm, their total and relative abundance, their presence in South-East England, and their main habitat association. All species are frequently found in arable habitats or field margins.

Species	Count	% of Sample	Presence	Habitat
Amara plebeja	2	0.04	common	arable, gardens
Amara ovata	2	0.04	common	arable, gardens
Anchomenus dorsalis	27	0.52	common	grassland
Badister bullatus	2	0.04	common	litter
Bembidion biguttatum	4	0.08	very common	open spaces
Bembidion tetracolum	1	0.02	common	open spaces
leistus fulvibarbis/spinibarbis	4	0.08	common	litter
Bembidion lampros	4	0.08	very common	arable, gardens
Bembidion nitidulum	1	0.02		
Bembidion obtusum	18	0.34	common	arable, gardens
Bembidion punctulatum	3	0.06	sparse records	grassland
Carabus violaceus/problematicus	174	3.33	common	arable, gardens
Demetrias atricapillus	5	0.10	common	arable, gardens
Paradromius linearis	4	0.08	common	arable margins
Harpalus affinis	15	0.29	common	wide spaces
Harpalus rufipes	1,703	32.58	common	arable
Leistus rufomarginatus	239	4.55	common	hedgerows
Nebria brevicollis	1	0.02	very common	arable,gardens
Notiophilus biguttatus	16	0.31	common	open woodland
Notiophilus germinyi	3	0.06	common	litter
Notiophilus rufipes	2	0.04	common	litter
Pterostichus cupreus	287	5.49	common	arable
Pterostichus madidus	2,244	42.91	common	arable, gardens
Pterostichus melanarius	440	8.42	common	arable, gardens
Pterostichus niger	23	0.44	common	woodland, grassland
Pterostichus niger/macer	2	0.04	sparse records	woodland, grassland
Pterostichus nigrita	1	0.02	common	hedgerows

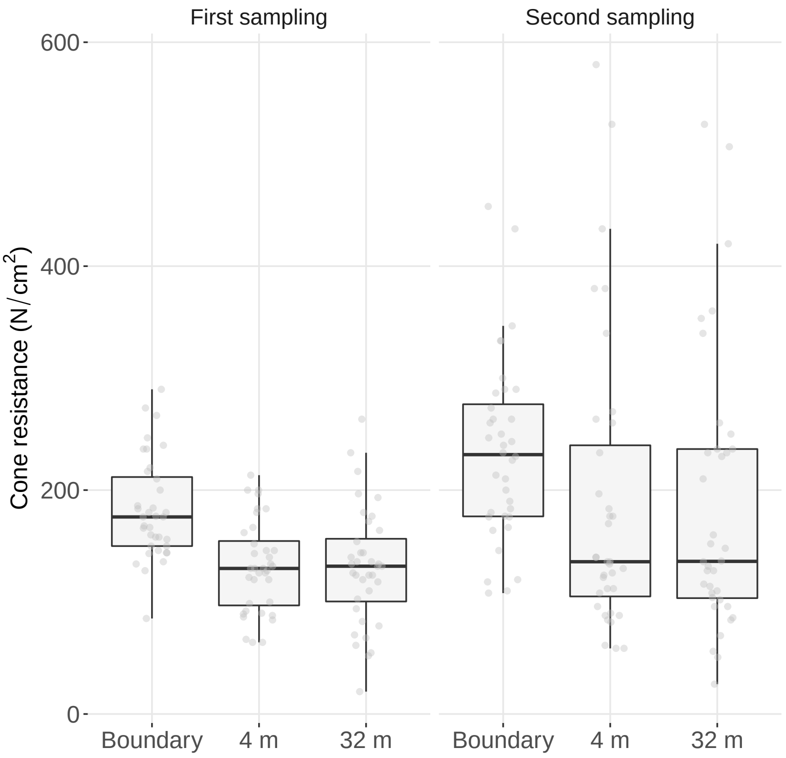

Figure B.1: Soil penetrability along the transects during the first and second round of sampling.

Table B.2: Pairwise comparisons using PERMANOVA of ground beetle community composition between soil treatment combinations.

Sample Type	Comparison	Mean Sq	*F*	*p*
Cover crop (YES)	Compost (YES vs NO)	0.398	2.558	**0.016**
Cover crop (NO)	Compost (YES vs NO)	0.252	1.414	0.061
Compost (YES)	Cover crop (YES vs NO)	0.117	0.688	0.445
Compost (NO)	Cover crop (YES vs NO)	0.498	2.533	**0.009**

Table B.3: Post-hoc pairwise comparisons of ground dwelling invertebrates abundance by soil treatment. Estimated marginal mean contrasts, using Bonferroni adjustment for multiple comparisons.

Sample type	Comparison	Estimate	SE	df	t-ratio	p
Cover (YES)	Compost (NO vs YES)	0.46	0.16	120	2.78	0.026
Cover (NO)	Compost (NO vs YES)	-0.19	0.15	120	-1.27	0.821
Compost (YES)	Cover (NO vs YES)	0.28	0.16	120	1.80	0.296
Compost (NO)	Cover (NO vs YES)	-0.37	0.16	120	-2.39	0.072

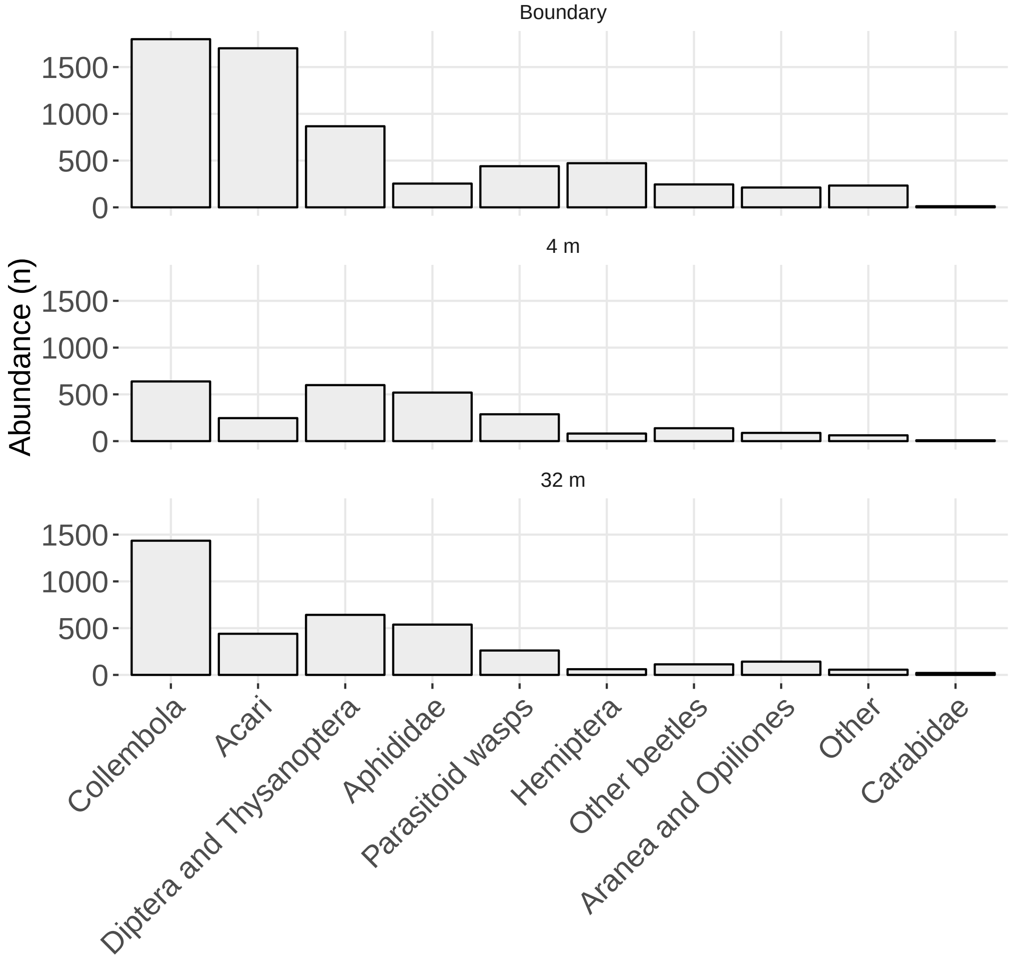

Figure B.2: Abundance of most common canopy dwelling invertebrate taxa collected along the transects extending from 2 m in the field boundary to 32 m into the fields.

Table B.4: Post-hoc pairwise comparisons of ground beetle abundance by soil treatment. Estimated marginal mean contrasts, using Bonferroni adjustment for multiple comparisons.

Sample type	Comparison	Estimate	SE	df	t-ratio	p
Cover (YES)	Compost (NO vs YES)	0.68	-0.20	120	3.46	0.003
Cover (NO)	Compost (NO vs YES)	-0.01	0.19	120	-0.04	1.000
Compost (YES)	Cover (NO vs YES)	0.22	0.20	120	1.09	1.000
Compost (NO)	Cover (NO vs YES)	-0.47	0.18	120	-2.56	0.047

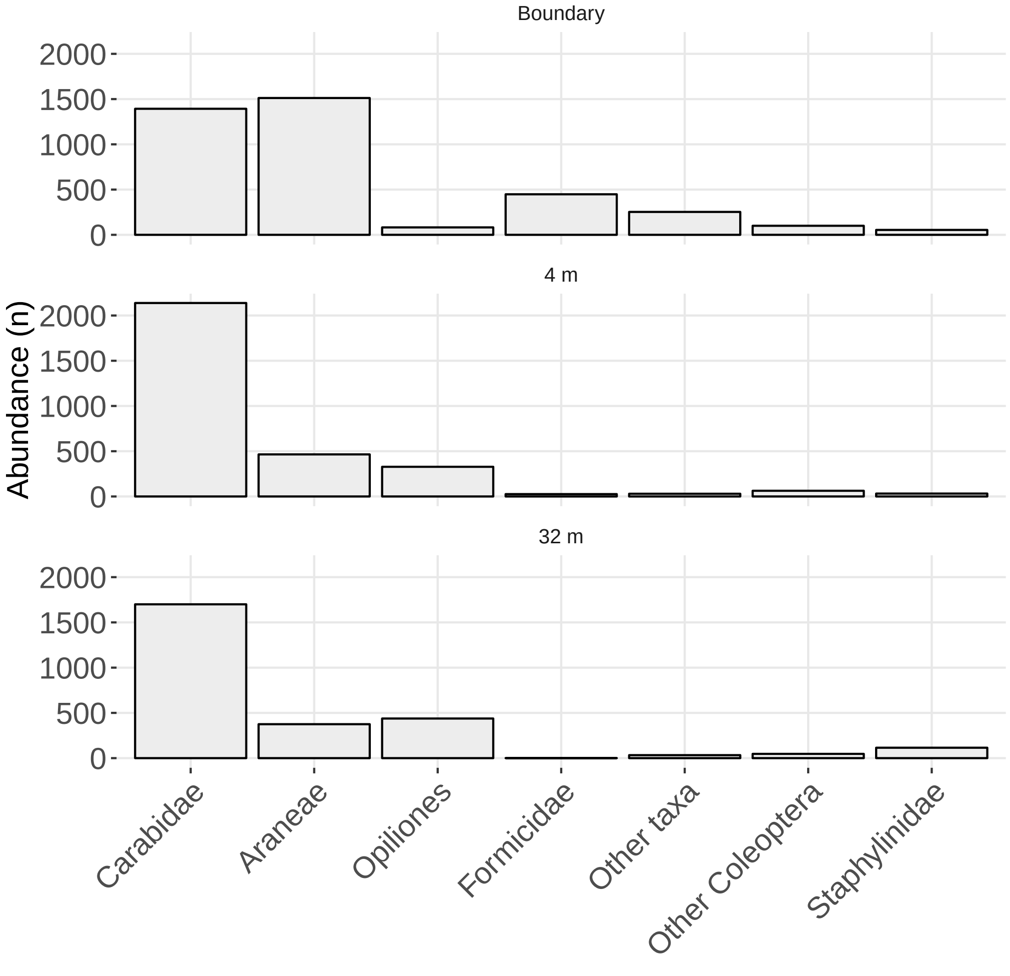

Figure B.3: Abundance of the most common ground dwelling invertebrate taxa collected along the transects extending from 2 m in the field boundary to 32 m into the fields.

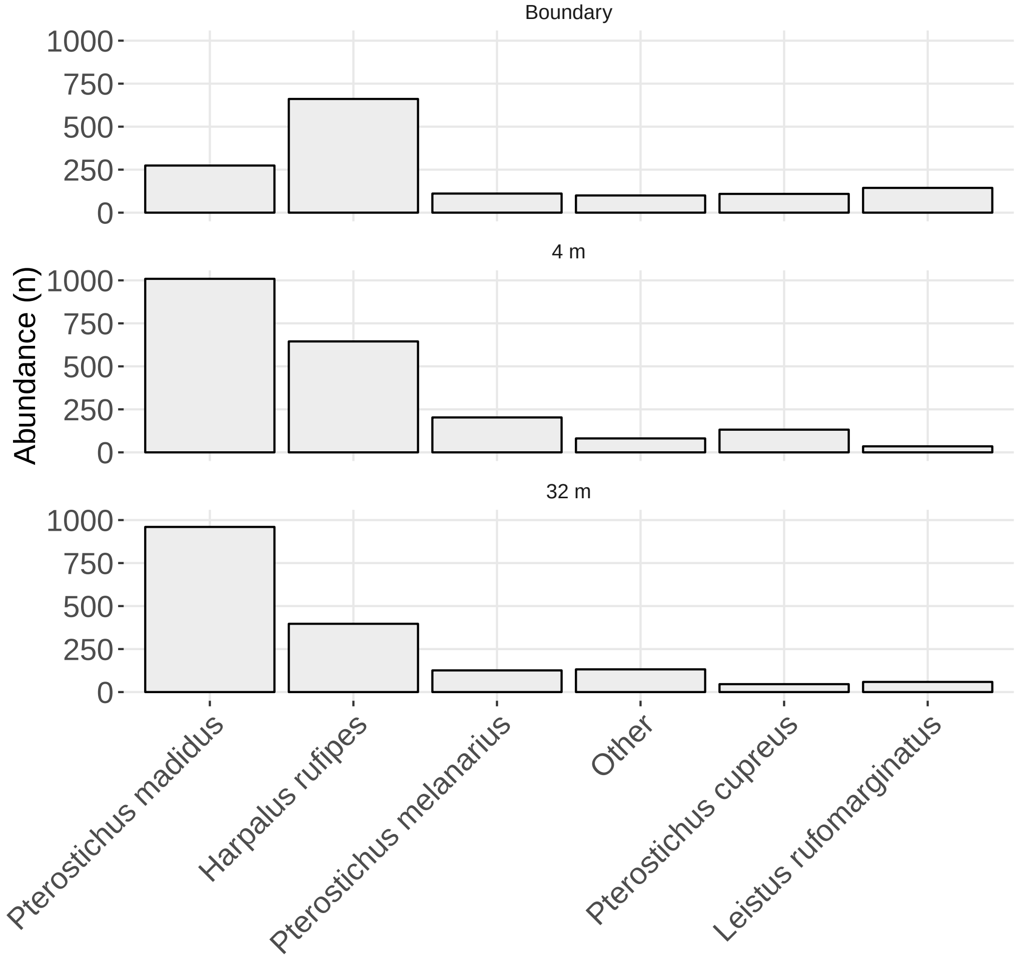

Figure B.4: Abundance of the most common ground beetle species collected along the transects extending from 2 m in the field boundary to 32 m into the fields.

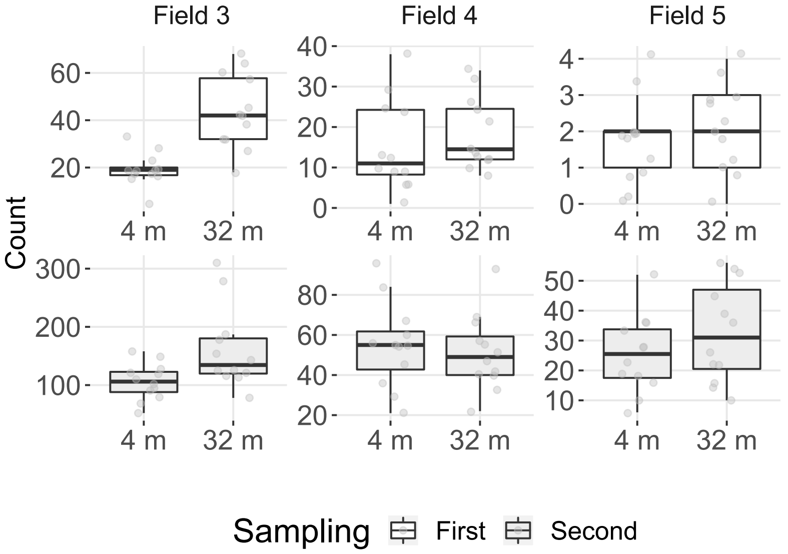

Figure B.5: In-field canopy dwelling invertebrate abundance at 4 m and 32 m from the field edge during the two sampling periods. Abundance was significantly higher towards the field centre than close to the field edge.

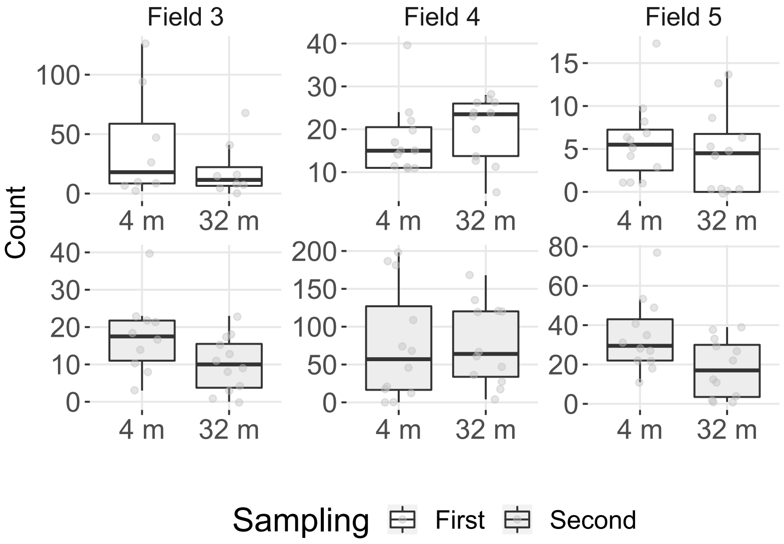

Figure B.6: In-field ground beetles abundance at 4 m and 32 m from the field edge during the two sampling periods. Total abundance was significantly lower towards the field centre than close to the field edge.

Appendix C

Effects of sustainable soil management on chick food availability

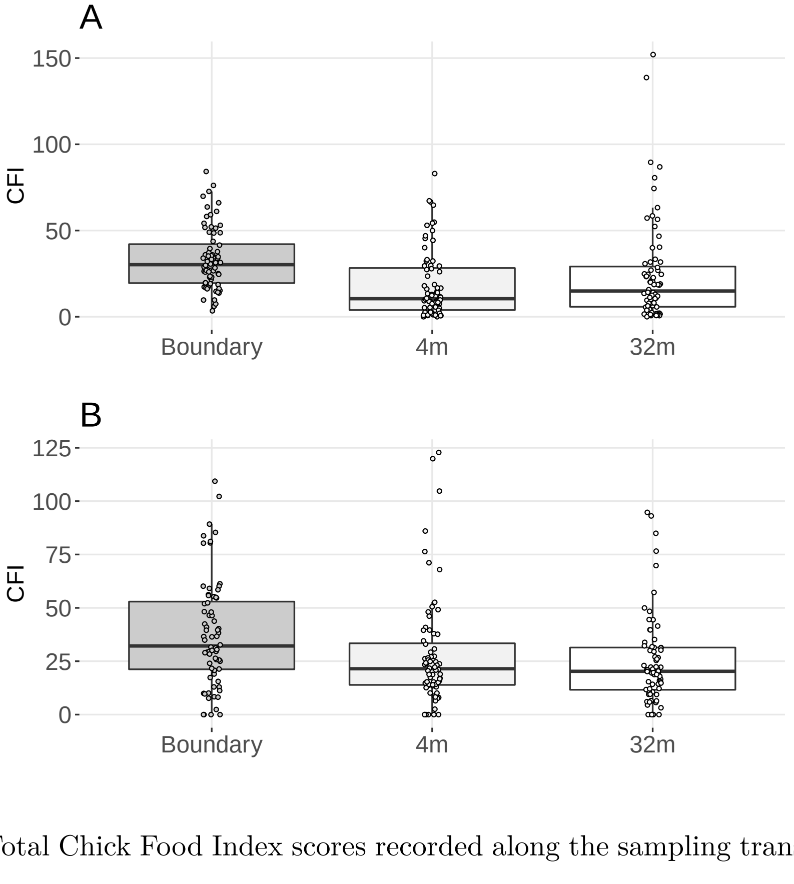

Figure C.1: Total Chick Food Index scores recorded along the sampling transects at Hope Farm during the two period of sampling in samples collected with suction sampling (A) and pitfalls (B).

Table C.1: Scores assigned to the physical characteristics of invertebrates. Palatability of colour, size, and speed of prey were ranked according to Moreby et al. (2006), with lower scores indicate lower prey palatability. Size scores were assigned differently when considering altricial and precocial species, as there are physical limitations to the size of preys available to precocial chicks. Scores were assigned following distribution curves shown in Fig.4.1.

Colour	Score	Speed	Score
Green	0.938	Slow	0.625
Yellow	0.813	Fast	0.375
Buff	0.688		
Brown	0.563		
Metallic green	0.438		
Black	0.313		
Blue	0.188		
Red	0.063		

Size	Length	Score (altricial)	Score (precocial)
XS	≤ 1 mm	0.000	0.000
S	$1 - 3$ mm	0.382	0.759
M	$3 - 6$ mm	0.559	0.981
L	$6 - 9$ mm	0.735	0.759
XL	> 9 mm	0.824	0.000

Figure C.2: Functions of curves shown in Fig.4.1, used to assign scores to traits of colour, size, and speed of invertebrate preys. Function parameters were set so that the curves ranged from 0 to 1 on both axis and had an underlying area of 0.5.

$$\text{colour}(x) = x \tag{C.1}$$

$$\text{size}_{\text{Altricial}}(x) = \begin{cases} b + ax & \text{for } 0.2 \leq x \leq 0.8 \\ 0.8a + b & \text{for } 0.8 \leq x \leq 1 \end{cases} \tag{C.2}$$

$$\text{size}_{\text{Precocial}}(x) = -\frac{7}{18} + \frac{50x}{9} - \frac{50x^2}{9} \tag{C.3}$$

$$\text{speed}(x) = 0.5 + 4 \times (0.5 - x)^3 \tag{C.4}$$

Table C.2: Dietary information for breeding farmland birds. I = ”important”; . = ”present”.

Class/Order	Family	RL	P.	PH	TN	L.	SD	WP	CD	TD	S.	SM	SL	HM	MP	YW	PW	WR	D.	R.	B.
Gastropoda				.	.	I	I	.	I	I	.				.	.	.	.	.	.	I
Isopoda				.	I						.		.		.		.	.		.	
Annelida		.		I	I	I	.	.		.	.				.	.	.		.	.	I
Myriapoda				.	I	.					.				.	.		.	.	.	.
Arachnida	Acari										.				.			.	.		
	Araneae	.	I	.	I	I		.			I	I	.	.	I	.	.	I	I	.	.
	Opiliones		.		.	.					.				.	.			I		.
	Pseudoscorpiones																		.		.
Collembola		.	I								.				.			.	I		.
Ephemeroptera						.					.	I	.	.	.	.	.	.			.
Odonata				.		.					.	.	.	.	.	I	.	.		.	.
Plecoptera					.						.	.	.	.	.	.		.	.	.	.
Orthoptera		I	I	I	I	I					I	.	.	.	.	.	.	.	.	.	.
Dyctioptera																					.
Dermaptera				I	I	.					.		.		.	.	.	.	.	.	.
Hemiptera	Aphididae	I	I	I		I			.		.	I	I	I	I	I	.	I	I		
	Cercopidae		.																.		
	Cicadellidae	.		.							.	I	.		.				.		
	Cicadidae					.					.		.		.	.			.		.
	Coccoidea										.				I						
	Delphacidae			I							.		.	.		.	.		.		
	Heteroptera	I	I	I	.	.					I	.	.		.	.	.	I	.	.	.
	Psyllidae		.									.	.	.	.	I			.		
	Tettigometridae										.										
Psocoptera												.	.	.	.				.		
Thysanoptera											.		.					.		.	
Neuroptera		.		.		.					.	.	.	.	.	I	.	.	.	.	.
Mecoptera													.		.			.	.		.
Lepidoptera				.	.	.					I	.	.	.	I	.	.	I	.	I	I
	larvae	.	I	I	I	.		.	.		.		.		I	.	.	I	I	I	I
Trichoptera			.			.					.	.	.	.	.	.	.	.		.	.
Diptera (all)		I	I	I	I	I			.		I	I	I	I	I	I	I	I	I	I	I
	Syrphidae		I			.					I	.	I	.	.	.	.	.	.		.
	Tipulidae		I	I	I	I					I	.	.	.	I	.	I	.	.	.	.
	larvae		I		.	I					.				I	.	I	.	.	.	I
Hymenoptera	Apoidea										I		.	.						.	.
	Chalcidoidea		.								.	.	.	.							

Table C.2 – continued from previous page

Class/Order	Family	RL	P.	PH	TN	L.	SD	WP	CD	TD	S.	SM	SL	HM	MP	YW	PW	WR	D.	R.	B.
	Formicidae	I	I	I	I	I		.	.		.	.	.	.	.	I	I	I	.	I	.
	Ichneumonidae	.	.	.							.	I	I	I	I	.	.				.
	Symphyta		I	I	.						I	.	.	.	I	.	.	.	.	.	.
	Vespidae										.		.			.					.
	larvae		I	I		.			.		.			.	.				.		.
Coleoptera	Anthicidae											.	.			.					
	Apionidae					.															
	Buprestidae			.	I	.															
	Byrrhidae				.	.					.				.	.	.			.	.
	Cantharidae			.	.						.			.	.				.	.	.
	Carabidae	.	I	I	I	I					I	.	.	.	.	I	I	.	.	.	.
	Cerambycidae				.	.						.	.				.			.	.
	Chrysomelidae	.	I	.		.	.				I	.	.	.	.	I	I	.	.	I	.
	Cicindelidae			.	.						.										
	Coccinellidae										.		.	.	.				.	.	.
	Cryptophagidae																		.		
	Curculionidae	I	I	I	I	I	.				I	.	.	.	.	I	I	I	I	I	.
	Dermestidae																			.	
	Dytiscidae					.						.	.		.	.	.				
	Elateridae	.		I	.	I					I		.	.	.	.	.	.	.	.	.
	Endomychidae																				
	Geotrupidae			.	I	.					.					.	.				
	Gyrinidae					.														.	
	Helodidae																				
	Histeridae				.	.							.					.			
	Hydraenidae					.															
	Hydrophilidae					.						.	.	.	.				.		.
	Lagriidae	.	.	.																	
	Lucanidae											.								.	
	Meloidae			.																	
	Melyridae																				
	Nitidulidae	I	.								.		.	.					I		
	Scarabeidae			.	I	I					.	.	.	.	.	.	.	.	.	.	.
	Scolytidae											.	.	.				.			
	Silphidae		I			.					.						.			.	.
	Staphylinidae	.	I	I	.	I					I	.	.	.	I	.	.	.	I	.	.
	Tenebrionidae				.	I					.										.
	larvae	I	I	I	.	.					.				.		.		.	.	.

Table C.3: Dietary information for breeding farmland birds. I = ”important”; . = ”present”.

Class/Order	Family	ST	M.	WH	SF	MG	JD	RO	C.	SG	HS	TS	CH	GR	GO	LI	BF	Y.	RB	CB	CL
Gastropoda		I	I	.	.	.	.	.	.	.	.	.	.			.	I	.	.	.	I
Isopoda		.				.	.	.	.	.	.	.						.			
Annelida		I	I		.	.	.	I	I	I	.	.	.			.		.		.	.
Myriapoda		.	.	.	.	.	.	.	.	.	.		.					.		.	
Arachnida	Acari	.				.	.	.				.					.		.		
	Araneae	.	.	I	.	.	.	.	.	.	I	I	I	.		I	I	I	I	I	I
	Opiliones	.		.		.	.	.	.	.		.	.					.	.	I	
	Pseudoscorpiones																				
Collembola		.				.	.	.			.	.	.				.	I	I		.
Ephemeroptera					.				.	.	.	.	.			.		.	I		.
Odonata				.	.	.		.	.	.	.	I	.			.		.	I	.	
Plecoptera		.			.							.	.						.		
Orthoptera		.	.	.	.	I	.	.	I	.	I	I	.			.		I	I	I	I
Dyctioptera		.		.		.	.	.	.		.	.	.					.			
Dermaptera		.	.	.	.	.	.	.	I	.	.	.	.					.		I	.
Hemiptera	Aphididae	.		.	I						I	I	I	I	I	I	.	.	.	.	I
	Cercopidae			.						.	.	.						.			
	Cicadellidae			.	.			.				.	.								
	Cicadidae	.		.	.						.	.						.			
	Coccoidea			.		.	.	.			.		.								
	Delphacidae			.								.						.			
	Heteroptera	.	.	.	.	.		.	.	.	.	.	.			.		.	.		I
	Psyllidae			.							.	.	.	I							
	Tettigometridae																				
Psocoptera				.	.		.				.	.									
Thysanoptera		.				.	.				.	.					.			.	
Neuroptera		.		.	.	.	.	.	.	.	.	.	.					.	.		.
Mecoptera		.			.		.					.	.			.					
Lepidoptera		I	.	I	I	I	I	.	.	I	.	.	I	.		.	I	I	.	.	I
	larvae	I	I	I	.	I	I	.	.	I	I	I	I	.	.	.	I	I	I	I	I
Trichoptera		.		.	.	.		.	.	.	.	.	I			.		.	.		
Diptera (all)		I	I	I	I	I	I	.	I	I	I	I	I	.	.	I	I	I	I	I	I
	Syrphidae	.	.	.	.	.	.	.	.	I	.	.	.					.		I	
	Tipulidae	.		.	.	.	.	.	.	I	.	.	I					I	I	I	I
	larvae	I	I	.		.	.	.	.	I	I	I	.		.	.		.	.	I	
Hymenoptera	Apoidea			.	.	.	.	.	.	I	.	.						.			
	Chalcidoidea									.											

Table C.3 – continued from previous page

Class/Order	Family	ST	M.	WH	SF	MG	JD	RO	C.	SG	HS	TS	CH	GR	GO	LI	BF	Y.	RB	CB	CL
	Formicidae	I	.	.	I	.	.	.	I	I	I	.	I	.			.	.	.	.	.
	Ichneumonidae	.		.	I	.	.	.		.	.	.	.					.			
	Symphyta	.		.	.		.			.	.	.	.				.	I	I	I	I
	Vespidae			.	.	.	.	.	.	.											.
	zlarvae	.		.		.	.	.	.	I	.	.	.		.			.	I	I	I
Coleoptera	Anthicidae			.				.													
	Apionidae			.																	
	Buprestidae			.	.	.	.			.		.							.		
	Byrrhidae	.			.	.	.	.	.	.	.	.									
	Cantharidae	.	.	.	.	.	.	.	.		.	.	.					.	.		
	Carabidae	.	.	.	.	I	.	.	I	I	.	I	.			.		I	.	I	I
	Cerambycidae	.		.		.	.	.	.	.	.	.	.					.			
	Chrysomelidae	.		.	.	.	.	.	.	I	.	I	.			I	.	.	.	.	.
	Cicindelidae			.		.	.	.	.												
	Coccinellidae	.		.	.	.		.	.	.	.	I		I							
	Cryptophagidae			.																	
	Curculionidae	.	.	.	.	I	I	I	I	I	I	I	I	I	.	.	I	I	I	I	I
	Dermestidae					.	.	.	.	.											
	Dytiscidae					.	.	.	.	.		.									
	Elateridae	.	.	.	.	.	.	.	.	.	.	.	.			I		.	.	I	.
	Endomychidae			.																	
	Geotrupidae	.	.			.	.	.	.	.											
	Gyrinidae				.	.	.	.	.												
	Helodidae			.																	
	Histeridae		.	.	.	.	.	.	.	.											
	Hydraenidae																				
	Hydrophilidae				.	.	.	.	.	.	.	.									
	Lagriidae																				
	Lucanidae	.				.	.	.	.	.											
	Meloidae			.	.	.	.	.	.												
	Melyridae			.																	
	Nitidulidae			.		.		.				.		.				.			
	Scarabeidae	.	.	.	.	I	I	I	I	.	.	.	.			.		.		I	
	Scolytidae												.						.		
	Silphidae	.			.	.	.	.	.	.	.	.						.			
	Staphylinidae	.	.	.	.	.	.	.	.	.	.	.	.					I	.	.	I
	Tenebrionidae	.		.	.	.	.	.	.	.	.	.			.			.			
	larvae	I	I			.	.	.	.		.	.		I	I	I			.	.	

Appendix D

Effects of sustainable soil management on Skylark territory distribution

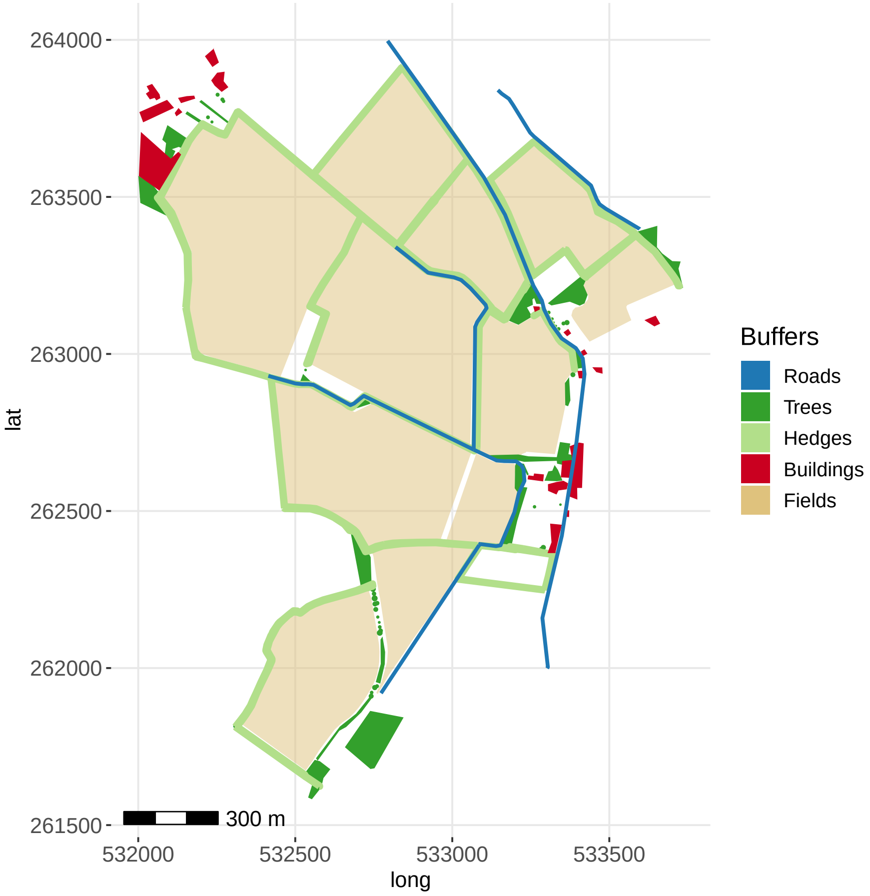

Figure D.1: Due to the uniform distribution of the hedgerows, these were eliminated from the optimisation process for the suitable area calculation.

Table D.1: Characteristics and territory density of the eleven fields at Hope Farm between 2008 and 2019, ordered by decreasing field size. Tot = total area, Suit = suitable area, TD = territory density, SI = shape index, BI = boundary index.

Field	Tot(ha)	Suit(ha)	Suit:Tot(%)	TD(ha)±SD	TD(suit)±SD	SI	BI
10	30.65	16.64	54.30	0.35±0.09	0.64±0.16	1.16	1.05
5	23.56	11.80	50.09	0.36±0.08	0.72±0.16	1.21	2.64
4	21.16	10.66	50.38	0.34±0.12	0.68±0.24	1.19	2.10
3	14.87	3.74	25.15	0.22±0.08	0.89±0.31	1.15	3.15
8	12.97	6.09	46.94	0.42±0.18	0.90±0.38	1.14	0.86
6	11.33	1.28	11.29	0.10±0.08	0.85±0.70	1.17	3.55
7	9.53	2.97	31.15	0.34±0.19	1.09±0.59	1.24	1.60
9	9.33	1.26	13.50	0.14±0.10	1.06±0.70	1.26	2.44
2	6.91	1.06	15.35	0.28±0.10	1.81±0.63	1.47	1.88
11	5.25	0.05	0.95	0.02±0.05	1.67±5.77	1.26	2.79
1	4.52	0.95	21.00	0.13±0.15	0.61±0.70	1.17	1.97

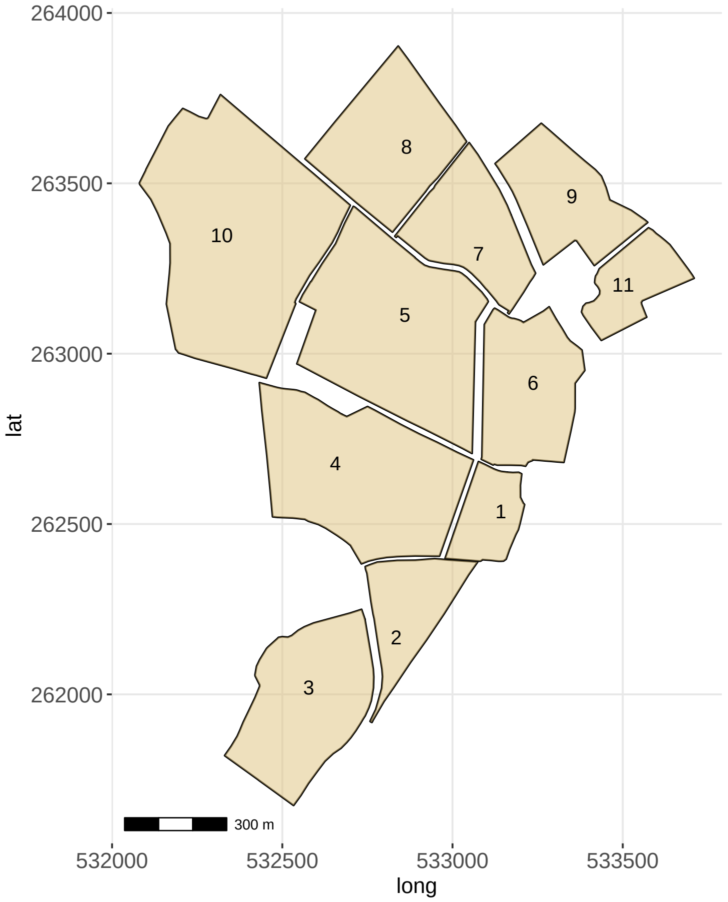

Figure D.2: The eleven study fields at Hope Farm. Fields 3, 4, and 5 are undergoing a Sustainable Soil Management trial since 2015 (§1.4).

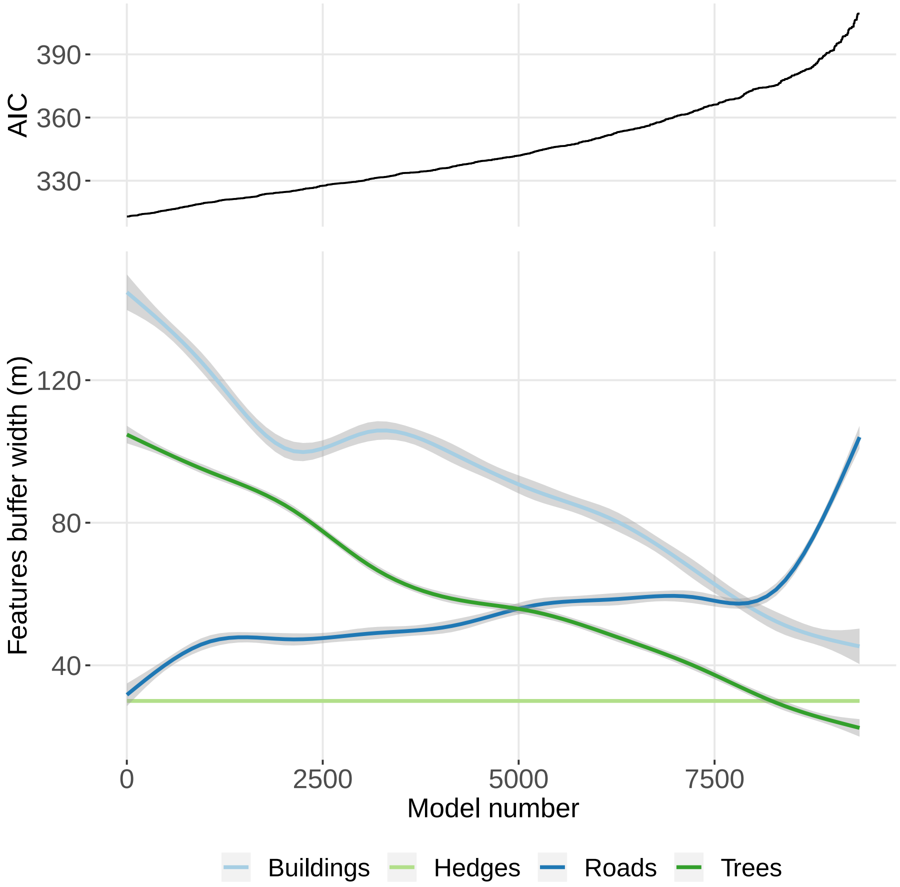

Figure D.3: Smooth curve of buffer width around disturbance features during the first round of optimisation, and AIC score of the respective model. Buffers were drawn around elements of disturbance for Skylark territory settlement and subtracted from the total field area to obtain a measure of suitable area for each field. The resulting GLMMs were ordered on goodness of fit to select the best *Suitable area* predictor for Hope Farm. Features lines indicate estimated values, and shaded areas the 95% CI. Hedges were removed during the second iteration of optimisation due to their lack of effect on AIC scores, shown by the slope = 0 of this predictor.

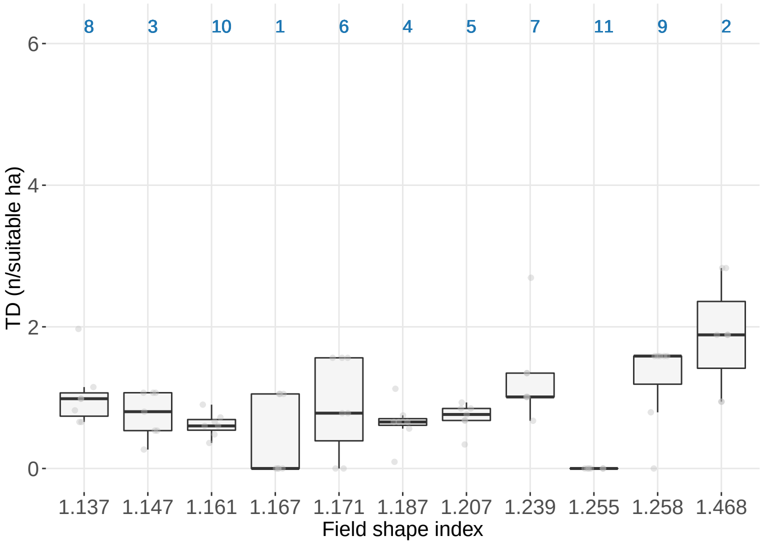

Figure D.4: Skylark territory density by field shape index in the eleven fields at Hope Farm between 2008 and 2014.

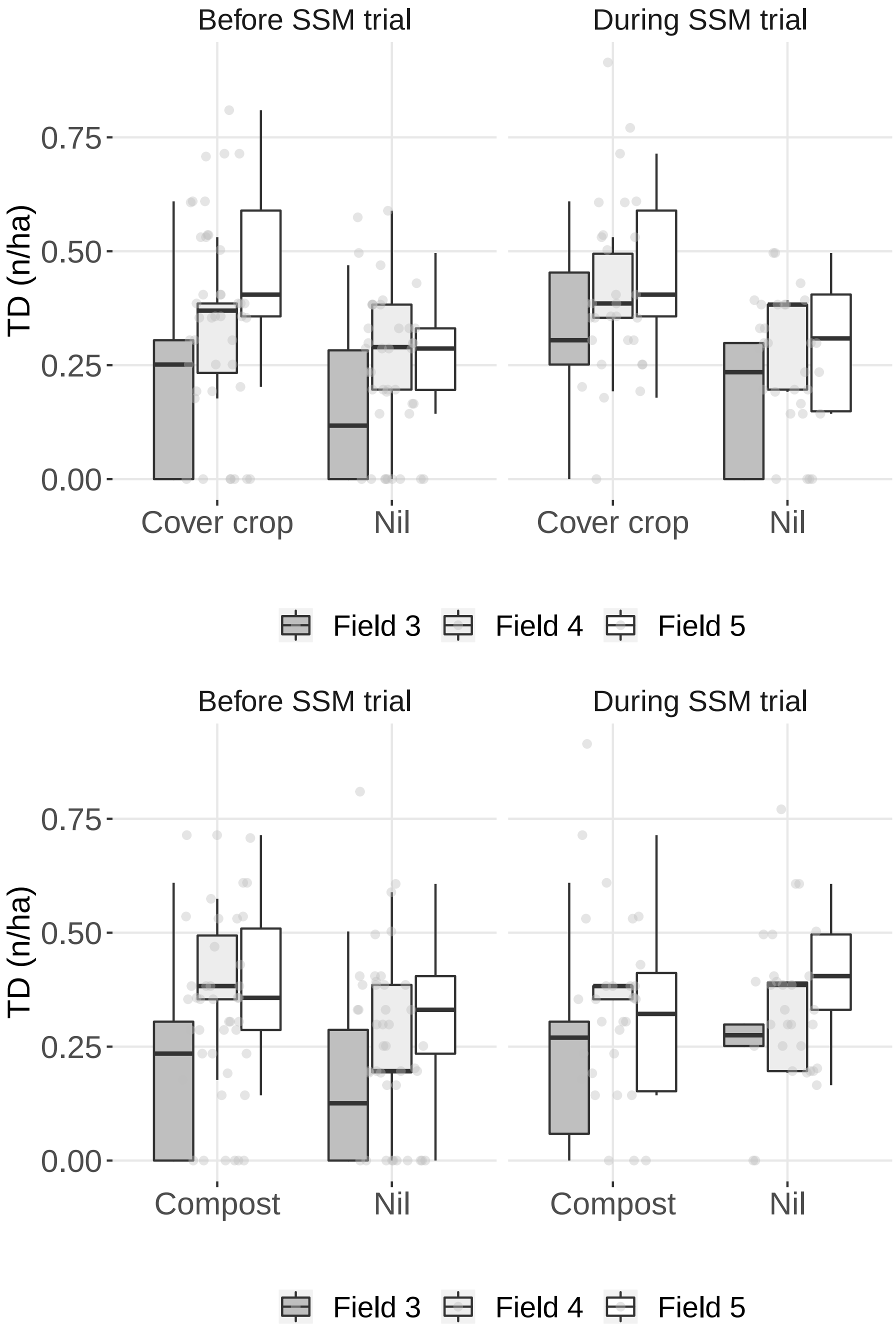

Figure D.5: Skylark territory density per hectare in plots with and without cover crop (top) and green waste compost amendment (bottom), with no *Suitable area* correction. Left-hand graphs show territory densities in the location of the SSM plots, before the trial was put in place (2008-2014), and right-hand graphs show densities during the SSM trial (2015-2019). Without adjusting by the suitable area of each treatment plot, Skylark territory distribution was skewed towards cover crop plots even before the SSM trial was put in place, due to the physical distribution of disturbance elements in the landscape around the fields. Fig.5.13 shows the distribution of data after adjusting for *Suitable area.*