Evaluating the ecosystem function of selected dipteran families and their role as indicators of farm habitat quality in Ireland

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Karzan Sabah D. Ahmed

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Supervisors and Affiliations:

Prof. Mike Gormally, Applied Ecology Unit, Centre for Environmental Science, NUI, Galway

Dr. Christopher Williams, School of Biological and Environmental Sciences, Liverpool John Moores University, UK

Dr. Daire Ó hUallacháin, Environment, Soils and Land use Department, Johnstown Castle, Co Wexford, Ireland.

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DECLARATION

I, **Karzan Sabah D. Ahmed** declare that this thesis was composed by myself, that the work contained herein is my own except where explicitly stated otherwise in the text, and that this work has not been submitted for any other degree or professional qualification except as specified.

This thesis is part of the FARM-ECOS project and parts of this thesis have been published in internationally peer reviewed journals in collaboration with other universities in Ireland. During the study, relevant scientific seminars and conferences were attended at which work from this thesis was presented as part of the FARM-ECOS project. Details of this are given in the thesis appendix. Botanical, habitat mapping and farm management data for this project were provided by GMIT and Teagasc (Co Sligo sites) and by GMIT, Teagasc and TCD (Co. Wexford sites).

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ABSTRACT OF THE THESIS

Biodiversity has undergone significant declines worldwide as a result of human activities. This trend is particularly apparent in agricultural ecosystems where farming intensification is recognised as one of the primary causes of rapid decline in farmland biodiversity particularly in insects with multiple ecosystem services including pollination, biological pest control and nutrient cycling. While the positive role of results-based agri-environment payment schemes in protecting and enhancing farmland biodiversity and increasing habitat quality has been recognised in recent years, there are still gaps in knowledge regarding how farmland habitats should be assessed particularly in relation to non-iconic invertebrates.

Two dipteran taxa selected for this study, Syrphidae and Sciomyzidae, have a range of ecosystem functions. These include pollination, biocontrol and nutrient cycling (Syrphidae) and biocontrol of molluscs (Sciomyzidae), Both families co-exist within many of the same habitats but have markedly different ecological requirements and ecosystem functions, making them good indicators of farm habitat quality. This study comprises three major elements conducted at different scales as follows: 1) an investigation of the ecosystem function/biocontrol potential of *Tetanocera elata* (Diptera: Sciomyzidae) which feeds on pestiferous slugs; 2) an examination of the contribution of linear habitats across a range of farming intensities to both Syrphidae and Sciomyzidae diversity and abundance; and 3) an exploration of how variables (at local and landscape level) could be used to incorporate hoverfly functional groups in rapid farmland biodiversity assessments.

The results of laboratory experiments investigating the biocontrol potential of *T. elata* demonstrated that larval survival outcomes depend on prey slug species with neonate and third instar larvae appearing to prioritise different predatory strategies. Of particular note is that successful pupariation in *T. elata* is reduced for larvae fed on slugs exposed to *Phasmarhabditis hermaphrodita* (Rhabditida: Rhabditidae), a slug parasitic nematode. This potentially negative effect of *P. hermaphrodita* (which can be purchased in Europe for the control of pestiferous slugs) on *T. elata* survival has implications for protecting wild populations of *T. elata* and for how biological control of slugs is undertaken in the future.

The contribution of farm linear habitats across a gradient of farming intensities to Syrphidae and Sciomyzidae abundance and diversity was investigated in Co. Sligo (Ireland) using bi-directional malaise traps. Overall, Syrphidae and Sciomyzidae species richness and community composition reflected a farming intensity gradient with significantly greater species richness for both families on extensively managed farms. In addition, although their abundance and species richness were greatest in dense hedgerows with adjacent watercourses, Syrphidae showed no significant difference overall between linear habitat types. This is in contrast to Sciomyzidae where open hedgerows with adjacent watercourses had significantly more Sciomyzidae species and individuals than dense or open hedgerows without adjacent watercourses. Syrphidae species richness was also significantly correlated with the flowering plant species richness of linear habitats while Sciomyzidae species richness was correlated with a habitat quality score for grasslands adjacent to the linear habitats. The results suggest that while low intensity farming appears to support species richness of both taxa, universal management prescriptions for a farmland linear habitat (such as a hedgerow) may not benefit all dipteran taxa equally. This has implications for future agri-environmental schemes and is discussed in the context of how the conservation value of linear habitats on farmlands is currently assessed.

Given that detailed field surveys of invertebrates at landscape scale are extensive and time-consuming, determining the best habitat features (reflecting Syrphidae ecosystem services) that can be rapidly measured through high resolution satellite imagery rather than field survey, is crucial for national agri-environmental payment schemes. With this in mind, selected habitat features that best reflect hoverfly requirements and their associated ecosystem services were investigated at local and landscape scale using pan traps in two study regions with contrasting farming conditions (Cos. Sligo and Wexford, Ireland). Predatory and non-predatory species dominated the Syrphidae in Cos. Wexford and Sligo respectively. In addition, results indicate that at landscape scale, hedgerows and tree lines (irrespective of county and farming conditions) are the most important driving factors positively affecting hoverfly abundance/richness, with other linear habitat features showing variable results within and between each study region. The implications of these results in the context of selecting the best explanatory variables that can be incorporated in rapid assessments for agri-environmental payment schemes are discussed with particular reference to total lengths of hedgerows/tree lines on farms.

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Chapter 1 General Introduction

1.1. General decline in farmland biodiversity (quantity and quality)

Farmland biodiversity has undergone a significant decline throughout Europe in recent decades mainly due to factors including farming intensification, specialisation and abandonment with significant negative effects on habitats, species, and associated ecosystem services (Anderson & Mammides, 2020; Török et al., 2021). In Europe, an estimated 23% of its farmland biodiversity were lost between 1970 and 2000 (De Heer et al., 2005). This decline in farmland biodiversity over the past few decades has been particularly apparent in insect communities (Benton et al., 2002; Sánchez-Bayo & Wyckhuys, 2019) which are known as basal components in the food webs in addition to which they provide vital services within ecosystems including pollination (Stanley et al., 2013; Rader et al., 2016; Cook et al., 2020; Doyle et al., 2020) and biological pest control (Hynes et al., 2014a,b&c; Wotton et al., 2019). Worldwide, the value of services by pollinating insects was estimated in general to be worth €153 billion in 2005 (Gallai et al., 2009) while a lack of pest control is known to result in a loss of 42% of crop yield (Wiederholt et al., 2017).

Agricultural intensification not only results in a decline in biodiversity in terms of quantity but also in quality where, for example, agricultural intensification and associated loss of natural and semi-natural habitats, result in the domination of predominantly generalist insect species (Tscharntke et al., 2012). These generalist insect species have the ability to utilize a wide range of different resources and to reach scattered habitat patches unlike habitat specialists which can be lost as a result of agricultural intensification (Tscharntke et al., 2012). Given that farmland accounts for a large percentage of land usage in Europe (50%; Lomba et al., 2015) and Ireland (64%; DAFM, 2020), developing effective measures to protect and enhance farmland biodiversity and associated ecosystem services is, therefore, of paramount importance in the context of providing global food security while using sustainable agricultural production approaches (Thrupp, 2000; Rodríguez-Gasol et al., 2020). The importance of this is even more evident when one considers that biodiversity is estimated to contribute US$125 trillion annually worldwide (WWF, 2018).

1.2. Role of agri-environmental schemes in combating biodiversity loss

Concerns over the decline in farmland biodiversity and their associated ecosystem services have led to the development, since 1992, of agri-environment schemes (here after AES) in the European Union (under Regulation EC/2078/92) to mitigate these challenges (Rotchés-Ribalta et al., 2020; Török et al., 2021). These schemes support farmers for biodiversity-based farming practices and to maintain semi-natural habitats on farmland (Rotchés-Ribalta et al., 2020; Török et al., 2021). Holland et al. (2017) defines semi-natural habitats (hereafter SNH) as "any habitat within or outside of the crop containing a community of non-crop plant species". These include linear habitats such as hedgerows and drainage ditches in addition to semi-natural grasslands (see Rotchés-Ribalta et al., 2020).

Studies of European AES indicate that quantity of natural/semi-natural habitats (i.e., new habitat patches) is often prioritised over quality ["measured as plant species richness or the abundance of plants needed for food and reproduction" - see Kleijn and Van Langevelde (2006)]. For example, under traditional action-based schemes such as the Rural Environment Protection Scheme (REPS), Agri-Environment Options Scheme (AEOS), and, more recently, Green Low Carbon Agri-Environment (GLAS) (DAFM, 2021), farmers get payments on the basis of actions taken regardless of the presence of higher quality biodiversity on their farms since a "one size fits all" is the approach taken by the funding agency. Feehan et al. (2005), for example, has demonstrated that farms with the greatest diversity in Ireland (for the taxa surveyed) were not in AES and Birrer et al. (2007) suggests limited positive impacts on biodiversity for the Swiss AE measures due to the lack of ecological quality. Therefore, improving local habitat quality at a regional scale has been suggested as being a more promising conservation strategy than habitat quantity (Jeltsch et al., 2011). However, the relationship between habitat quality and quantity is often complex (Vickery et al., 2004) and a recent study by Rotchés-Ribalta et al. (2020) indicate that retention of semi-natural habitats on farms (i.e., habitat quantity) in Ireland does not necessarily guarantee greater quality of habitats. Indeed, small habitat patches with high habitat quality can provide better resources for invertebrates than a larger habitat patch of poor habitat quality (Haenke et al., 2009; Meyer et al., 2009).

As a result of low impact of traditional action-based AESs, the Results-based Agri-environmental Payments Schemes (RBAPS) have been recommended in Ireland (and Spain) as an alternative approach (Ruas et al., 2021) in that famers are paid for the quality of the biodiversity on their farmland (i.e., results based) rather than quantity (i.e., action based). RBAPS are based on biodiversity scores (e.g., on a scale of 1-10), developed using objective criteria, with the highest payment awarded to the best quality habitat. Thus, RBAPS incentivise delivery of higher quality biodiversity and associated ecosystem services from farmland (https://rbaps.eu/, accessed 05/07/2021).

1.2.1. Limitations with current AE approaches

While most farm biodiversity within RBAPS is assessed based on habitat type, condition, and/or plant indicator species (Power et al., 2016; Carey et al., 2017a; Ruas et al., 2021), biodiversity is still declining (IPBES, 2019; Newbold et al., 2020), and further knowledge is needed to guarantee the successful implementation of agri-environment measures (Rotchés-Ribalta et al., 2020) in the context of the next CAP reform (January 2023). While plant indicator species are normally used to assess habitat quality in Result-Based Agri-Environment Payments schemes (Ruas et al., 2021), recent studies have shown that assessing farm habitats based on plant indicator species alone is not sufficient since many terrestrial invertebrates have different requirements and do not reflect plant indicator species alone (Maher et al., 2014; Williams et al., 2014; Carey et al., 2017a & b).

1.2.2. Opportunities for greater use of invertebrates

While there has been an increasing trend in using invertebrates in agri-environmental schemes, this has largely been confined to better studied, iconic groups such as bees (Hymenoptera) and butterflies (Lepidoptera) (Carey et al., 2017a). However, less has been done to address the conservation of diverse and abundant groups such as the true flies (Diptera) (Keiper et al., 2002). This is due to the abundance and diversity of such groups making them time consuming to identify and requiring considerable taxonomic expertise. This has resulted in groups such as Diptera largely being excluded from routine habitat assessments primarily due to resource limitations (Carey et al., 2017a). Nevertheless, their inclusion in agri-environmental studies is important since habitat management recommendations which support non-dipteran pollinators such as bees, for example, may not support some dipteran species that deliver multiple ecosystem services (Power et al., 2016).

1.3. Diptera and their ecosystem services

Diptera, commonly known as true flies, are among the most diverse groups of organisms, with about 160,000 named species in about 150 families worldwide (Evenhuis et al., 2008). Almost 50% of these families contain flower-visiting flies pollinating at least 555 flowering plant species (Larson et al., 2001). Therefore, they are one of the most important groups of pollinating organisms, second only to the Hymenoptera with both having a major contribution to plant diversity and agricultural production (Ssymank et al., 2008). In addition to pollination, they provide important pest control and nutrient cycling services (Knutson & Vala, 2011; Schirmel et al., 2018; Wotton et al., 2019; Speight, 2020). Marsh flies (Sciomyzidae) and ~~H~~overflies (Syrphidae), for example, are two well-known dipteran families that co-exist within many of the same farm habitats but have different ecosystem services (Carey et al., 2017a & b).

1.3.1. Marsh flies

Marsh flies or shade flies, often known as snail-killing flies (Family Sciomyzidae), are a group of 540 small to medium-sized acalypterate fly species. To date, 60 species of Sciomyzid flies have been recorded in Ireland (Chandler et al., 2008; Maher et al., 2018). Adults occur in both aquatic and terrestrial habitats such as ponds, marshes, swamp, wood and meadows with larvae the occurring in ponds, emergent vegetation and damp soil (Knutson & Vala, 2011).

There are two tribes within the subfamily Sciomyzinae: The Tetanocerini and the Sciomyzini (Vala et al., 2012). The Tetanocerini have typically evolved as aquatic gastropod predators whereas many of the Sciomyzinae have evolved terrestrial parasitoid strategies. However, exceptions do occur, particularly with regard to various tetanocerine larvae, which are terrestrial parasitoids for part of their development (Knutson et al., 1965). The shade fly species *Tetanocera elata* (Diptera: Sciomyzidae), for example, is a host-specific parasitoid of slugs until the penultimate instar, at which stage the larva emerges from the host and becomes a generalist gastropod predator prior to pupariation (Knutson et al., 1965).

1.3.1.1. Marsh flies and ecosystem services

While some adult sciomyzidae have been reported to be minor pollinators (Stoffolano Jr et al., 2015), most larval stages of sciomyzid flies are gastropod predators, or display parasitoid behaviour, with a select few aquatic or semi-aquatic species predating on snail eggs or bivalves (Barker, 2004; Knutson & Vala, 2011). In the field, sciomyzid larvae parasitize and/or predate on one or more species in the Mollusc families Ariophantidae, Arionidae, Clausiliidae, Cochlicopidae, Endodontidae, Eulotidae, Haplotrematidae, Helicidae, Limacidae, Littorinidae, Lymnaeidae, Physidae, Planorbidae, Polygyridae, Succineidae, Vertiginidae, Vitrinidae and Zonitidae (Knutson et al., 1965).

A number of snail-feeding sciomyzid species have been studied with a view to determining their potential to control snails which act as vectors of parasites such as *Fasciola hepatica* (Mc Donnell et al., 2007; Dillon et al., 2014) . On the other hand, *Tetanocera elata*, the focus of Chapter 2, is an obligate mesoparasitoid (i.e., partly embedded in its host) of the pestiferous slug *Deroceras reticulatum* (Stylommatophora: Agriolimacidae) for its first and much of its second larval instar, becoming a more generalist slug predator in its third instar. Due to its parasitoid and predatory larval stages, *T. elata* a as a biocontrol agent of pestiferous slugs using augmentation and/or conservation biocontrol strategies shows some potential (Hynes et al., 2014a,b & c). Recent studies have contributed significantly to our knowledge of the biology, and ecology of *T. elata* (Hynes et al., 2014a, b & c; Bistline-East et al., 2018; Bistline-East et al., 2020a & b). A hypothetical larval population based on the maximum: number of eggs (373) laid by one female (Knutson et al., 1965); observed egg hatch rate (86%) (Beaver, 1973); observed rate of survival to third larval instar (69%) (Hynes et al., 2014b); observed rate of survival from third larval instar to pupariation (89%) (Hynes et al., 2014b); and number of slugs killed per larva (11) (Knutson et al., 1965), has the potential to kill 2,167 slugs under laboratory conditions. However, to fully assess the viability of *T. elata* as a biocontrol agent of pestiferous slugs, further work is required including host range of larval stages, climatic adaptability, synchrony with host (prey) life cycle, prey searching ability, prey handling time, survival at low host (prey) densities and the natural enemy assessment in addition to detailed habitat requirements for larval, pupal and adult stages. In addition, potential interactions with the only commercially available nematode biocontrol agent of pestiferous slugs *Phasmarhabditis hermaphrodita* (Schneider, 1859) need to be fully explored.

1.3.1.2. Ecological requirements of the group

While a full understanding of the habitat requirements for every sciomyzid species is still required, much has been discovered about the family in recent years (Williams et al., 2009a; Williams et al., 2009b; Williams et al., 2010; Maher et al., 2014; Maher et al., 2015; Carey et al., 2017a; Carey et al., 2017b; Bistline-East et al., 2020). In general, sciomyzids are relatively sedentary (Williams et al., 2010) and adults of a number of species are likely to lay eggs in the vicinity of their aquatic or semi-aquatic mollusc hosts which usually have a clustered or patchy distribution in wet sites (Knutson and Vala, 2011). Adult Sciomyzidae are also known to show quantitative and qualitative changes in relation to environmental variability, scale of observation, management type and hydrology (Speight, 2001; Williams et al., 2009a & b; Williams et al., 2010; Carey et al., 2017a). Sciomyzidae abundance and species richness in grassland habitats are shown to be greater in sites with intermediate wet-dry gradients where the more terrestrial species are found in drier grassland areas with hygrophilous species in wetter areas. Most species prefer long vegetation with the exception of the smaller species (Williams et al., 2009b). Other studies on Sciomyzidae habitats in wet grasslands have shown positive correlations between Sciomyzidae abundance and/or species richness with increased hydroperiod (Williams et al., 2009a; Maher et al., 2014) while others have shown highest Sciomyzidae species richness in wet grasslands with lowest stocking density (Ryder et al., 2005). This is likely due to changes in plant biomass over time resulting in taller and more dense vegetation thereby providing a range of microhabitats for dipteran families including Sciomyzidae (Ryder et al., 2005).

A recent study of the requirements of the terrestrial slug killing fly *T. elata* in a dry meadow site in western Ireland was undertaken by Bistline-East et al. (2020). They found that the areas where the largest aggregation of *T. elata* were found were areas with a greater percentage cover of dead vegetation, and vegetation height in agreement with previous studies about wetland Sciomyzidae in general (Williams et al., 2009a & b; Maher et al., 2014; Carey et al., 2017a). In addition, Bistline-East et al. (2020) found that the abundance of *T. elata* is positively correlated to hedgerow proximity indicating that hedgerows may play a significant role in sustaining *T. elata* populations. However, given that these studies were based on a single site study, further studies are required to better understand habitat requirements of *T. elata* to develop successful conservation programmes for these ecologically and potentially economically important taxa. The effects of different farming intensities on Sciomyzidae abundance and richness and the role of different semi-natural linear habitats in conserving them has not yet been investigated. Detecting and enhancing habitats for sciomyzid species which predate pestiferous slugs and snail vectors of liver fluke disease in grazing animals will support an important ecosystem function. This is particularly important in the context of the EU recent restriction of methiocarb slug pellets and the growing problem of anthelmintic resistance in cattle and sheep (European Commission, 2018).

1.3.2. Hoverflies

Hoverflies, also known as syrphid flies (family Syrphidae), are one of the most abundant group of true flies with more than 6000 species described globally (Sommaggio, 1999; Marín-Armijos et al., 2017). More than 930 species are known across Europe and 185 species have been recorded to date in Ireland (Speight 2020).

1.3.2.1. Hoverflies and ecosystem services

Hoverflies have important ecosystem services by acting as plant pollinators which is primarily linked to food production (Jauker & Wolters, 2008; Stanley et al., 2013; Dunn et al., 2020). This value of ecosystem services by insects as pollinators in general has been estimated to be worth €153 billion globally in 2005 (Gallai et al., 2009). In addition to the pollination activities of adult syrphids, the immature stages of many species have also important ecosystem service roles. Some are important beneficial predators of crop pests, particularly aphids (Amiri-Jami et al., 2017; Hatt et al., 2017; Dunn et al., 2020), where seasonal migrations of nearly 4 billion individuals between Britain/Ireland and continental Europe result in the predation of approximately 6 trillion aphids (Wotton et al., 2019). In addition to pollination and pest control, some other syrphid species play important roles in the decomposition of dung and nutrient cycling in their larval stages, thereby increasing soil fertility (Power et al., 2016; Moquet et al., 2018; Speight, 2020).

1.3.2.2. Ecological requirements of the group

While hoverflies are known to be influenced by the availability of floral resources (i.e., high quality habitats) at local scale (Kleijn & Van Langevelde, 2006; Haenke et al., 2009; Meyer et al., 2009; Power et al., 2016; Garratt et al., 2017; Moquet et al., 2018), a recent study across 35 agricultural landscapes in Switzerland and Germany by Schirmel et al. (2018) indicate that the abundance and richness of floral resources are poor predictors of hoverfly abundance, richness and community composition at local scale. Therefore, other quality measures need to be developed for rapid farmland biodiversity assessments at local scale such as habitat quality measures described by Rotchés-Ribalta et al. (2020) which includes not only floral resources, but also includes other local structural (e.g., vegetation) and ecological elements (e.g., positive/negative plant indicators) important to habitat quality (see Rotchés-Ribalta et al., 2020). In addition, hoverflies are also known to be influenced by landscape context where complex landscapes (i.e., with high proportions of SNHs) are known to promote hoverflies across different types of semi-natural habitats in particular woody semi-natural habitats (Schirmel et al., 2018). However, while the value of semi-natural habitats for hoverfly abundance/richness and their communities are known, less is understood about the value of specific features at landscape scale (Schirmel et al., 2018) including non-woody linear features such as watercourses, stone walls and grassy margins. This is particularly important for rapid biodiversity assessments at landscape scale where not all the linear features at landscape scale will be equally beneficial for hoverflies' abundance, richness and their communities.

1.4. Diptera as indicators of habitat quality

Traditionally, many studies have used vascular plants to assess the quality of farmland habitats (Pharo et al., 1999; Falkengren-Grerup & Schöttelndreier, 2004; Odland, 2009). However, recent publications indicate that some agricultural habitats classified as being " poor " in ecological terms based on plant species composition, often contain rarer terrestrial invertebrates which the "better" sites do not (Maher et al., 2014; Williams et al., 2014). The wide diversity of feeding and breeding habits among phytophagous, saprophagous, predaceous, and parasitic groups of Diptera influences their habitat selection (Taillefer & Wheeler, 2010) which in turn can make them good indicators of habitat quality (Frouz, 1999; Williams et al., 2009b; Gelbič & Olejníček, 2011; Maher et al., 2014; Carey et al., 2017a). Previous studies which have identified this potential have included the use of various dipteran species (larval and/or adult stages) to assess the ecosystem health of soil (Frouz, 1999), water (Saether, 1979; Metcalfe, 1989; Arimoro et al., 2018), forests (Brown, 1997; Gittings et al., 2006), sand dunes (Mattoni et al., 2000) and grasslands (Carey et al., 2017a).

More recently, studies suggest that Syrphidae which reflect biodiversity at local and/or landscape scale (Moquet et al., 2018; Schirmel et al., 2018) and Sciomyzidae (marsh flies) which reflect biodiversity at a local scale (Carey et al., 2017a; Carey et al., 2017b) have the potential to be useful in assessing farm habitat quality. These two taxa co-exist within many of the same habitats (Carey et al., 2017a; Carey et al., 2017b) but have markedly different ecological requirements making them suitable taxa for assessing the ecosystem health of farm habitats (Williams et al., 2009b; Maher et al., 2014; Power et al., 2016; Carey et al., 2017a; Carey et al., 2017b).

1.4.1. Marsh flies as bioindicators

In addition to their ecosystem function, Speight (1986) suggests that Sciomyzidae have potential as bioindicators of habitat quality in wetlands since they meet the important criteria in terms of ease of collection, identification and their ubiquity across a range of habitats. Abundance/species richness of Sciomyzidae are known to be greater in areas with intermediate/high hydroperiod and/or high vegetation height (Williams et al., 2009a & b; Maher et al., 2014). In addition, Sciomyzidae species richness are known to be highest in areas with lowest stocking density (Ryder et al., 2005). Recent studies in Ireland have contributed significantly to our understanding of Sciomyzidae as bioindicators across a range of habitat types in particular ephemeral wetlands and/or grasslands (Ryder et al., 2005; Williams et al., 2009a; Williams et al., 2009b; Maher et al., 2014; Carey et al., 2017a; Carey et al., 2017b).

In addition to the role of Sciomyzidae as indicators of habitat quality and farming practices, adults (along with Syrphidae) have been used as indicators of nine other dipteran families in grassland habitats in Western Ireland including the families Dolichopodidae, Empididae, Hybotidae, Limoniidae, Pipunculidae, Scathophagidae, Stratiomyidae, Tabanidae, and Tipulidae (Carey et al., 2017a). This makes Sciomyzidae a particularly important taxon when selecting insects as indicators of wider biodiversity in agri-environmental studies since these less studied dipteran taxa are often overlooked (so called "Dark Taxa"; https:// bolgermany.de/home/gbol3/dark-taxa-2/; accessed 17/06/2021) while other iconic insects such as pollinators have been prioritised (Carvell et al., 2007; Pywell et al., 2011; Garratt et al., 2017). This is an important consideration since a single prescription for habitat type that suits bees may not necessarily be of equal benefit to other taxa such as Diptera with multiple ecosystem services (Power et al., 2016).

1.4.2. Hoverflies as bioindicators

Hoverflies also meet important environmental and functional bioindicator criteria as suggested by Speight (1986); (Sommaggio, 1999; Sommaggio & for species identification (particularly in Europe), and differences in environmental requirements of larvae are features that make hoverflies potentially good bioindicators (Sommaggio, 1999). Indeed, Sommaggio and Burgio (2014) studied the role of syrphid flies as biological indicators of two different farm habitats (conventional vs organic vineyards) in Northern Italy over three years. They found that the use of hoverfly functional groups is a better strategy in understanding differences in the biodiversity value between the two vineyards. Carey et al. (2017a) investigated the role adult Syrphidae (and Sciomyzidae) play as indicators of a wider range of dipteran diversity in wet grassland habitats in Ireland and found that Syrphidae has the potentialto be a useful indicator of nine other dipteran families in wet grassland habitats (Dolichopodidae, Empididae, Hybotidae, Limoniidae, Pipunculidae, Scathophagidae, Stratiomyidae, Tabanidae, and Tipulidae). The study highlighted the need to examine dipteran groups at scales atypical of current agri-environmental assessments, namely field and farm level since certain dipteran groups are more influenced by small spatial scales. In addition, Moquet et al. (2018) who studied insect visitors of Belgian heathlands, found that hoverflies require complementary resources at both landscape and local scales. Larval habitat availability was relevant at the landscape scale, whereas adult resource availability (floral density) was relevant at local scale, highlighting the importance of including both adult and larval habitat requirements when studying hoverfly communities in agroecosystems.

1.5. Linear farm habitats as biodiversity reservoir

Linear farm habitats, one of the more common semi-natural habitats on farmland in Ireland, have been the focus of study by conservationists for some time given that they are not within the cropping system and are, therefore, not subject to normal, day-to-day intensive farming practices (Tattersall et al., 2002; Herzon & Helenius, 2008; Garratt et al., 2017; Volpato et al., 2020). Linear habitats can play an important role in the supply of resources for invertebrates that deliver pollination and pest control services in addition to preventing soil erosion and run-off in both urban and rural areas (Spellerberg & Gaywood, 1993; Holland et al., 2017; Hevia et al., 2021). However, changes in agricultural land use over the last century has resulted in dramatic changes in landscape structure and composition through the removal hedgerows and small woodlots to enlarge agriculturally productive areas (Leonard & Cobham, 1977; Agger & Brandt, 1988; Anderson & Mammides, 2020). In north-west Europe, such declines in non-cropped habitats (i.e., field boundaries) have coincided with increases in farm size and changes in farming systems from extensive to intensive farms (McMahon et al., 2012), resulting in a significant loss of biodiversity over the last two decades (Benton et al., 2003; Donohue et al., 2006). While the protection and appropriate maintenance of remaining farmland linear features is critical for the future maintenance of biodiversity in agricultural landscapes, much is still unknown regarding optimum maintenance of linear features for all associated taxa, particularly the less well-known invertebrate groups i.e., the "Dark Taxa" (https://bolgermany.de/home/gbol3/dark-taxa-2/; accessed 17/06/2021).

1.5.1. Hedgerows

Foulkes & Murray (2006) define hedges or hedgerows as “Linear strips (4m wide or less) of woody plants with a shrubby growth form that covers more than 25% of the length of a field or property boundary that has been deliberately established or managed. They often have associated banks, walls, ditches (drains), or trees”. Hedgerows which are considered as one of the most valuable semi-natural linear habitats across many farms, particularly in lowland areas, contribute significantly to the biodiversity of farmland (Le Cœur et al., 2002; Boughey et al., 2011; Garratt et al., 2017). They are considered important features of the Irish landscape covering 4% of the total land area (Forest Service, 2018). Hedgerows also provide shelter and food for a range of functional taxa including pollinators, predators, parasitoids, herbivores and scavengers (Pollard & Holland, 2006; Garratt et al., 2017; Dover, 2019; Volpato et al., 2020) and supply important niches for invertebrates such as Araneae (spiders), Coleoptera (beetles), Diptera (true flies), Hemiptera (true bugs), Lepidoptera (butterflies and moths) and Hymenoptera (bees, wasps, ants) (Pollard et al., 1974; Pollard & Holland, 2006).

Hedge quality greatly affects the benefits derived from ecosystem services such as pollinators and/or biological control agents (Garratt et al., 2017; M'Gonigle et al., 2017; Sutter et al., 2017). Undisturbed hedgerows rich in woody species are the preferred habitat for bumble bees and Linipid spiders whereas other spiders (e.g., Lycosidae) prefer hedgerows with trees. In addition, hoverflies are most frequently found in hedgerow habitats where a diversity of flowering plants is available as a foraging resource (Garratt et al., 2017; Graham et al., 2018). Thus, hedgerows in farmlands with high species diversity can protect and enhance biodiversity and associated ecosystem services (Garratt et al., 2017). Wolton et al. (2013) studied the response of 157 UK priority and farmland quality indicator species (vertebrates and invertebrates such as various butterflies) to different hedgerow structural elements such as shrub layer, tree layer, base and margin and found that almost 65% of the species preferred more than one hedgerow element and 35% preferred three or more elements. This was particularly apparent in widespread species in which 81% of them preferred a mixed hedgerow element consisting of a combination of Shrub/Tree (19 species), Shrub/Base/Margin (9 species) and Shrub/Tree/Margin (8 species) (Wolton et al., 2013). Heard et al. (2012) suggest that increasing hedgerow quantity helps in the protection of their associated wildlife through the provision of summer and winter food resources but this would require a diversity of hedgerows at the farm scale level. Hedgerows with tall or mature trees are likely to increase the species richness of invertebrates by providing shelter and food sources for species that feed as larvae on trees and shrubs (Wolton et al., 2014). In another study, Peng et al. (1992) found that hedgerows with trees have influential effects on many dipteran families with the majority of them found at the middle of an 11. 5m high flowering lime tree emerging from a hedgerow. A few more other dipteran families such as Anisopodidae and Mycetophilidae were found at 8.8m and 1.2m high, respectively. Similar results were found for other invertebrates (macro moths) in which abundance and species richness were greater in hedgerows with trees than without trees (Merckx et al., 2009).

Haenke et al. (2014) investigated the abundance of hoverflies in three semi-natural linear habitats (forest edges, hedgerows connected to native forest, and isolated hedgerows) along with adjacent oilseed rape or winter wheat fields and with varying degree of connectedness to annual crops and forest. It was found that the abundance of aphidophagous hoverflies was greater in hedgerows connected to forest than in forest edges with an intermediate of isolated hedgerows (with no forest nearby). The abundance was even greater in crop fields adjacent to hedgerows than adjacent to forest edges. The study highlighted the importance of increasing length of hedgerow in the agricultural matrix to support biodiversity and associated ecosystem services in agricultural lands.

Whilst providing important linear habitats, hedgerows are also important in linking different habitats. This is particularly important in farmed landscapes, where invertebrates with particular ecosystem services (e.g., pollinators and biological control agents) depend on the use of hedgerows as corridors for their survival (Wiens et al., 1993; Dover, 2019). Bumble bees and solitary wasps, for example, use hedgerow networks to shift between their food source and nesting site (Cranmer et al., 2012; Heard et al., 2012), while some others such as butterflies (Dover & Sparks, 2000), moths, (Coulthard et al., 2016) and ground beetles (Charrier et al., 1997) move along hedgerows to access their preferred food source, mating and breeding sites.

Dense continuous hedgerows that are diverse in woody species and floral resources are generally considered to be good quality hedgerows and are recognised as important habitats for invertebrates with strong mobility such as bumblebees

(Garratt et al., 2017; Volpato et al., 2020) and hoverflies (Garratt et al., 2017). However, hedgerow density/permeability can influence the spatial distribution of some dipteran and hymenopteran species depending on their flight ability (Burel et al., 2004; Dover, 2019). Pollard and Holland (2006) state that high numbers of arthropods at the hedgerow nodes (hedge ends) reflect their preferred microclimate since these ends were usually close to woodland; high numbers at a gap indicates the gap as a barrier to movement due to blocking connectivity with the neighbouring woodland; and high numbers at the centre may be indicative of species' movements along the hedgerow as a result of diffusion processes. Joyce et al. (1999) stated that in fragmented arable farmlands, hedgerows have important roles in providing food source and shelter for various invertebrate species but their role as corridors depends on hedgerow connectivity and the quality of habitats surrounding them. The authors observed the movement of carabid beetle *Nebria brevicollis* (Fabricius) between hedgerow nodes and the linear section of the hedgerow, and across gaps (7-9 m) in the hedgerow using a mark and recapture method. The insect activity was more at hedgerow nodes than mid-section of the hedgerow likely due to their connectivity to other patches of suitable habitat such as woodlands, thereby indicating the role of hedgerows as corridors for species movement.

Speight (2001) studied the role of different parts of a farm in maintaining the Syrphidae and Sciomyzidae fauna in Co. Cork (Ireland) and concluded that efforts that can make intensive and productive farms more ‘’eco-friendly’’ may be difficult to obtain without maintaining farm infrastructural habitats mainly field margins and associated hedgerows and ditches. In addition, maintenance of disused lands around farms can further increase the number of Syrphidae species surviving in the farm by more than one third. The author also demonstrated that hedgerows dominated by a shrub layer along with field margins and ditches maintain more hoverflies than any of these features alone. The latter two increased the Syrphid abundance by twofold and a further 30% respectively (Speight, 2008). However, the study was based on a single farm as a case study and showed no clear evidence on how these effects reflect other fauna and farmland in general.

Wolton et al. (2014) studied the dipteran diversity associated with a single hedgerow over two years in a low intensity organic farm in central Devon, southwest of England comprising a 35 ha of permanent pasture for cattle and sheep. The farm was situated on poorly draining acidic clay soils of Culm with the surrounding landscape comprising small fields dominated by soft rush *Juncus effusus* bordered with species-rich hedgerows in all directions. The hedgerow area was 0.056 ha (85 m long and 6.55 wide). A total of 830 species of Diptera was recorded from this single hedgerow by the authors comprising 16.7% of all dipteran species on the British list with the Sciomyzidae and Syrphidae comprising 22% and 33.1% of the British list, respectively. These two families and several other dipteran families were recorded including Dolichopodidae, Empididae, Hybotidae, Limoniidae, Scathophagidae, Stratiomyidae, and Tipulidae; indicating the importance hedgerows in protecting abundance and diversity of many dipteran species. However, these results by Wolton et al. (2014) were also based on a single farm and hedgerow, and only with one farming intensity. There is no clear evidence how these results might vary between different types of hedgerows (e.g., gappy vs intact hedgerows) as well as across different farming intensities.

While there is extensive evidence from the above-mentioned literature that hedgerows in general, are important in sustaining invertebrate diversity including pollination and pest control species (e.g., Wolton et al., 2014; Garratt et al., 2017; Graham et al., 2018; Volpato et al., 2020), there are still gaps in the knowledge in relation to the value of different types of hedgerows for different invertebrate groups including those less studied taxa with non-pollinating ecosystem services. It is likely that one type of hedgerow such as a dense, continuous thick hedgerow that is suitable for one taxon, such as bees (Power et al., 2016; Garratt et al., 2017; Volpato et al., 2020), can be detrimental to other less studied taxa (Graham et al., 2018), such as parasitiod wasps with low mobility (Dover, 2019) or sedentary species such as snail and slug killing flies (Williams et al., 2010). This is because semi-natural habitats (including hedgerows) can vary in their resources which can strongly affect their utilization by invertebrates (Cole et al., 2017). Further research is, therefore, required to address this research question in more detail.

1.5.2. Drainage Ditches

Drainage ditches as defined by Davies et al. (2008) are “man-made channels created primarily for agricultural purposes and which usually: (i) have a linear planform; (ii) follow linear field boundaries, often turning at right angles; and (iii) show little relationship with natural landscape contours". They are one of the diverse freshwater environments worldwide providing a suitable habitat for a range of aquatic organisms such as invertebrates in both adult and immature stages (Herzon & Helenius, 2008; Biggs et al., 2017). They represent an important feature of farmland through their role in enhancing drainage during wet seasons and protecting plant roots and reducing stress to crops especially in areas with flat surface and high water table (Leslie & Lamp, 2017). Drainage ditches can be either permanent wet channels of up to several meters deep and wide or may be small channels of less than a meter width which are only wet during periods of high rainfall (Macdonald & Feber, 2015).

In the lowlands of north-western Europe, drainage ditches are one of the common features of agricultural landscapes harbouring a large number of macroinvertebrate communities, similar to those found in other small natural waterbodies such as lakes and ponds (Verdonschot et al., 2011, 2012). Drainage ditches are a suitable habitat for a range of aquatic fly larvae such as some species of Sciomyzidae where their larvae feed on non-operculate snails (e.g., *Galba truncatula*) at the water surface or foraging on vegetation and exposed wet shorelines. For some other Sciomyzidae species, their larvae feed on fingernail clams beneath the water surface (Knutson & Vala, 2011). Drainage ditches are also a suitable habitat for larvae of common hoverflies which feed on decaying material (Herzon & Helenius, 2008). However, the biodiversity value of drainage ditches is less well investigated than other linear farm habitats (Herzon & Helenius, 2008; Verdonschot et al., 2011; Hill et al., 2016; Biggs et al., 2017). The biodiversity of drainage ditches can be influenced by the quality of habitats existing in the ditch such as water type (colour), soil, and vegetation structure. Many species may depend on drainage ditches as their preferred habitat for a specific period during their life cycle. These species are, therefore, primarily influenced by changes in the hydro-cycle which includes hydroperiod, depth, flow, and chemical composition of water. All these changes are dependent upon drainage ditch type (i.e., disturbed vs undisturbed) and size, surrounding farm habitats and farming systems (Herzon & Helenius, 2008).

Kleijn and Van Langevelde (2006) estimated the effects of ditch habitat quality (flower abundance) and quantity (total length of drainage ditches) on the species richness and abundance of bees and hoverflies in intensively managed grassland and arable lands in Netherlands. Flower abundance and the area of semi-natural habitats within a radius of 500 to 1000m from the sampling sites were significantly correlated to species richness of hoverflies and bees, and these factors had interacting effects on both species' groups. The species richness and abundance of hoverflies were not significantly correlated with drainage ditch length or the interaction between drainage ditch length and number of inflorescences at any spatial scale. Bee species richness was similar to those described for hoverfly species richness, but their abundance was positively related to the number of inflorescences of bee plants (Fussell & Corbet, 1992).

Verdonschot et al. (2011) compared the aquatic macroinvertebrate value of nine agricultural drainage ditches and small lakes in The Netherlands. They found that the macroinvertebrate community of drainage ditches were comparable with those found in small lakes including rare species despite the fact that the studied drainage ditches consisted of smaller water bodies, had less vegetation structure, more variation in nutrient content and were managed periodically. At the regional scale, the macroinvertebrate diversity was even greater for drainage ditches than small lakes, demonstrating the importance of drainage ditches as a significant habitat for aquatic invertebrates. Aquilina et al. (2006) studied the effect of two types of drainage ditches (dammed and non-dammed) in arable and pastoral areas of Leicestershire, UK on invertebrate and bird abundance. The study found that wetting-up drainage ditches by raising water levels had a positive effect on the abundance of invertebrates and birds with greater numbers found in dammed ditches. Macdonald and Feber (2015) also surveyed the macroinvertebrate communities of 49 drainage ditches. A total of 190 aquatic invertebrate taxa were recorded with an average of 19.6 ± 5.76 SD taxa per drainage ditch. The frequency for taxa found in all the drainage ditches were Chironomidae (98% of the ditches), oligochaete aquatic worms (82%), water beetle *Helophorus brevipalpis* (87%) and a rare species such as the minute moss beetle *Hydraena palustris* (4 %). Water depth and shading along the surveyed drainage ditch channels and banks were found to affect the biodiversity of drainage ditch fauna and flora. Plant and invertebrate communities were positively correlated with increasing water depth from 6cm to 56cm. For the latter, the species richness declined with increasing water depth (>56cm) with only few other families found in deeper drainage ditches including Caenidae, Ephemeridae and Brachycentridae. The presence of shade along ditch channels negatively affected vegetation species richness, percentage cover of vascular plants and consequently species richness of aquatic invertebrates.

Despite their role in harbouring many species, the majority of drainage ditch ecosystems in farmlands are degraded nowadays due to excessive input of agrochemicals as a result of modern farming practices. This changes drainage ditch habitat quality from a mesotrophic system containing a species-rich mosaic of submerged, emergent and floating plant species to a hypertrophic state dominated by Lemnaceae or phytoplankton (Verdonschot et al., 2012). Nevertheless, future studies at local scale should further investigate the value of drainage ditches for diffent invertebrates and across different farming practices to determine how the effects can change according to the quantity, quality of drainage ditches and/or taxa and their interactions. This will be particularly important for landscape scale studies to determine whether measuring length or proportion of drainage ditches alone (based on aerial photogarpahs) could be used as a rapid tool to quantify ecosystem services on farmland. This will be particularly important for quantifying less mobile species such as Sciomyzidae (Williams et al., 2010) where adults remain within the vicinity of their immature stages or their gastropod hosts (e.g., aquatic snails) which are often associated with drainage diches (Knutson & Vala, 2011).

1.6. Thesis overview

The overall aim of this thesis is to assess the ecosystem function of selected dipteran families and their role as indicators of farm habitat quality in Ireland, with specific reference to the dipteran families Syrphidae and Sciomyzidae. While both taxa co-exist within many of the same habitats, they have different ecological requirements and ecosystem functions; thereby making them a potential tool to assess the value of different farm habitats for invertebrates with multiple ecosystem services. This is particularly important in developing new measures aimed at protecting and enhancing farmland biodiversity. Dipteran families of farmland which are generally less studied, have the potential to provide greater indicator potential than the more commonly studied taxa such as bees and butterflies.

The specific objectives of this thesis can be summarised as follows:

1- Study the ecosystem function of the slug killing fly *Tetanocera elata* in the laboratory and evaluate its viability in the biological control of selected pestiferous slugs either individually or in combination with a commercial biocontrol agent *Phasmarhabditis hermaphrodita.*
2- Assess the value of different linear habitats across a gradient of farming intensities in sustaining Sciomyzidae and Syrphidae flies that coexist but deliver different ecosystem services.
3- Determine explanatory variables at local and/or landscape scale that best reflect hoverfly abundance/richness with a view to incorporating them in future agri-environment schemes.

1.7. Thesis structure

Chapter 1 comprises a general introduction where the status of farmland biodiversity and measures to protect them are reviewed with particular reference to Ireland. The chapter explores the ecosystem services provided by two dipteran families (Syrphidae and Sciomyzidae) where their role as bioindicators to assess the ecosystem health of farmland habitats has been reviewed. In addition, the importance of quantity and quality of selected farm habitats in protecting and enhancing biodiversity and delivering ecosystem services is reviewed.

Chapter 2 comprises an independent original research article which has been published in Biological Control-Elsevier (Volume 135, August 2019, Pages 1-8). This chapter describes the potential of *T. elata* larvae in controlling selected pestiferous slugs. In this chapter, the first evidence of neonate *T. elata* larvae feeding on the introduced slug *Deroceras invadens* and slugs exposed to the biocontrol agent *Phasmarhabditis hermaphroditain* (Nemaslug) is presented. This study demonstrates that while *T. elata* has biocontrol potential, its survival appears to be reduced when fed (from hatching onwards) on nematode exposed slugs only or on *Deroceras invadens* only. This chapter has implications for biological control of pestiferous slugs in the context of the optimal time for using the commercial slug control agent *Phasmarhabditis hermaphroditain* in areas where *T. elata* larvae are naturally occurring.

Chapter 3 comprises an independent original research article which has been published in Insect Conservation and Diversity-Wiley (Volume 14, Issue 3, August 2019, Pages 335-347). This chapter discusses how different linear habitats across a range of farming intensities contribute to dipteran abundance and diversity. The results indicate that some of the commonly perceived less valued linear habitats for dipteran pollinators such as Syrphidae can be important for other Diptera with biological control functions such as snail/slug killing Sciomyzidae flies. the study also demonstrates that low intensity farming practices and habitat quality can play a key role in sustaining species richness and abundance of both dipteran families in agricultural lands. This study has important implications for developing new agri-enviroment measures aiming at protecting and enhancing farmland biodiversity and their associated ecosystem services including both pollination and pest control.

Chapter 4 comprises an independent original research article and has been prepared for submission to Agriculture, Ecosystems & Environment. This chapter discusses the effect of farm management and a range of environmental variables at local and landscape scale on the abundance of hoverflies and their associated functional groups. The results of this chapter indicate that at local scale, both altitude and field habitat quality score are important driving factors significantly affecting selected hoverfly group abundances/richness. In addition, at landscape scale, woody semi-natural habitats such as hedgerows and treelines are the most important driving factors positively affecting hoverflies and their functional group abundances/richness. The findings of this chapter are important in the in the context of selecting the most important variables that can be used to quantify ecosystem services on farmland at both local and landscape scale and have implications to inform future agri-environmental measures aimed at protecting and enhancing biodiversity at a national scale.

Chapter 5 is a general discussion where the findings of chapters 2-4 are discussed in a wider context with implications for applying the findings of this thesis to the development of new agri-environmental measures aimed at protecting and enhancing wider biodiversity and associated ecosystem services on farmland. This chapter also gives a road map for future research needs in this area.

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Chapter 2 Biological control of pestiferous slugs using *Tetanocera elata* (Fabricius) (Diptera: Sciomyzidae): Larval behaviour and feeding on slugs exposed to *Phasmarhabditis hermaphrodita* (Schneider, 1859)

Karzan Sabah D. Ahmed[1,2,], Claire Stephens[1,2], Allison Bistline-East[1,2], Christopher D. Williams[3], Rory J. Mc Donnell[4], Manuela Carnaghi[1,2], Daire Ó Huallacháin[5], Michael J. Gormally[1,2]

[1] Applied Ecology Unit, School of Natural Sciences, National University of Ireland Galway, Ireland; [2] Ryan Institute, National University of Ireland Galway, Ireland; [3] School of Natural Sciences and Psychology, Liverpool John Moores University, UK; [4] Dept of Crop and Soil Science, Oregon State University, Corvallis OR 97333, USA; [5] Teagasc, Environment, Soils and Land Use Department, Johnstown Castle, Co. Wexford, Ireland

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2.1. Abstract

While the larval stage of *Tetanocera elata* (Diptera: Sciomyzidae) is a known parasitoid and predator of pestiferous slugs, its biology and predatory behaviour as well as its interaction with slug parasitic nematodes requires further investigation. In this study, survival of larvae fed from the neonate stage on *Deroceras reticulatum* Müller (a previously known prey species) was significantly greater (P=0.023) than for larvae fed on *Deroceras invadens* Reise with 100% and 40% survival respectively. However, when fed solely on *D. reticulatum* which were previously exposed to *P. hermaphrodita*, less than 20% of neonate larvae pupariated successfully. Ninety percent of neonate larvae maintained without food for the first four days and subsequently fed on *D. reticulatum* pupariated successfully although this decreased to below 50% for ≥6 days without food. Predatory third instar *T. elata* larvae appeared to select nematode-exposed *D. reticulatum* over non-exposed slugs with the continued feeding on nematode-exposed slugs also reducing the chances of successful pupariation by 25%. Records of maximum egg laying by laboratory-reared female adults were greater (487 eggs) than previously recorded for field-caught adults (373). The implications of these results for the potential use of *T. elata* as a biological control agent of pestiferous slugs are discussed.

Keywords: Biological control, Pestiferous slug, *Phasmarhabditis hermaphrodita*, Sciomyzidae, *Tetanocera* elata

2.2. Introduction

Pestiferous slugs cause damage to a diverse range of agricultural and horticultural crops throughout the world (Askary, 2010, Askary et al., 2017, Cordoba et al., 2018, Fritz et al., 2001, South, 2012, Speiser et al., 2001) primarily by feeding directly on the plant (Ester and Trul, 2000, Frank, 1998, Gould, 1961, Jaskulska et al., 2017) and/or as vectors of plant pathogens (Hasan and Vago, 1966). This results in significant economic losses to growers (Willis et al., 2006).

The grey field slug, *Deroceras reticulatum* Müller, which is native to Europe but now has a worldwide distribution, is one of the most serious pestiferous slugs particularly across Europe (Godan, 1983, Tulli et al., 2009). It is also considered as one of the most important pestiferous slugs in Ireland and the UK (Tan and Grewal, 2001) due to its widespread distribution and its ability to survive in a range of habitats (Wilson et al., 1993). Another pestiferous slug, *Deroceras invadens* 2011 originated in southern Italy but has been introduced to many countries over the last century (Hutchinson et al., 2014). *D. invadens*, considered one of the most important non-native slug pests to UK agriculture (Williams et al., 2010), was first detected in Ireland in the 1950s and it is now widespread in all but extreme western districts (Anderson, 2016).

Molluscicides (generally in the form of methiocarb, metaldehyde or ferric phosphate slug pellets) have been one of the most common methods of control for pestiferous slugs, with France spending €45 million per year (Howlett, 2012) and the UK spending more than £1.5 million per year on slug pellets alone (Williams et al., 2010). However, the decision by the EU in 2014 to ban methiocarb (Commission, 2014, Commission, 2015) due to its toxic effects on non-target organisms(Jones, 2014); concerns regarding the occurrence of metaldehyde in drinking water (Busquets et al., 2014, Kay and Grayson, 2014); and more recently reports that some ferric phosphate pellets can have negative effects on other soil fauna (Castle et al., 2017) necessitate the development of alternative pestiferous slug control strategies.

Biological control offers an alternative approach to chemical pest control. Augmentative biological control is currently applied on more than 30 million ha globally, with Europe being the largest commercial market for invertebrate biological control agents (van Lenteren et al., 2018). The slug-parasitic nematode *P. hermaphrodita* is the only commercial nematode biological control agent available today in Europe as an alternative to chemical molluskicides. It is available under the trade name Nemaslug® and is sold in 15 different European countries primarily for the control of *D. reticulatum* (Pieterse et al., 2017a, Pieterse et al., 2017b). While it is a welcome addition to pestiferous slug control, some laboratory studies have shown that these nematodes kill only approximately 60% of *D. reticulatum* when applied directly onto the body of the slug or onto the soil, with efficacy likely to be even lower under field conditions. In addition, other pestiferous slug species appear to be less susceptible (Dankowska, 2006, Rae et al., 2008).

An alternative potential biological control agent of pestiferous slugs is the marsh/shade fly *Tetanocera elata* Fabricius. Knutson et al. (1965) were the first to demonstrate that while first and second instar parasitoid larvae of *T. elata* appeared to feed only on the pestiferous slug species *D. reticulatum* and *D. laeve*, predatory third instars could feed on a range of slug species. Since then, Hynes et al., 2014a, Hynes et al., 2014b, Hynes et al., 2014c) have contributed significantly to our knowledge of the biology of *T. elata* with regard to adult oviposition rates and longevity (using predominantly field-caught adults) and temperature effects on egg development (2014a), and the duration of the larval stages (2014b). Hynes et al. (2014c) also undertook preliminary larval feeding behaviour studies. Nevertheless, the biology of *T. elata* still needs to be understood in a systematic manner to exploit this species against different pestiferous slugs.

The aims of this study are to:

1- Quantify the fecundity of laboratory-eclosed female *T. elata* to inform the design of potential mass cultures of the species (previous oviposition studies by Knutson et al. (1965), Beaver, 1973, Hynes et al., 2014a used predominantly field-caught adults).

2- Determine whether the obligate parasitoid first instar life stage of *T. elata* can feed and survive to pupal stage on a pestiferous slug species (i.e., *D. invadens*) not previously documented as a potential host for *T. elata*.

3- Assess the impacts of *P. hermaphrodita*-exposed slugs on the development of neonate *T. elata* larvae to determine the potential for both control agents to be used synergistically for pestiferous slug control.

4- Determine, using prey-choice experiments, the susceptibility of slugs exposed to *P. hermaphrodita* to predation by third instar *T. elata* larvae.

5- Assess the effects of the absence of prey over different periods on neonate larval development and survival.

6- Investigate the feeding behaviour s of first and third instar larvae to inform our understanding of how *T. elata* larvae locate pestiferous slugs (second instar larvae develop inside the first slug host and, therefore, do not exhibit predatory behaviour).

2.3. Materials and methods

2.3.1. Field collections and maintenance of laboratory cultures of T. elata

Adult *T. elata* flies were collected from extensive, low input grasslands in Co. Galway in the west of Ireland (lat: 53.289750, long: −9.066056) using a sweep net and pooter on July 12th and 23rd (2017). Captured flies were sexed, paired as female-male couples, and maintained in the laboratory, in covered glass jars (11.6 × 6 cm) containing water and food consisting of a mixture of honey and brewer's yeast in a 3:1 ratio (Hynes et al., 2014a). The number of eggs laid per day was recorded and eggs were transferred to Petri dishes (55 mm × 15 mm) containing filter paper on top of damp cotton wool. Parafilm® (Bemis Parafilm M™) was used to seal the Petri dishes, which were then kept at room temperature (∼19–20 °C) until egg hatching.

2.3.2. Slug collections and maintenance

D. reticulatum and *D. invadens* were collected using slug metric traps (de Sangosse Pont du Casse, France) placed on grasslands at a number of locations in Co. Galway (lat: 53.277083, long: −9.062333 and 53.279833, −9.056778). Slugs collected in the field were maintained in the laboratory in plastic boxes (16.5 × 11.4 × 5 cm) on damp tissue covered with filter paper on which a piece of carrot was placed. Boxes were cleaned every 3-4 days at which stage food was replaced and filter paper changed, and any dead slugs were removed from the box. Slug cultures were maintained for at least 21 days post-collection, after which they were examined to ensure none of the slugs displayed symptoms of naturally-occurring infections of the nematode *P. hermaphrodita* or other congeneric malacophagous species (Carnaghi et al., 2017). For those experiments which investigated the impact of nematode-exposed slugs on *T. elata* behaviour, predation and subsequent development, it was essential that field-collected slugs were deemed to be nematode-free before experiments commenced.

2.3.3. T. elata neonate larval development and survival on control slugs and slugs exposed to P. hermaphrodita

Using the procedures of Carnaghi et al. (2017), each slug was exposed to a uniform suspension of *P. hermaphrodita*. For every suspension used, three subsamples of similar volumes were created and the number of moving nematodes in each subsample were counted under a stereo microscope. The average number of nematodes per subsample was then calculated to ensure that 30 nematodes/cm^2 (approx.) were applied directly to the mantle of each slug using a micropipette (see Carnaghi et al., 2017 for details). Two days later, a neonate *T. elata* larva was placed directly on the mantle of each of 11 nematode-exposed *D. reticulatum* and 11 nematode-exposed *D. invadens* slugs. Similarly, a neonate *T. elata* larva was placed directly on the mantle of each of 10 control (nematode-free) *D. reticulatum* and 10 *D. invadens* slugs. *T. elata* larval development and survival were subsequently recorded until pupariation or death.

2.3.4. Effect of absence of prey on neonate T. elata larval development and survival

Preliminary experiments using 50 neonate larvae (maintained in batches of 10) were placed in Petri dishes containing a damp cotton pad covered by filter paper and maintained without prey to determine neonate larval survivorship in the absence of food. Based on the results of these experiments the impact of the absence of prey for 4, 6, 8 and 10 days on the subsequent number of slugs killed and the duration of *T. elata* life-cycle stages was recorded using the pestiferous slug species *D. reticulatum* and *D. invadens*. Ten replicates were conducted for each slug species and for each period without food (total of 100 experiments including controls) and these were monitored until larvae pupariated or died before pupariation.

2.3.5. Comparison of P. hermaphrodita-exposed and control slugs (D. reticulatum) on third instar T. elata prey choice

Feeding trials of third instar larvae on nematode-exposed versus control *D. reticulatum* were undertaken to determine whether nematode-exposed slugs were more or less susceptible to predation by third instar *T. elata* larvae. Slugs (n = 11) of similar weights were distinguished from each other using colored elastomers (blue for exposed slugs and orange for control) injected just below surface of the foot of the slug (Mc Donnell and Gormally, 2011). One exposed and one control slug were then placed together with one third instar *T. elata* larva in individual Petri dishes which were sealed with Parafilm® to prevent the slugs escaping. Larval food choice was recorded, and subsequent slugs provided thereafter reflected the initial food choice (i.e., a control or nematode-exposed slug). This process continued until the larvae pupariated or died before pupariation.

2.3.6. Larval feeding behaviour

While the experiments described in 2.4 (above) examined the effects of absence of prey on *T. elata* development and survival, further experiments were undertaken to examine the feeding behaviour of larvae after different durations without prey. With this in mind, the attacking and feeding behaviour of neonate and third instar *T. elata* larvae were observed using *D. reticulatum* and *D. invadens*. The latter was selected since, to date, the first (and second) instar of *T. elata* has been recorded as a known obligate parasitoid of *D. reticulatum* and *D. laeve* only (Knutson et al., 1965). Trials began on 14th November 2017 using neonate larvae hatched from eggs collected from laboratory-reared, second generation adults. Separate behaviour al experiments using *D. reticulatum* and *D. invadens* were conducted by exposing neonate *T. elata* larvae to a slug on the day of hatching, 3 days after hatching, and 6 days after hatching. In each case, 10 replicates for each slug/*T. elata* larval treatment were conducted resulting in 60 replicates in total with each slug being weighed before the experiments commenced. Each replicate was directly observed for up to a maximum of 8 h and larval behaviour categorized according to Hynes et al. (2014c). In cases where larvae did not feed during the 8-hour observation period and had not fed by the following morning, new experiments were set up using fresh slugs and new *T. elata* larvae to ensure the completion of 60 successful feeding experiments in total.

Feeding behaviour trials with third instar *T. elata* specimens were also undertaken using larvae which had been fed on either *D. reticulatum* only or *D. invadens* only from the time of hatching. Once the larvae reached third instar, they were not fed for 3–8 days to maximize the chances that larvae would demonstrate predatory behaviour (Hynes et al., 2014c). Fifteen replicates for *D. invadens* using larvae previously fed on *D. invadens* only and 12 replicates for *D. reticulatum* using larvae previously fed on *D. reticulatum* only were used for this trial. Third instar trials took place in a darkened room and were recorded using the “Night shot” setting on a SONY Handycam FDR – AX33. This permitted direct observations in darkened conditions which would be more similar to field situations. Using this approach, however, was not possible for first instar behaviour al trials due to their small size.

2.3.7. Statistical analyses

While *T. elata* occurs widely in the Palaearctic region, it has a patchy distribution because of which it can sometimes be challenging to find specimens in large quantities in the field. Nevertheless, where there were sufficient numbers of replicates in this study, statistical comparisons between the treatments were undertaken in SPSS (IBM, SPSS Statistics v. 24) using non-parametric Mann-Whitney U tests. The Spearman’s rank correlation coefficient was used to determine correlations between the treatments. Statistical differences in larval feeding behaviours were predicted using a chi square test. All data were analysed at the $P < 0.05$ standard level of significance.

2.4. Results

2.4.1. Egg-laying by laboratory-reared T. elata

The mean number of eggs laid per female was 291.4 eggs (±50.5 SE) (Fig. 1.1) with a total of 1457 eggs laid over a period of 38 days by five paired individuals that emerged from laboratory-reared pupae. The mean pre-oviposition period was 5.2 days (±0.7 SE) and the total number of eggs laid by individual females ranged from 207 to 487 eggs (Fig. 2.1), with the numbers laid each day ranging from 1 to 46 eggs per female. The mean duration of the egg stage (n = 31) at laboratory/room temperature was 9.3 days (±0.17). No significant correlations were found between the age at which the males died and total number of eggs laid by the females (R = 0.82, P = 0.089, n = 5) nor between the age at which the females died and the number of eggs laid (R = 0.5, P = 0.391, n = 5).

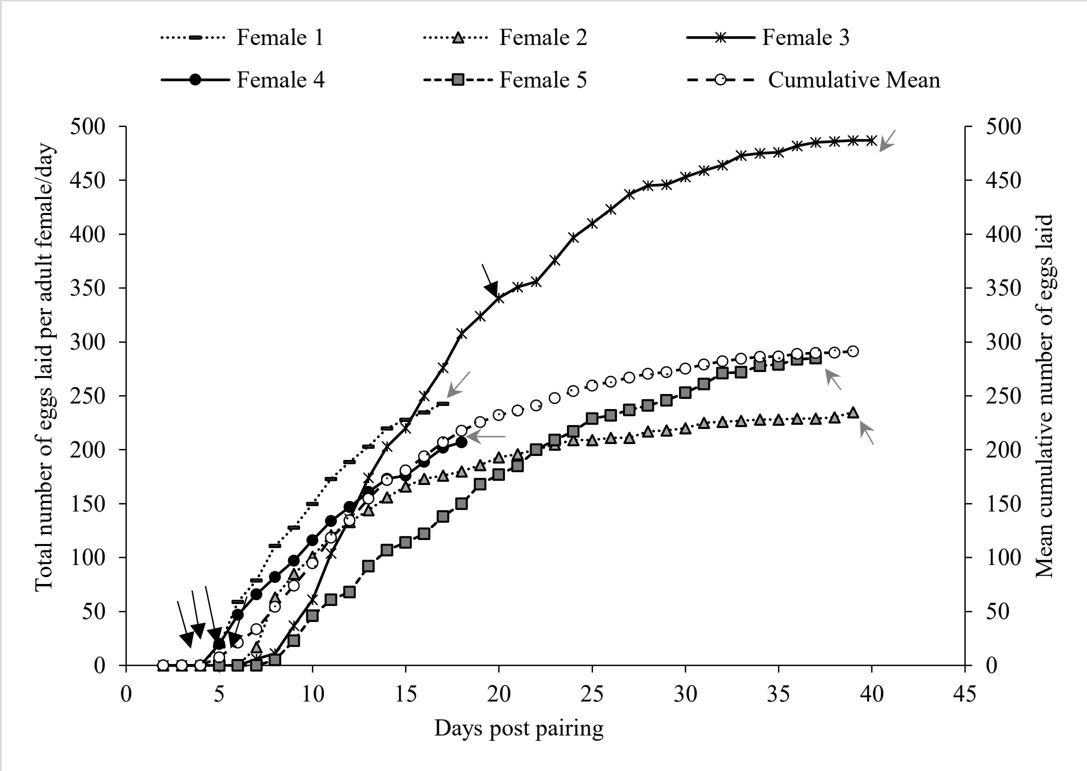

Figure 2.1. Mean cumulative and total cumulative number of eggs laid per day by 5 laboratory-reared *T. elata* adult females. The date each fly died is marked with an arrow (black = males; grey = females).

2.4.2. T. elata neonate larval development and survival on control slugs and slugs exposed to P. hermaphrodita

This experiment demonstrated that neonate *T. elata* larvae were able to feed successfully on another slug species (i.e., *D. invadens*) apart from *D. reticulatum* and *D. laeve* (as previously described by Knutson et al. [1965]). In larvae reared on control slugs, the pupariation rate was, however, significantly (U = 20, P = 0.023) less when larvae fed on *D. invadens* only (just four out of 10 pupariated successfully) compared to *D. reticulatum* only (all 10 larvae pupariated successfully) (Fig. 2.2). In addition, the duration of the larval stage was significantly longer (U = 2, P = 0.008) for *D. invadens*-reared larvae than for *D. reticulatum*-fed larvae (Table 2.1). Despite this, there were no significant differences between slug species tested regarding the weight of slug killed per larva (U = 15.5, P = 0.53) and pupal weight (U = 11, P = 0.24) (Table 2.1).

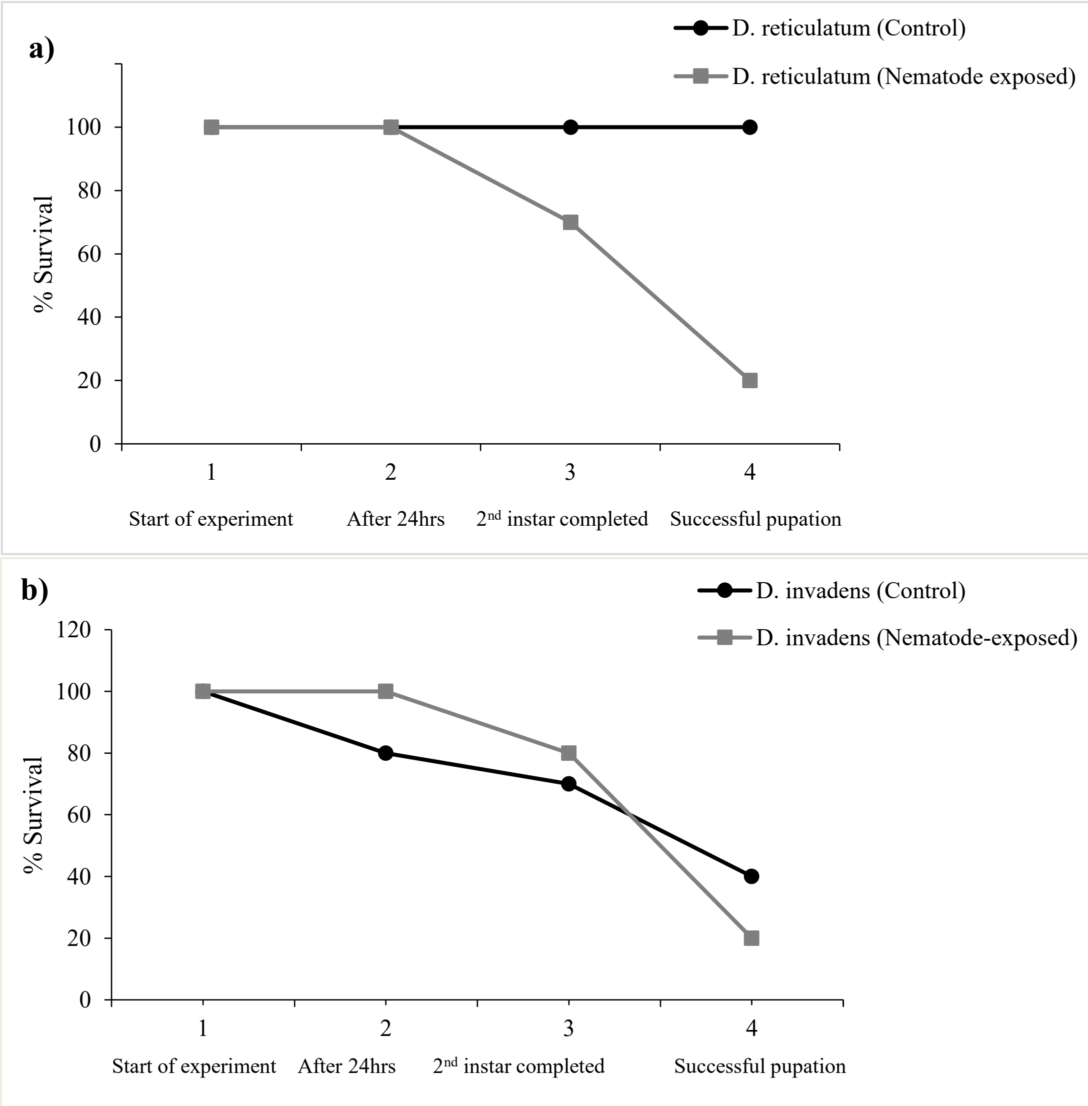

Figure 2.2. Survivorship of neonate larvae to puparial stage when fed solely on either (a) *D. reticulatum* or (b) *D. invadens* control (n = 10) and nematode-exposed slugs (n = 11).

Neonate survivorship to puparial stage was less successful where *D. reticulatum* was exposed to *P. hermaphrodita* with only two out of 11 (18.2%) larvae pupariating successfully compared to 100% successful pupariation using control slugs (Fig. 2.2a). There was little difference in mean larval period and mean pupal weight between control and nematode-exposed *D. reticulatum*, although mean slug weight killed per larva was almost 40% greater for larvae reared on control versus nematode exposed *D. reticulatum* (Table 2.1). Similar poor pupariation rates (Fig. 2.2b) and low slug weights killed per larva were observed for neonate larvae fed on *D. invadens* exposed to the nematodes, in addition to which mean larval period was considerably reduced in nematode-exposed *D. invadens* compared to controls (Table 2.1).

Table 2.1. *T. elata* development from neonate to pupa when fed on either *D. reticulatum* or *D. invadens* using control slugs (Control) or slugs exposed to *Phasmarhabditis hermaphrodita* (Ph).

***Tetanocera elata* development**	***Deroceras reticulatum*** (Mean + SE)		***Deroceras invadens*** (Mean + SE)	
	Control n = 10	**Ph** n = 2	**Control** n = 4	**Ph** n = 2
Mean larval period (day)	38.20 ± 1.25	40.00+ 3.00	51.00 ± 4.64	38.00 ± 5.00
Mean number of slugs killed / larva	8.30 ± 0.63	7.50 + 0.50	9.80 ± 0.85	5.50 + 0.50
Mean slug weight killed / larva (g)	0.27 ± 0.01	0.17 + 0.01	0.28 ± 0.01	0.17 + 0.01
Mean pupal weight (g)	0.05 ± 0.00	0.05 ± 0.00	0.04 ± 0.00	0.03 ± 0.00

2.4.3. Comparison of availability of P. hermaphrodita nematode-exposed and control slugs (D. reticulatum) on third instar T. elata prey choice

When offered the choice of either control *D. reticulatum* or *D. reticulatum* exposed to nematodes, third instar larvae chose to feed more frequently on nematode-exposed slugs than control slugs (χ^2 (n = 11) = 4.54, P = 0.033). However, while the three larvae which initially selected a control slug (and were subsequently fed on control slugs only) pupariated successfully, only six of the eight larvae which initially selected a nematode-exposed slug (and were subsequently fed on nematode-exposed slugs only) pupariated successfully.

2.4.4. Effect of absence of prey on neonate T. elata larval development and survival

While more than 95% of neonate *T. elata* larvae (n = 50) survived without prey for seven days (Fig. 2.3), by day 14 all larvae were dead. The impact of different duration periods without food post-egg-hatching on subsequent larval development showed that successful pupariation was significantly more frequent for neonate larvae without food for 4 days (subsequently fed on *D. reticulatum*) compared to neonate larvae maintained without food for the first 6 days (χ^2 (n = 10) = 5.49, P = 0.019) as well as 8 and 10 days respectively (χ^2 (n = 10) = 9.9, P = 0.0016, for both 8 and 10 days) (Fig. 2.4).

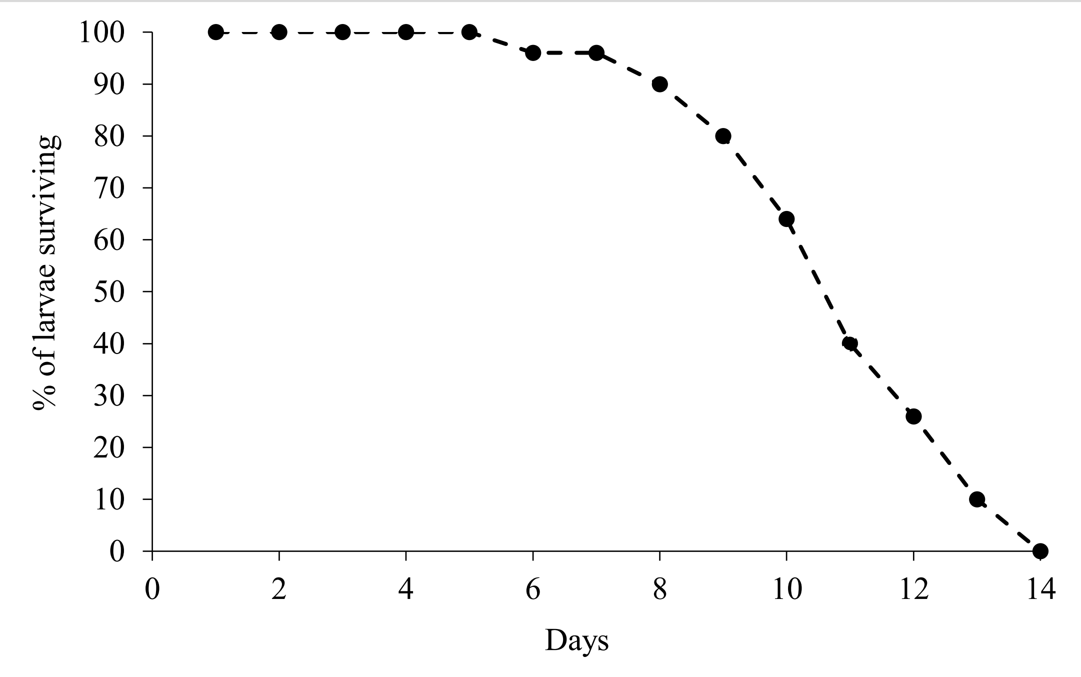

Figure 2.3. Survivorship of neonate *T. elata* larvae in the absence of slug prey (n = 50).

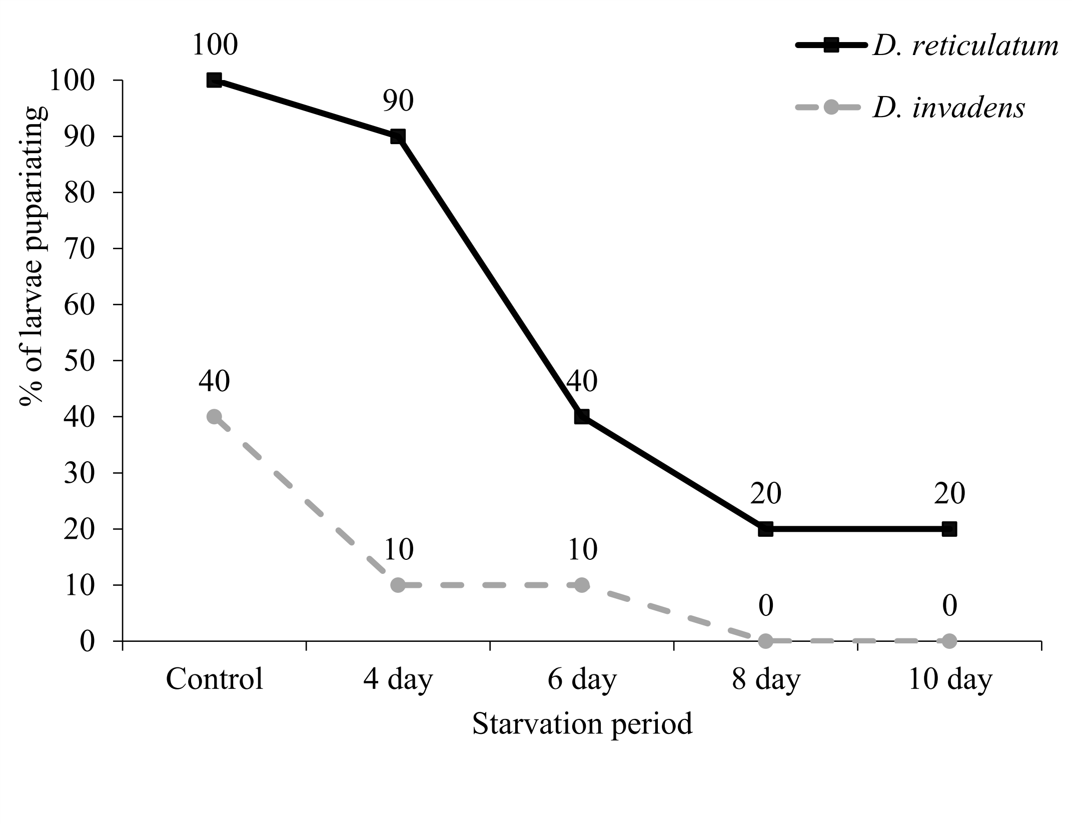

Figure 2.4. Impact of absence of prey from 4 to 10 days during the neonate larval stage on *T. elata* pupariation success (n = 10).

The pupariation success of larvae fed on *D. invadens* in comparison with *D. reticulatum* after four days was also significantly (χ^2 (n = 10) = 12.8, P = 0.0003) lower. Only 10% of larvae fed on *D. invadens* pupariated successfully after 4 and 6 days without food, and none pupariated after 8 and 10 days without food (Fig. 2.4). While it is difficult to draw firm conclusions due to larval mortalities, there appears to be little difference in mean slug weight killed per larva between controls and those larvae where food was absent at the neonate stage regardless of the slug species tested (Table 2.2).

Table 2.2. Comparison of neonate larval period to pupariation, mean number of slugs killed per larva and mean weights of slugs killed per larva feeding on either *D. reticulatum* (*Dr*) or *D. invadens (Di)* *.

Absence of prey (days)	**0 (Control)**		**4**		**6**		**8**		**10**	
	Dr	Di	Dr	Di	Dr		Dr	Di	Dr	Di
	n = 10	n = 4	n = 9	n = 1	n = 4	n = 1	n = 2	n = 0	n = 2	n = 0
Mean larval period (days)	38.2 ± 1.25	51.0 ± 4.64	37.0 ± 1.05	*30.0 ± 0.00*	37.5 ± 1.44	*34.0 ± 0.00*	43.5 ± 0.50	L. Died	37.0 ± 1.00	L. Died
Mean number of slugs killed / larva	8.3 ± 0.63	9.8 ± 1.71	7.9 ± 0.61	*9.0 ± 0.00*	8.5 ± 0.96	*9.0 ± 0.00*	6.5 ± 0.50	L. Died	10.0 ± 3.00	L. Died
Mean slug weight killed / larva (g)	0.27 ± 0.01	0.28 ± 0.01	0.27 ± 0.02	*0.3 ± 0.00*	0.25 ± 0.01	*0.27 ± 0.00*	0.25 ± 0.01	L. Died	0.23 ± 0.02	L. Died

* While each experiment commenced with 10 larvae, *n* in each column represents the number of larvae which pupariated successfully. Where n = 1, raw data are presented in italics.

2.4.5. Larval feeding behaviour

Given that no obvious differences in feeding behaviour were observed between larvae which were allowed to feed on both slug species immediately after hatching and those starved for up to 8 days (see Appendix I), these data were combined to give an overall picture of larval feeding behaviour. An examination of the predation strategies adopted by *T. elata* larvae on *D. reticulatum* and *D. invadens* revealed significant differences between neonate ($n = 60$) and third instar ($n = 27$) larvae ($\chi^2 = 18.67$, $df = 2$, $P < 0.05$) (Fig. 2.5). Over 50% of neonate larvae simply waited for slugs to pass nearby before attacking with 35% searching first followed by waiting, and 13.3% actively searching and attacking slug prey. In comparison, 59.3% of third instar larvae actively searched and attacked slugs with 22.2% actively searching and then waiting; and only 18.5% simply waiting for a slug to pass by. The mode of attacking, latching on (position adopted by larva prior to immobilizing the slug Hynes et al., 2014c) and feeding also varied according to the larval stage (Fig. 2.6). While there appeared to be little difference in the location on the slug that was attacked by neonate larvae, the majority (18 out of 27 i.e., 66.7%) of third instar larvae attacked slug tails. The tail of the slug was also the most popular location for latching on by third instar larvae but was the least popular location for latching on by neonate larvae. While the latero-ventral surface of the slug was the most frequently recorded feeding site for neonate larvae, there was no clear preference among third instar larvae. All the neonate larvae were subsequently found feeding under the slug mantle one day after the feeding trials commenced.

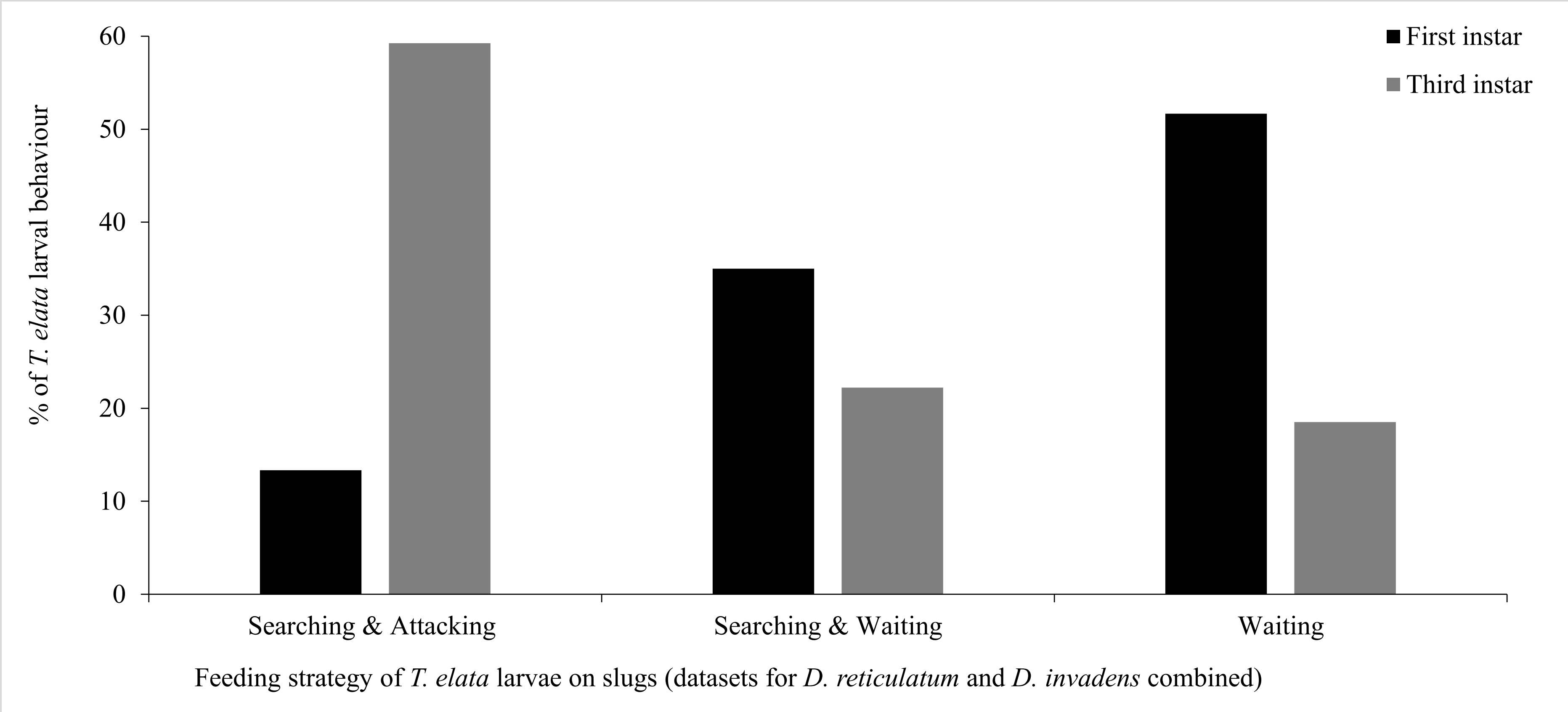

Figure 2.5. Predatory strategies (Searching & Attacking, Searching & Waiting and Waiting) adopted by *T. elata* larvae (n = 60 neonate larvae; n = 27 third instar larvae) in the presence of slugs (*D. reticulatum* and *D. invadens* datasets combined).

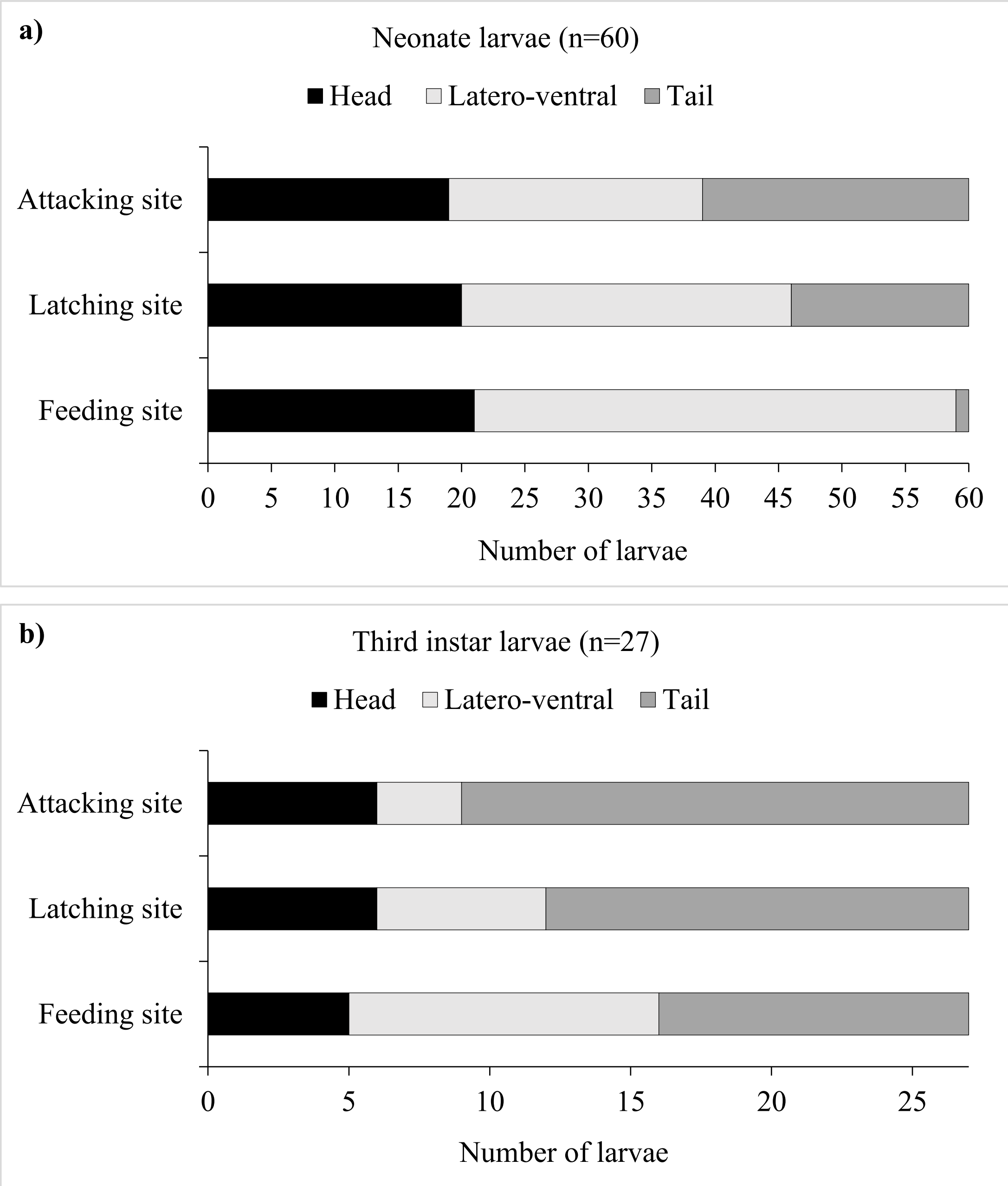

Figure 2.6. Attacking, latching on* and feeding locations of (a) neonate and (b) third instar *T. elata* larvae on slugs (*D. reticulatum* and *Deroceras invadens* datasets combined) (black = head of slug; light grey = latero-ventral surface of slug; grey = tail of slug).

*Latching on: Position adopted by larva before immobilising the slug (after Hynes et al., 2014c).

2.5. Discussion

The number of eggs laid by laboratory-reared flies ranged from 207 to 487 eggs, the latter being greater than the maxima (373, 196 and 285 eggs) previously recorded by Knutson et al., 1965, Beaver, 1973, Hynes et al., 2014a respectively. Knutson et al. (1965) suggested that repeated copulation is not necessary for fertilizing large numbers of eggs and this is supported by Hynes et al. (2014a), who recorded one field-caught female laying 166 eggs (44% of which subsequently hatched) without a male partner in the laboratory. However, in the current study, one of the females where the male partner survived longest, and which was observed pairing repeatedly, laid 487 eggs. The potential effects of repeated mating on egg production requires further investigation to ensure maximum egg production in large scale cultures for biological control programs.

Prior to commencing the behaviour experiments, the survivorship of neonate *T. elata* larvae in the absence of prey was measured. Our results showed that larvae can survive without food for up to two weeks, although their ability to feed and survive on a host is greatly reduced after six days without food to below 40%. This contrasts with the findings of Knutson et al. (1965) who stated that neonate *T. elata* larvae die after four or five days without food. This could be due to their experiments being conducted in laboratories in continental Europe where late summer and early fall temperatures can be higher than in the west of Ireland, thereby increasing metabolic rates (Gormally, 1988). Knutson et al. (1965) also stated that *T. elata* neonates are host-specific to only *D. reticulatum* and *D. laeve*. While the results of our study demonstrate that *T. elata* neonate larvae can be reared on both *D. invadens* and *D. reticulatum*, larval survivorship to pupariation was greatly reduced on *D. invadens* compared to *D. reticulatum*. Additionally, larval development was significantly longer when larvae were reared on control *D. invadens* than on control *D. reticulatum* despite there being little difference in mean slug weight killed or final puparial weight between slug species. While this could be due to differences in the nutritional value (prey quality) between both control slugs, similar mean larval durations for *T. elata* larvae when fed on nematode-exposed *D. reticulatum* and *D. invadens* suggests that more than one factor may be at play and further experiments with more replicates are required to provide an answer for this. This ability of neonate larvae to feed on a number of pest slug species makes *T. elata* more important as a biological control agent against pestiferous slugs. While the reasons for greater mortality of larvae feeding solely on *D. invadens* are yet unknown, third instar larvae in the wild would likely have a choice of other slug species on which to feed which could improve chances of successful pupariation but further experiments are required to address this question.

Of the three larval feeding behaviour strategies observed (searching & attacking, searching & waiting, and waiting; see Hynes et al. (2014c)), more than half (51.7%) of neonate larvae exhibited the “waiting” response, while for third instar larvae, almost 60% of larvae displayed the “searching & attacking” response without any physical contact being made by the third instar larva with a slug. Foraging animals are typically classified as either “ambush”, in which they stay motionless for long periods of time waiting for their prey to pass by, or “cruise” foragers, searching actively for their prey (O’Brien et al., 1990). We show here that *T. elata* displays different strategies during different life stages, which is supported by Knutson and Vala (2011) who have noted the labile nature of feeding behaviour in Sciomyzidae in general. Since *T. elata* is an obligate parasitoid in its first and second larval stages, the neonate waiting response probably reflects general parasitoid behaviour. This type of behaviour has been observed in the juvenile stage of parasitoid waspswhen in the vicinity of their hosts (Mohamad et al., 2015). Waiting also suggests the behaviour of a generalist parasitoid (as evidenced by neonate *T. elata* larvae feeding on more than one species of slug) since specialist parasitoids tend to actively search and find their hosts (Wang and Keller, 2002). The preference among third instar larvae reflects more the behaviour of predators (searching & attacking) and this is supported by previous studies on third instar larval behaviour by Hynes et al. (2014c). The latero-ventral region of the slug appears to be the preferred area for “latching on” and “feeding” by neonate *T. elata* larvae while the slug tail region for both “attacking” and “latching on” appears to be preferred by third instar larvae. No obvious trend was discernible for sites of “attack” for neonate larvae or sites of “feeding” for third instar. While Hynes et al. (2014c) studied the behaviour of only limited numbers of third instar larvae, they found that the slug tail was also used for latching on but was less popular than the latero-ventral surface of the slug. Their findings are in contrast with the results of the current study where the slug tail appeared to be the most preferred site for attacking and latching on. Further investigation is required to explore contributing factors to variations in individual predatory strategies since "searching & attacking" gives a clear advantage for an effective biological control agent when used by an active predator (Matthews and Matthews 2009).

Since many neonate larvae wait until a slug passes before attacking, it is vital to determine the length of time a neonate larva can survive before successfully obtaining its first slug host. Not surprisingly, neonate larvae reared on *D. invadens* had low survival rates up to pupariation (10%) after four and six days without food and none survived after eight and ten days without a host. In contrast, neonate larvae reared on *D. reticulatum*_resulted in 90%, 40%, 20%, and 20% survival up to pupariation after four, six, eight, and ten days (respectively) without a host. It is interesting to note a 90% survival rate after four days without a host for larvae reared on *D. reticulatum*, indicating a likely adaptation to facilitate the "waiting" behaviour exhibited by many neonate larvae. This survival period is likely to be longer in the wild where temperatures would frequently be below those recorded in the laboratory where these experiments were conducted.

Given that *P. hermaphrodita* is currently the only commercially-available nematode biocontrol agent of slugs on the market (Askary et al., 2017), the implications of using this in areas where *T. elata* is naturally-occurring is of interest. The key question here is whether both *T. elata*_and *P. hermaphrodita* could be used in tandem as a part of an integrated pest management (IPM) approach or whether slugs exposed to *P. hermaphrodita* may have an adverse effect on *T. elata* larvae. The survivorship of neonate larvae to pupariation when reared only on nematode-exposed *D. reticulatum* was just 20% compared to 100% survivorship for controls, while in neonates reared on similarly exposed *D. invadens* survivorship was also 20% in comparison to 40% for controls. The average number of slugs killed, larval period, and puparial weight by both control and nematode-exposed *D. reticulatum* did not show any major differences. Although mean larval duration was shorter and mean weight of slug killed per larva was less for nematode-exposed *D. invadens*, it is difficult to make concrete inferences for *D. invadens* given the low number of individuals which survived to pupariation (n = 2).

Where third instar larvae were given the choice of feeding on either control or nematode-exposed *D. reticulatum*, they showed a preference for the nematode-exposed slugs. One possible explanation for this is that *P. hermaphrodita*-exposed slugs could be immunocompromised making them an easier target for *T. elata* larvae, but further research needs to be undertaken using recorded trials for review and confirmation of results to prove this definitively. Only 75% (six out of eight) of those third instar larvae which selected nematode-exposed slugs and were subsequently fed only nematode-exposed slugs pupariated successfully compared to successful pupariation for all third instar larvae fed on control *D. reticulatum*. The fact that third instar larvae may preferentially select nematode-exposed *D. reticulatum* over non-exposed individuals with a subsequent lower than expected outcome for successful pupariation may be of some concern. Nermuť et al. (2014) shows that *P. hermaphrodita* use slug tissue as a nutrient-rich source for reproduction and it is possible that this may affect the subsequent nutritional value for *T. elata* larvae. However, this experiment needs to be repeated with a greater number of replicates and where third instar larvae are permitted to choose between nematode-infected and non-infected slugs each time they attack a new slug. Limited numbers of larvae prevented such experiments being conducted in this study. Similarly, the no-choice feeding trials where the neonate larvae were given no choice other than to parasitize nematode-exposed slugs resulted in very low survivorship to pupariation (20%). This low survivorship could possibly be due to a deterrent effect by the nematode-bacteria complex which defends the host slug against being predated or scavenged by other organisms as a result of bacterial metabolites associated with the nematodes (Pechova and Fontan, 2008, Foltan and Puza, 2009). On the other hand, given the findings of Wilson et al. (1994) and Rae et al. (2010) that *P. hermaphrodita* has large amounts of associated bacteria, the possibility that they produce toxins that affect *T. elata* larvae directly requires further investigation.

2.6. Conclusions

Our results show that the maximum number of eggs laid by laboratory-eclosed *T. elata* females is greater than previously recorded for field-caught females and that neonate larvae are capable of parasitizing slug species (i.e., *D. invadens*) other than the previously recorded *D. reticulatum* and *D. laeve*. Neonate larvae, which primarily exhibit a “waiting” strategy to find a slug host, can survive more than

10 days without food. However, larvae have a greater chance of reaching the puparial stage if they feed on *D. reticulatum* throughout the larval stage than if they feed on *D. invadens*. Results also indicate that while predatory third instar *T. elata* larvae appear to select nematode-exposed *D. reticulatum* over non-exposed slugs, continued feeding on nematode-exposed slugs during development reduces the chances of successful pupariation. While this study suggests that feeding on *D. invadens* and nematode-exposed slugs can reduce the chances of *T. elata* pupariating successfully, further work involving prey-choice experiments throughout the larval stage is required. This will be particularly important in predicting biotic factors (e.g., prey type) that may determine shifts in *T. elata* populations in the field which consequently may affect pestiferous slug populations.

2.7. Credit authorship contribution statement

Karzan Sabah D. Ahmed: Conceptualization, Methodology, Validation, Formal analysis, Investigation, Resources, Data curation, Writing - original draft, Writing - review & editing, Visualization. Claire Stephens: Conceptualization, Methodology, Writing - review & editing, Formal analysis, Investigation, Resources, Data curation, Visualization. Allison Bistline-East: Conceptualization, Methodology, Writing - review & editing. Christopher D. Williams: Conceptualization, Methodology, Writing - review & editing, Validation, Supervision. Rory J. Mc Donnell: Methodology, Writing - review & editing. Manuela Carnaghi: Methodology, Writing - review & editing. Daire Ó Huallacháin: Writing - review & editing, Supervision, Project administration, Funding acquisition. Michael J. Gormally: Conceptualization, Methodology, Validation, Writing - review & editing, Supervision, Project administration, Funding acquisition.

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2.9. References

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Chapter 3 Linear habitats across a range of farming intensities contribute differently to dipteran abundance and diversity

Karzan S. D. Ahmed[1,2], Alessio Volpato[1,2], Michael F. Day[1,2], Collette J. Mulkeen[1,2], Aidan O'Hanlon[1,2] , John Carey[3], Christopher Williams[4], Sara Ruas[5], James Moran[5], Roser Rotchés-Ribalta[6], Daire Ó hUallacháin[6], Jane C. Stout[7], Simon Hodge[7], Blanaid White[8], Michael J. Gormally[1,2]

[1]Applied Ecology Unit, School of Natural Sciences, National University of Ireland Galway, Ireland; [2]Ryan Institute, National University of Ireland Galway, Ireland; [3]National Parks and Wildlife Service, Agri Ecology Unit, Flood St Galway; [4]School of Biological and Environmental Sciences, Liverpool John Moores University, UK; [5]Department of Natural Sciences; Galway Mayo Institute of Technology, Galway, Ireland; [6]Teagasc Environment, Soils and Land use Department, Johnstown Castle, Co Wexford, Ireland; [7]School of Natural Sciences, Trinity College Dublin, Dublin, Ireland; [8]School of Chemical Sciences, Dublin City University, Dublin, Ireland.

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3.1. Abstract

While the value of linear farm habitats for the protection and enhancement of farmland biodiversity in general is known, less is understood about their contribution to Diptera, especially those with different ecological requirements. In this study, we examined the impact of a range of linear farm habitats in agricultural grassland on Syrphidae and Sciomyzidae (Diptera) both of which provide important ecosystem services.

Species richness and abundance for each family were measured across five different linear habitat types (dense and open hedgerows with/without adjacent watercourses and watercourses only).

While dense hedgerows with adjacent watercourses showed the greatest numbers of Syrphidae species and individuals, open hedgerows with adjacent watercourses had significantly more Sciomyzidae species and individuals than dense hedgerows without watercourses or open hedgerows only. Syrphidae species richness was significantly correlated with the flowering plant species richness of linear habitats, while Sciomyzidae species richness was correlated with a habitat quality score for grasslands adjacent to the linear habitats. Overall, Syrphidae and Sciomyzidae species richness and community composition are shown to reflect a farming intensity gradient with significantly greater species richness for both families on extensive farms. Our results suggest that different dipteran taxa which utilise linear habitats may have different requirements. This has implications for future agri-environmental schemes in the context of how we assess the conservation value of linear habitats on farmlands.

Key words: Biodiversity, farmland habitats, hedgerows, watercourses, habitat quality, Syrphidae, Sciomyzidae, Bi-directional Malaise traps.

3.2. Introduction

The global decline in biodiversity, due to anthropogenic activities such as climate change and land use intensification is now acknowledged (IPBES, 2019), with approximately 60% of global ecosystems damaged or overexploited beyond their capacity to recover (Brickhill, 2015). Agricultural ecosystems, in particular, have been subject to significant increases in farming intensity, one of the primary causes of the rapid decline in farmland biodiversity over the past two decades (Robinson & Sutherland, 2002; Benton et al., 2003; Larkin et al., 2019). In Europe, agricultural area accounts for approximately 42% of total land (European Environment Agency, 2018) of which less than 40% is categorised as low intensity management (Eurostat, 2019). The intensification of agriculture has negatively affected not only farmland biodiversity but also associated ecosystem services, including those linked to food production such as pollination and biocontrol (Stoate et al., 2009; Cole et al., 2020).

Linear farm habitats such as hedgerows and watercourses are important habitats on farmland and have attracted considerable interest amongst conservationists in recent years due to their value as habitats for biodiversity (Tattersall et al., 2002; Brooks et al., 2012) and their role as wildlife corridors (Coulthard et al., 2016). Hedgerows provide valuable resources for wild bees (Stanley & Stout, 2014; Ponisio et al., 2017), butterflies (Cole et al., 2017) and dipteran families with high mobility such as hoverflies (Haenke et al., 2014; Garratt et al., 2017). Hedgerows are also considered as one of the most valuable semi-natural linear habitats on many farms, contributing significantly to the biodiversity of farmland (Baudry et al., 2000; Dover, 2019). On many intensive farming landscapes, they are the only remaining semi-natural habitat that can provide a valuable habitat for wildlife and deliver essential ecosystem services (Dover, 2019; Larkin et al., 2019). For this reason, hedgerows are given protection in several European countries including Ireland (Baudry et al., 2000) where hedgerows cover 4% of the total land area (Forest Service, 2018). Hedgerows can provide important food sources for pollinators and natural enemies during periods when crops are absent or not in flower (Cole et al., 2017; Dover, 2019). Moreover, hedgerows can provide additional resources including prey/hosts, shelter, breeding sites and protection from pesticides (Dover, 2019). Dense continuous hedgerows that are diverse in woody species and floral resources are generally considered to be good quality hedgerows and are recognised as important habitats for invertebrates with strong mobility such as bumblebees (Garratt et al., 2017; Volpato et al., 2020) and hoverflies (Garratt et al., 2017). However, little is known about the effect of dense hedgerows on flying insects with low mobility (Burel et al., 2004) and studies on whether dense hedgerows can act as barriers to movement for weak flying insects (e.g., parasitoids) are lacking (Dover, 2019), particularly in agricultural lands with different levels of farming intensity.

Watercourses (e.g., drainage ditches, streams) on farmland can also provide valuable habitats as well as food sources to both aquatic and terrestrial invertebrates (including rare species), especially in dry and intensive farmlands where food sources are limited (Herzon & Helenius, 2008). In addition, they play an important role in habitat connectivity within wider landscapes and their function in regulating water flow and nutrient retention is likely to depend on the biological communities of watercourses (Herzon & Helenius, 2008). However, while good quality hedgerows (e.g., dense hedgerows) are known to support some invertebrate groups (Garratt et al., 2017; Graham et al., 2018), less is known about terrestrial invertebrates associated with watercourses (drains/streams) particularly the non-iconic insect groups (Kleijn & Van Langevelde, 2006). Moreover, studies evaluating the value of linear farm habitats for invertebrates often focus on individual habitats such as hedgerows or watercourses separately (Kleijn & Van Langevelde, 2006; Wolton et al., 2014; Garratt et al., 2017), whereas studies evaluating the combined effects of both habitats on farmland invertebrates are scarce (Speight, 2001).

While utilising invertebrates in the assessment of farm habitats for developing conservation strategies has been well investigated for some iconic insect groups such as bumblebees and butterflies (Carvell et al., 2007; Pywell et al., 2011), less is known about the use of other invertebrate groups such as Diptera in habitat assessments (Carey et al., 2017a). This is likely due to the greater abundance and diversity of Diptera, and the associated taxonomic challenges, in comparison with other taxa (Barnard, 2011). Including wider and less studied invertebrate groups such as Diptera in the assessment of conservation strategies could help in developing evidence-based measures with strong environmental effectiveness and cost-efficiency to protect and enhance biodiversity on farms. Moreover, Diptera are one of the most abundant animals in temperate habitats (Hughes et al., 2000) with almost 50% of all dipteran families containing flower-visiting flies or pollinators of at least 555 flowering plant species (Larson et al., 2001). Therefore, Diptera are one of the most important groups of pollinators, second only to Hymenoptera, with both having a major contribution to plant diversity and agricultural production (Ssymank et al., 2008; Cook et al., 2020; Rader et al., 2020). In addition to pollination, Diptera have other important ecosystem services such as decomposition (Frouz & Šimek, 2009) and biological control of agricultural pests (Hynes et al., 2014a, b & c; D'Ahmed et al., 2019; Bistline-East et al., 2020).

This study aims to fill current knowledge gaps by exploring the value of different types of linear farm habitats (individually or in combination) to insects, across a gradient of farming intensities, using adult Syrphidae and Sciomyzidae (Diptera). Both taxa co-exist within many of the same habitats but have markedly different ecological requirements. Syrphidae are known as strong flyers (Dover, 2019; Speight, 2020), reflecting landscape scale effects, while Sciomyzidae appear to have limited movement (Williams et al., 2010), and reflect local scale differences. Sciomyzidae have also been shown to reflect dipteran richness and abundances in wet grasslands (Hayes et al., 2015; Carey et al., 2017a, b). In addition, both fly families have important ecosystem services linked to food production in that adult Syrphidae are pollinators (Stanley et al., 2013; Stanley & Stout, 2014; Power et al., 2016; Doyle et al., 2020) while the larval stages of many species act as predators of agricultural pests such as aphids (Schirmel et al., 2018; Speight, 2020). Other Syrphidae larval species contribute to dung breakdown and nutrient cycling (Speight, 2020). On the other hand, Sciomyzidae larvae feed primarily on molluscs, some of which act as intermediate hosts of liver fluke disease, and on pestiferous slug species (Knutson et al., 1965; Knutson & Vala, 2011; Hynes et al., 2014a,b & c; D'Ahmed et al., 2019; Bistline-East et al., 2020). Adults of some Sciomyzidae species can also be minor pollinators (Stoffolano et al., 2015). In addition, adults of both fly families are characterized by their ease of collection, identification, and their ubiquity across a range of habitats (Speight, 1986); thereby fulfilling the criteria for suitable bioindicators as outlined by McGeoch (1998).

With this in mind, the objectives of this study were to:

1- Evaluate different linear farm habitats (separately and in combination) in sustaining Syrphidae and Sciomyzidae, known indicators of dipteran diversity in agricultural grasslands.
2- Ascertain the role of habitat quality and other environmental variables in determining abundance, species richness and assemblages of each taxon.
3- Establish how these dipteran families reflect a farming intensity gradient and address current thinking on the conservation value of farmland hedgerows.

3.3. Materials and methods

3.3.1. Site selection and classification

The study was conducted on farmland in the north-west of Ireland in County Sligo (Geographic Location: 54.1553° N, 8.6065° W; Appendix II.1) as part of a larger project entitled "Farming and Natural Resources: Measures for Ecological Sustainability" or "FARM-ECOS". Mean annual temperature and precipitation in Sligo are 9.6°C and 1260.1 mm respectively (https://www.met.ie/, accessed 08/04/2020). Grassland (including rough grazing) accounts for approximately 99% of the farmed area of this study (www.cso.ie). Grass-based farms dominated by cattle and/or sheep grazing were classified according to land use intensity into extensive, intermediate, and intensive farms. Farm classification was based on a Nature Value (NV) index developed by Boyle, Hayes et al., (2015), which considers the area owned and farmed, the stocking rate, the proportion of improved grasslands and a visual assessment of the size of fields and linear habitats. The NV score (calculated for each farm using the on-line tool (http://www.high-nature-value-farmland.ie/is-your-farm-hnv/) allowed us to classify the farms into one of three farming intensities; extensive (NV index > 5; n = 5); intermediate (NV index between 3.5 and 5; n = 5); and intensive farms (NV index < 3.5; n = 5).

In each of the three farming intensities, five categories of linear farm habitats were selected at field level for comparison as follows: a) dense hedgerow with < 50% cover of gaps (DH); b) open hedgerow with > 50% gap cover (OH); c) dense hedgerow with < 50% cover of gaps immediately adjacent to a watercourse (DH_W); d) open hedgerow with > 50% cover of gaps immediately adjacent to a watercourse (OH_W);

and e) watercourse only (W) (See Appendix II.2 in the Supporting Information). For the purposes of this study, hedgerows were defined as woody components of a linear habitat (often associated with banks, walls, ditches or trees) with a maximum width of 4 m and with shrubs covering at least 25% of the length of a field (Foulkes et al., 2013). Gaps were defined as any area of hedgerow where woody species were absent in addition to spaces composed of brambles, walls, fences, non-structural hedgerow species e.g., climbers, and dead sections of hedgerow (Defra, 2007; Foulkes et al., 2013). Watercourses (ditches/streams) were defined as either channels created by humans (e.g. open drains) or watercourses resulting from natural processes (e.g. streams) (after Williams et al., 2004).

3.3.2. Sample collection and identification

Invertebrate sampling was conducted from May to September 2018 using Townes style bi-directional (or double headed) Malaise traps (Macfadyen & Muller, 2013; Macfadyen et al., 2015; Samaranayake & Costamagna, 2018) protected from livestock by portable electric fences. In each of the five selected linear habitats across the three farming intensities, a pair of Malaise traps (as recommended by Speight et al., (2000) were set up 2 m from the linear habitat (after Wolton et al., (2014). Within a site, each pair of traps were placed 20 m apart (after Carey et al., (2017a), with trap pairs between sites at least 200 m apart after Gittings et al., (2006). This resulted in a total of 30 Malaise traps across farms, each with two collection bottles half filled with 70% ethanol, giving a total of 60 collection bottles. Each trap was positioned parallel to a linear habitat running in an east-west direction, with trap collection heads facing in an easterly direction thereby permitting catches from the linear habitats and open fields to be collected in separate collection heads (trap side – Appendix II.3). Traps were activated on 24 May 2018 and insect samples were collected every two weeks until September 13th (2018) resulting in a total of 8 field visits and 480 samples. Vegetation immediately around the traps but inside the electric fences was cut periodically with a hand shears to maintain similar vegetation heights inside and outside the electric fences (Carey et al., 2017a).

All collected samples were stored in the laboratory for later identification. Samples were sieved through a fine mesh strainer (1 mm), and the remaining insects sorted to order and family level. Species of the families Syrphidae and Sciomyzidae, focal species for this study, were separated and subsequently identified to genus and species level using Ball and Morris (2015) and Stubbs and Falk (2002) for Syrphidae, and Rozkošný (1987) for Sciomyzidae. Sciomyzidae species were also compared with collected reference samples in the laboratory which were previously identified by taxonomic experts. A small number of female syrphids which could not be identified to species level are presented to genus level only (see Appendix II.4 & II.5 in Supplementary Information).

3.3.3. Environmental data

The quality of habitats was assessed using Rapid Assessment Cards (RACs) developed for each habitat type in both fields and linear habitats (Rotchés-Ribalta et al., 2020) and used to rate the ecosystem condition and provide a picture of the conservation status of habitats. Surveys of habitat quality in both grasslands and linear habitats involved the collection of several variables that were identified as indicators of environmental condition (e.g., vegetation structure, vegetation cover, height, shape of the hedgerow), habitat significance (e.g., number and cover of positive/negative plant species indicators) and management pressure (e.g., visual assessment of the level of grazing or poaching pressure) (see Rotchés-Ribalta et al., 2020). Habitat quality surveys in grasslands were conducted while walking a “W” shaped route in fields, as recommended in the RBAPS assessment (Maher et al., 2018b). For linear habitats, the quality surveys were conducted along 30 m length; two surveys were conducted when a linear habitat was > 80 m long (Foulkes et al., 2013). From the RACs, a score of quality was obtained for each habitat, which was scaled between 0 and 1, with 0 being the lowest quality habitat and 1 the highest quality (see Rotchés-Ribalta et al., 2020). The number of flowering plant species (flowering plant species richness) in the linear features and in the grassland was recorded at each site. Soil samples were also collected within each trap location on November 6th (2018) using a standard soil auger (Eijkelkamp) and stored in a cold room (4°C) prior to processing. Soil organic matter, pH and soil moisture content were measured within five days of sampling following British Standards protocols (BSI, 1990).

3.3.4. Data analysis

A total of 420 samples from 7 collections were included for data analysis (excluding 60 samples from 21st of June due to trap damages by Storm Hector). Prior to performing statistical analysis, abundance and species richness data were combined for all the 7 sampling periods (separately for the linear and field side of the traps). Species area curves calculated for Syrphidae and Sciomyzidae showed adequate trapping effort for both species (Appendix II.6).

Univariate analysis (IBM, SPSS Statistics v.24) was undertaken using Generalized Linear Mixed Models (GLMMs) with Poisson distribution and log link function to examine the effects of linear habitat type, farming intensity and trap side on species richness and abundance of Syrphidae and Sciomyzidae. To take account for the non-independence of trap side and trap numbers per site, trap side was nested within the random factor trap ID, and trap numbers nested within site ID in all the models. Post-hoc pairwise comparisons were conducted thereafter to determine the individual effects of linear habitat types and farming intensity on Syrphidae and Sciomyzidae (mean species richness and abundance). Model fitness was validated by analysing and verifying normality of residuals. We excluded the interaction terms between linear habitats and farming intensity in the models (after Volpato *et al.*, 2020) due to the low number of linear habitat types occurring per farming intensity category ($n = 1$). Given that the environmental variables did not follow a normal distribution, we used nonparametric Kruskal-Wallis test followed by Bonferroni's pairwise comparison corrected for multiple ties to compare environmental variables measured across categories of farming intensities and farm linear habitats. In addition, Spearman's rank correlation coefficient was used to determine the correlations between taxa abundance, species richness and environmental variables. All univariate data were analysed at the $P < 0.05$ standard level of significance.

Prior to multivariate analysis, species data were $\log 10(x + 1)$ transformed to reduce the influence of very abundant species (Carey et al., 2017a; Schirmel et al., 2018). Moreover, an outlier analysis was performed in PC-ORD v.6 and no faunistic outliers with > 2.0 standard deviations were detected. Samples (traps) were also examined for extreme outliers with standard deviations > 3.0 using the Sørensen distance measure (after Carey et al., 2017a) and no potential outliers were detected among the samples in each site. Permutation-based Multivariate Analysis of Variance

(PERMANOVA) was utilised to test the effects linear habitat type and farming intensity on the similarity within both taxa communities using PRIMER (v.7.0.13) with the PERMANOVA add-on (Anderson et al., 2008). Trap sides were nested within the random factor trap ID, and trap ID within site ID in the analysis and we used Sørensen as a distance measure with 999 permutations. Non-metric multidimensional scaling (NMS) ordinations (McCune et al., 2002) of samples was undertaken to understand the community structure of both taxa at each farming intensity using the Sørensen distance in PC-ORD v.6 (McCune & Mefford, 2011). The number of significant axes was determined through 250 runs of real data to 250 runs with randomised data. An orthogonal principal axis output was selected for each NMS to illustrate maximum community variation along axis 1. Environmental data were utilised as a second explanatory matrix and variables with Pearson R^2 values > 0.2 overlain as a biplot (McCune & Mefford, 2011). Multi Response Permutation Procedures (MRPP), which are also nonparametric procedures for testing the hypothesis of no difference between two groups, were utilized to test for significant difference between habitat types based on the species composition of each assemblage (McCune & Mefford, 2011).

3.4. Results

3.4.1. General results

A total of 9,047 adult Syrphidae and Sciomyzidae insects were captured during the study (excluding June 21st samples), representing a total of 8,774 individuals of Syrphidae and 273 Sciomyzidae. Seventy-six species of Syrphidae representing 41.3% of all known Irish species and 17 species of Sciomyzidae representing 28.3% of all known Irish species (Chandler et al., 2008; Maher et al., 2018a) were captured. The dominant syrphid species were *Helophilus pendulus* (L.), 1758 (13%), *Platycheirus clypeatus* (Meigen), 1822 (12%), *Platycheirus granditarsus* (Forster), 1771 (11%) and *Eupeodes latifasciatus* (Macquart), 1829 (10%) comprising 46% of the total syrphid catches. The dominant sciomyzid species were *Tetanocera arrogans* (Meigen), 1830 (21%), *Renocera pallida* (Fallén), 1820 (18%), *Tetanocera elata* (Fabricius), 1781

(15%), and *Tetanocera ferruginea* (Fallén), 1820 (13%) comprising 67% of total sciomyzid catches (Appendix II.4 & II.5 in Supplementary Information).

3.4.2. Effect of linear habitats and farming intensity on taxa

Overall Sciomyzidae species richness and abundance across all farming intensities (Fig. 3.1; Table 3.1) were greatest in open hedgerows with adjacent watercourses (OHw). Sciomyzidae species richness and abundance were significantly greater in open hedgerows with adjacent watercourses (OHw) and watercourses only (W) than in dense hedgerows (DH) or open hedgerows (OH). However, there were no significant differences between dense hedgerows with adjacent watercourses (DHw) and dense hedgerows (DH)/open hedgerows (OH) (see Appendix II.7 for *P* values). In contrast, while Syrphidae species richness and abundance were greater in dense hedgerows with adjacent watercourses (DHw), no significant differences across categories of linear habitat types were detected (Fig. 3.1; Table 3.1).

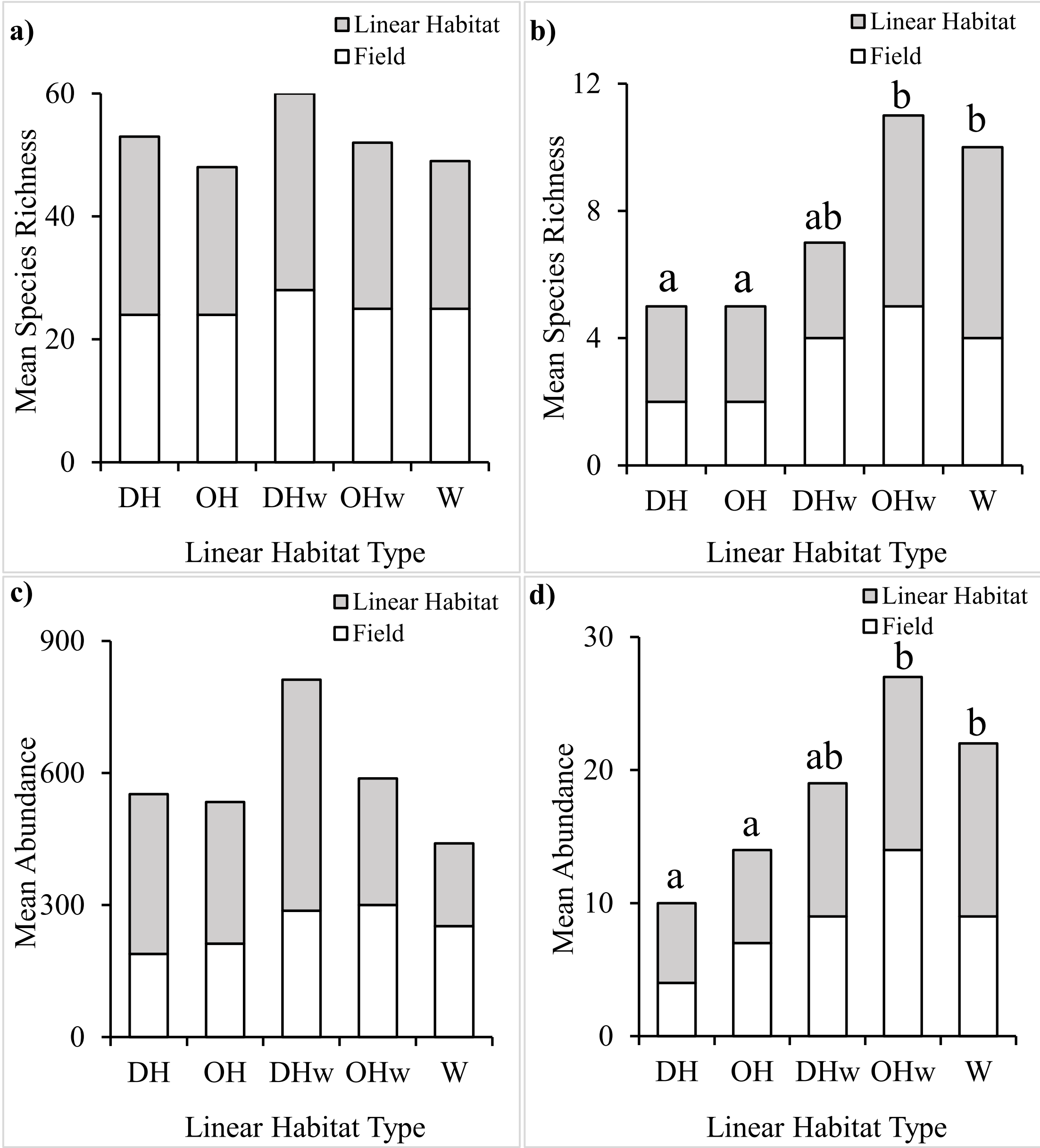

Figure 3.1. Mean species richness and abundance of Syrphidae and Sciomyzidae collected per site at each of the five categories of linear habitat type*: (a) Syrphidae species richness, (b) Sciomyzidae species richness, (c) Syrphidae abundance and (d) Sciomyzidae abundance. Columns annotated with the different letters are significantly different within each separate category (GLMM followed by LSD pairwise comparisons, $P < 0.05$).

* DH = dense hedgerow, OH = open hedgerow, DHw = dense hedgerow + watercourse, OHw = open hedgerow + watercourse and W = watercourse only.

Table 3.1. Overall effects of farming intensity, linear habitat type and trap side on the species richness and abundance of Syrphidae and Sciomyzidae. Bold numbers indicate significant differences (GLMM, $P < 0.05$) *.

	Farming intensity				Linear habitat type				Trap side			
	d.f.1	d.f.2	F	P	d.f.1	d.f.2	F	P	d.f.1	d.f.2	F	P
Richness												
Syrphidae	2	52	10.51	**<0.001**	4	52	2.67	0.07	1	52	1.21	0.28
Sciomyzidae	2	52	13.58	**<0.001**	4	52	3.24	**0.02**	1	52	2.02	0.16
Abundance												
Syrphidae	2	52	1.84	0.17	4	52	1.51	0.21	1	52	2.58	0.12
Sciomyzidae	2	52	25.02	**<0.001**	4	52	4.26	**0.01**	1	52	0.80	0.38

* Due to there being only one replicate of each linear habitat type for each farming intensity (n = 1), it was not possible to include the interaction term in the model. d.f.1 represents the between-groups degrees of freedom, while d.f.2 represents the residual degrees of freedom.

Comparisons of farming intensities (Table 3.1, Fig. 3.2) showed that Syrphidae and Sciomyzidae species richness were significantly greater in extensive than in either intermediate ($P < 0.01$; $P < 0.001$ respectively) or intensive farms ($P < 0.001$; $P < 0.001$ respectively). Syrphidae abundances were not significantly different across farming intensities, but Sciomyzidae abundances (Table 2.1, Fig. 2.2) were significantly greater on extensive than on either intermediate ($P < 0.001$) or intensive ($P < 0.001$) farms (see Appendix II.8 for all the *P* values).

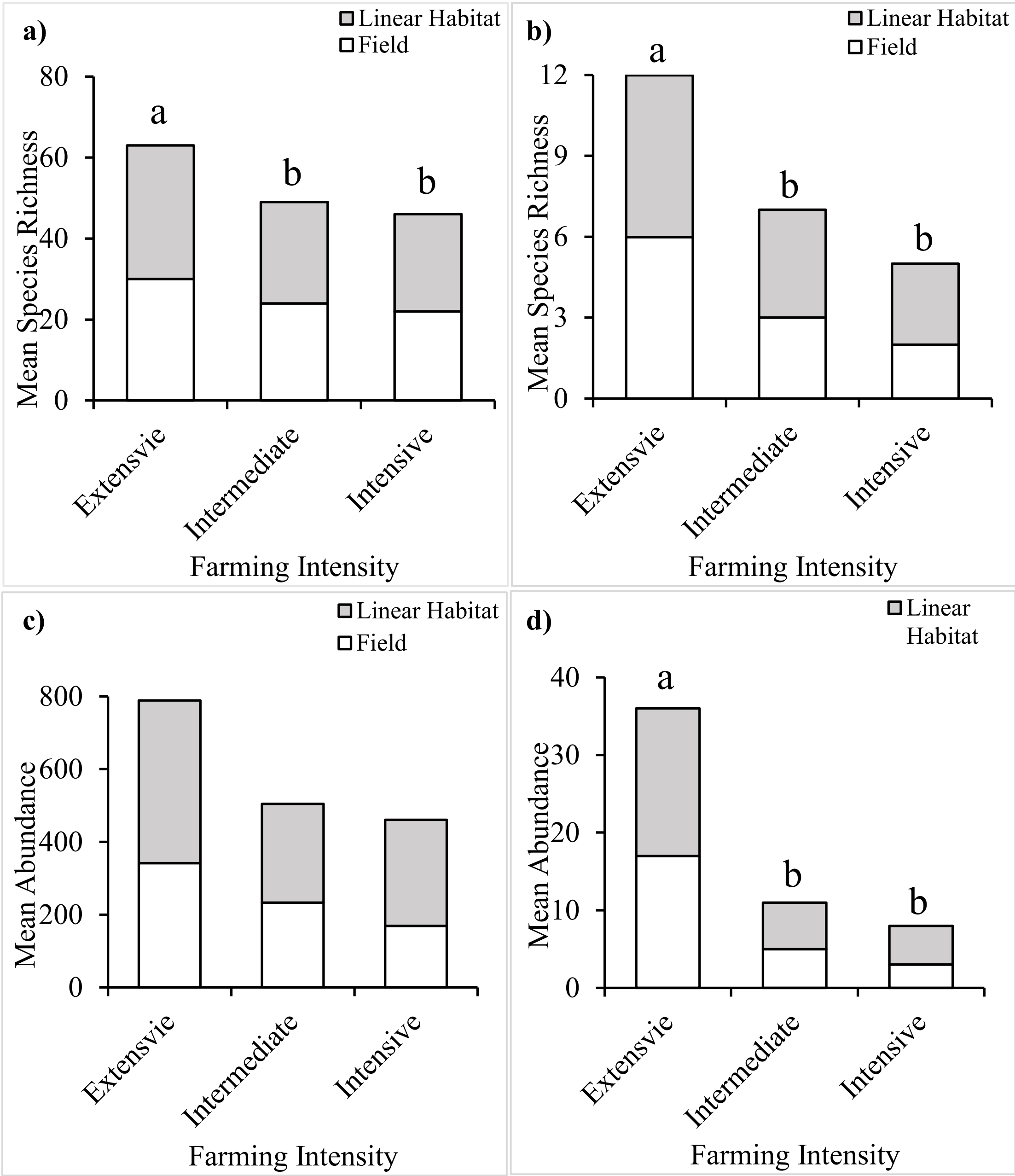

Figure 3.2. Mean species richness and abundance of Syrphidae and Sciomyzidae collected per site at each of the three-farming intensities: (a) Syrphidae species richness, (b) Sciomyzidae species richness, (c) Syrphidae abundance and (d) Sciomyzidae abundance. Columns annotated with the different letters are significantly different within each separate category (GLMM followed by LSD pairwise comparisons, $P < 0.05$; Appendix II.8).

3.4.3. Effect of environmental variables and habitat quality on taxa

The environmental variables measured throughout the study differed across categories of farming intensities. Mean percentage soil moisture was significantly greater in extensive farms in comparison to intermediate ($P = 0.03$) and intensive farms ($P = 0.04$; Table 2.2 & Appendix II.9). Moreover, mean percentage soil organic matter was also significantly greater in extensive than intensive farms ($P = 0.03$; Table 3.2 & Appendix II.9). Of all environmental variables, Syrphidae species richness significantly correlated ($P = 0.04$) with linear habitat flowering plant species richness only (Table 3.3). Sciomyzidae, on the other hand, were correlated with adjacent grassland flowering plant species richness ($P = 0.04$ abundance) and the grassland habitat quality score ($P = 0.03$ species richness; $P < 0.01$ abundance). Structural elements contributing to the grassland habitat score which had significant positive correlations with Sciomyzidae (Table 3.3) included vegetation structure ($P = 0.02$ species richness; $P < 0.001$ abundance), encroaching scrub ($P < 0.01$ abundance) and plant litter ($P = 0.04$ abundance). Sciomyzidae species richness and abundance were also significantly correlated with percentage soil moisture ($P < 0.01$, $P < 0.001$ respectively) and soil organic matter ($P = 0.01$, $P < 0.01$ respectively).

Table 3.2. Environmental variables (median ± IQR) measured throughout the study across categories of farming intensities and farm linear habitats. Linear habitat types are categorised as: Dense hedgerow (DH), Open hedgerow (OH), Dense hedgerow with adjacent watercourse (DH_W), Open hedgerow with adjacent watercourse (OH_W) and watercourse only (W). Different letters indicate significant differences ($P < 0.05$) between each category using the Kruskal-Wallis test followed by Bonferroni's pairwise comparison corrected for multiple ties (see Appendix II.9 for P values).

		Linear habitat score	Grassland habitat score	Flowering plant species richness/linear habitat	Flowering plant species richness/grassland habitat	% Soil moisture	% Soil organic matter	Soil pH*
		Median ± IQR	Median ± IQR	Median ± IQR	Median ± IQR	Median ± IQR	Median ± IQR	Median ± IQR
Farming intensity	Extensive (n = 5)	0.36 ± 0.25	0.68 ± 0.13	11.00 ± 13.00	23.00 ± 18.00	64.78 ± 21.42^{a}	18.82 ± 23.58^{a}	5.20 ± 1.03
	Intermediate (n = 5)	0.62 ± 0.23	0.40 ± 0.65	12.00 ± 6.00	10.00 ± 15.00	42.47 ± 17.51^{b}	13.78 ± 8.49^{ab}	5.96 ± 0.70
	Intensive (n = 5)	0.48 ± 0.26	0.35 ± 0.53	10.00 ± 1.00	8.00 ± 11.00	33.35 ± 14.61^{b}	9.50 ± 5.02^{b}	5.79 ± 0.16
Linear habitat type	DH (n = 3)	0.43 ± 0.06	0.40 ± 0.28	12.00 ± 1.50	10.00 ± 2.50	28.08 ± 21.16	7.34 ± 4.11	5.54 ± 0.49
	DH_W (n = 3)	0.62 ± 0.08	0.60 ± 0.15	10.00 ± 1.00	10.00 ± 4.00	42.47 ± 11.97	15.75 ± 4.57	7.68 ± 0.82
	OH (n = 3)	0.51 ± 0.17	0.35 ± 0.23	9.00 ± 12.00	17.00 ± 6.50	43.33 ± 13.57	15.11 ± 10.93	5.86 ± 0.40
	OH_W (n = 3)	0.39 ± 0.07	0.00 ± 0.33	11.00 ± 3.00	13.00 ± 11.00	33.35 ± 8.75	9.50 ± 5.04	5.88 ± 0.63
	W (n = 3)	0.32 ± 0.31	0.68 ± 0.18	10.00 ± 1.50	29.00 ± 14.00	52.66 ± 16.08	14.75 ± 15.70	5.68 ± 0.69

* The test on soil pH was done on hydrogen ion concentrations.

Table 3.3. Spearman's rank correlation coefficient between Syrphidae/Sciomyzidae species richness/abundance and environmental variables. Numbers in bold indicate significant correlations (P < 0.05). Variables for the grassland habitat quality score subcategory (structural condition) are presented in italics.

	Syrphidae				Sciomyzidae			
	Richness		Abundance		Richness		Abundance	
	Corr. Coef.	P	Corr. Coef.	P	Corr. Coef.	P	Corr. Coef.	P
Flowering plant species richness/linear habitat	0.53	**0.04**	0.42	0.12	0.06	0.82	0.25	0.36
Flowering plant species richness/grassland habitat	0.06	0.83	0.01	0.96	0.38	0.16	0.54	**0.04**
Linear habitat score	0.04	0.89	0.07	0.81	-0.14	0.62	-0.26	0.35
Grassland habitat score	0.39	0.15	0.15	0.59	0.56	**0.03**	0.78	**<0.01**
Vegetation structure	0.47	0.08	0.16	0.57	0.06	**0.02**	0.85	**<0.001**
Cover of ground flora	-0.21	0.46	-0.25	0.36	0.04	0.89	-0.26	-0.35
% Encroaching scrub	0.41	0.13	0.26	0.34	0.42	0.12	0.73	**<0.01**
Plant litter	0.33	0.23	0.19	0.50	0.43	0.11	0.52	**0.04**
% Soil moisture	0.34	0.19	0.24	0.40	0.70	**<0.01**	0.86	**<0.001**
% Soil Organic matter	0.39	0.16	0.23	0.42	0.71	**0.01**	0.74	**<0.01**
Soil pH	0.26	0.34	0.26	0.35	0.03	0.93	-0.06	0.83

3.4.4. Community Analysis

Permutation-based Multivariate Analysis of Variance (PERMANOVA) showed that farming intensity and linear habitat types had a significant effect on the similarity of both taxa communities (Table 3.4). NMS ordination biplots (Figs. 3.3a & b) show a two-dimensional solution for both taxa with stress values less than 11, where values of about 10 are known to indicate a good ordination with little chance of false inferences (McCune & Mefford, 2011). Environmental variables with Pearson R^2 values of > 0.2 are shown as biplots (Fig. 2.3). MRPP analysis showed significant differences in community structure for both taxa in relation to farming intensity. Farming intensity was a significant grouping variable in both the Syrphidae and Sciomyzidae species matrices (A = 0.08, P = 0.008; A = 0.07, P = 0.046 respectively). In addition, the community composition of Syrphidae was positively (R^2 > 0.02) correlated with the grassland habitat score, percentage soil moisture and percentage soil organic matter in extensive farms while the species composition of intermediate and intensive farms was similar with some degree of overlap (Fig. 3.3a). For Sciomyzidae communities, however, there was some overlap between all three farm types but with positive correlations (R^2 > 0.02) with grassland habitat score (Fig. 3.3b).

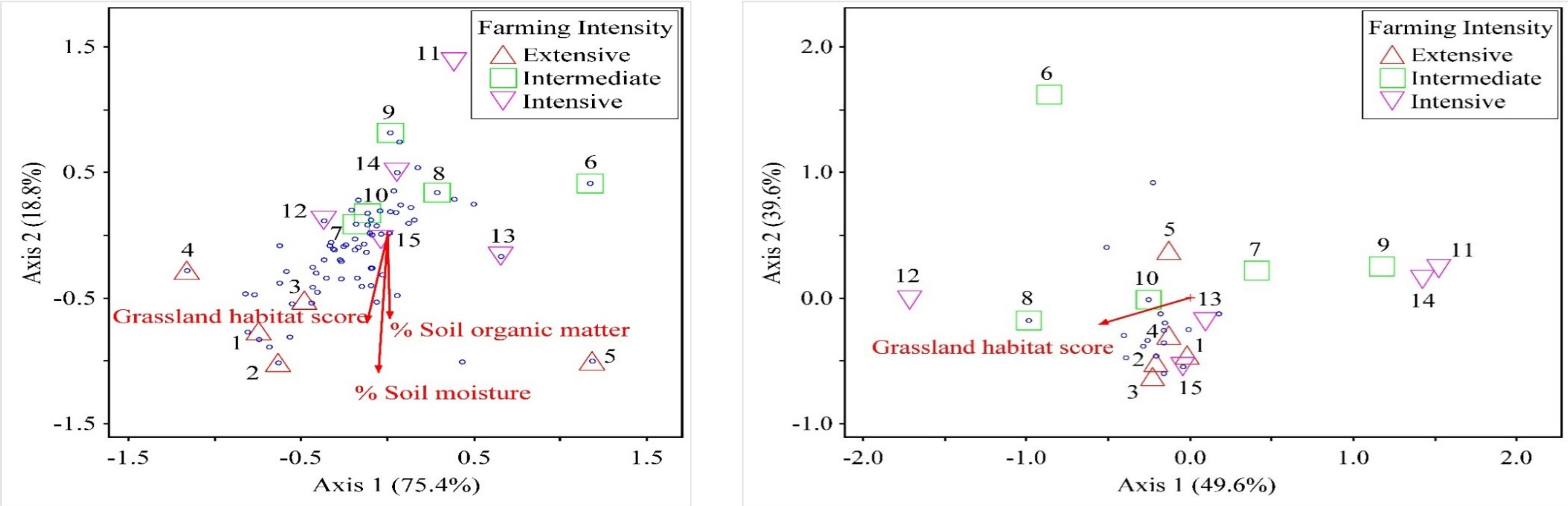

Figure 3.3. Non-metric multi-dimensional scaling ordination of (a) Syrphidae and (b) Sciomyzidae samples collected on Irish farmland. For Syrphidae: first two axes explain 94.2% of the variation (75.4% axis 1 and 18.8% axis 2) with an orthogonality of 100%. Farming intensity is a significant grouping variable (P = 0.008) and explains approximately 8% of the variation in the species matrix (MRPP chance-corrected within-group agreement A). For Sciomyzidae: first two axes explain 89.2% of the variation (49.6% axis 1 and 39.6% axis 2) with an orthogonality of 100%. Farming intensity is a significant grouping variable (P = 0.046) and explains approximately 6.7% of the variation in the species matrix (MRPP chance-corrected within-group agreement A).

3.5. Discussion

While the incorporation of linear habitats to counteract biodiversity decline on farmland has already been proposed (Tattersall et al., 2002; Brooks et al., 2012; Garratt et al., 2017; Schirmel et al., 2018), much remains unknown about its impact on specific insect species and communities, particularly on livestock-based grassland systems of different intensities. Moreover, the conservation of invertebrate diversity in agricultural lands requires that invertebrate indicators (particularly non-iconic groups which generally receive less attention) be incorporated in assessment methodologies at field and farm level to understand and predict biodiversity (Plantureux et al., 2005). This study was designed to examine the response of adult Syrphidae and Sciomyzidae, indicator species with different ecological requirements and ecosystem functions, to different linear farm habitats and to ascertain whether levels of farming intensity classified primarily on the basis of physical features and farming practices also reflect these insect indicator species.

3.5.1 Effect of linear habitats on taxa

The results of this study demonstrate that different types of linear habitats contribute differently to selected dipteran abundance and diversity in agricultural grassland. Both taxa demonstrated different responses to linear habitat types with mean Sciomyzidae species richness being significantly greater at open hedgerows with an adjacent watercourse than dense hedgerows or open hedgerows only. In contrast, there was no significant difference between dense hedgerows with an adjacent watercourse and dense/open hedgerows only. This finding is particularly important in the context of current advice on best practice for hedgerow maintenance, i.e., keeping the shrub layer dense ((Hedgelink leaflet (2013) - www.hedgelink.org.uk)) or in hedgerows being assessed as less favourable on the basis of increased gappiness (Foulkes et al., 2013). The majority of Sciomyzidae species found in this study (> 70 %) have semi-aquatic or aquatic larvae (Williams et al., 2007; McDonnell et al., 2010; Knutson & Vala, 2011) which may explain their higher species richness in open hedgerows adjacent to watercourses. Moreover, Sciomyzidae are relatively sedentary (Williams et al., 2010), and thus it is possible that dense hedgerows could inhibit their movements across habitats as has been suggested for other weak flying insects, particularly parasitoids (Dover, 2019) although this would need to be substantiated for Sciomyzidae using suitable mark-recapture methods (Williams et al., 2010).

While Sciomyzidae (species richness and abundance) showed no significant correlations with linear habitat quality in this study, significant correlations were detected with overall grassland habitat quality and good vegetation structure in adjacent grassland fields (i.e., >50% of the field having a heterogeneous vegetation structure). This is supported by previous studies where Sciomyzidae (as well as other dipteran families of grassland – Ryder et al., 2005) have been shown to demonstrate positive correlations with vegetation structure, particularly taller vegetation (Ryder et al., 2005; Williams et al., 2009a, b; Maher et al., 2014). In this study, heterogeneous vegetation structure is likely to be a result of the less intensively managed, wetter fields carrying lower stocking densities than the more improved fields with drier soils. In addition, the positive correlations with longer flooding periods of many Sciomyzidae species (Williams et al., 2009b; Maher et al., 2014) which feed on aquatic/semi-aquatic snails during the larval stage, further substantiates the need for wetter conditions (including adjacent watercourses) for many species of this family. The significant correlation of Sciomyzidae abundance with plant litter probably reflects the greater litter depths commonly found in wetter, seasonally flooded grasslands. In addition, the correlation of Sciomyzidae abundance with scrub encroachment likely reflects similar conditions to those of open hedgerows, i.e., providing some shelter but with gaps for ease of movement. Since dense hedgerows adjacent to watercourses in this study do not have significantly greater Sciomyzidae species richness/abundances than dense/open hedgerows, while open hedgerows with adjacent watercourses/watercourses only do, further work is required to fine tune the advice currently given to landowners on the maintenance of hedgerows, particularly those adjacent to water bodies.

In contrast to Sciomyzidae, the species richness and abundance of Syrphidae captured were greater (although not significantly) in dense hedgerows adjacent to watercourses than in other linear habitat types. Dense continuous hedgerows, which are diverse in plant species and structure, have been shown to provide valuable resources to Syrphidae as for other strong flying insects such as bumblebees (Garratt et al., 2017). They are unlikely to inhibit Syrphidae movement across habitats since Syrphidae are capable of long-distance migrations (Dover, 2019; Doyle et al., 2020). This may explain why Syrphidae species richness and abundance showed no significant differences between linear habitat types including dense hedgerows. However, it is noteworthy that dense hedgerows adjacent to watercourses are likely to provide multiple resources for Syrphidae, particularly standing water in addition to dead wood, litter, sap runs, host plants and damp holes important for larval development (Schirmel et al., 2018). This may explain greater (albeit non-significant) Syrphidae species richness and abundance in dense hedgerows adjacent to watercourses. In addition, the proportion of Syrphidae captured on the linear habitat side on intensive farms (27% greater than on the field side) was comparatively larger than that on intermediate and extensive farms (<15%), indicating the likely importance of linear habitats on intensive farms where fewer nectar resources would be available in the adjacent intensive grasslands. Linear habitats would also play an important role by providing shelter (Sutherland et al., 2001), overwintering sites (Hondelmann & Poehling, 2007) and protection from agrochemical applications (Schirmel et al., 2018), particularly on intensive farms.

3.5.2. Effect of farming intensity on taxa

Species richness of both taxa showed a significant decline with increasing farming intensity; suggesting that farming intensification is a primary driver in reducing species richness of both families. Syrphidae are known to be positively influenced by pollen and nectar as food sources (Ricarte et al., 2011) and this is likely reflected by greater (albeit non-significant) flowering plant species richness in extensive farm grasslands. Sciomyzidae, on the other hand, are more likely to be influenced by the vegetation structure (i.e., taller plants) and wetter soils, found on the extensive farms (Ryder et al., 2005; Williams et al., 2009a, b; Maher et al., 2014). A similar trend was observed in terms of the number of individuals captured for both families, but only Sciomyzidae abundance showed significant decline in abundance with increasing farming intensity. More than 70% of total Sciomyzidae species found in this study are hygrophilous in their larval stages, feeding on either freshwater snails at or below water surface and/or semi-terrestrial snails, or on fingernail clams and pea mussels beneath the water surface (Williams et al., 2007; McDonnell et al., 2010; Knutson & Vala, 2011). This, coupled with the limited distances (up to 25 m) adult Sciomyzidae may travel (Williams et al., 2010), is likely to reflect their overall preferences, at a local scale, for grassland fields with good habitat quality (particularly good structural condition and low management pressure) and wetter, more organic soils associated with the grasslands of more extensive farms. Many Syrphidae, on the other hand, are strong flyers (Dover, 2019; Speight, 2020) and polylectic as adults (Speight, 2020) visiting flowers in a wide range of habitats that can be far from their breeding sites (Ball & Morris, 2015; Speight, 2020). The effect of floral resources on hoverfly richness/abundance can also depend on management e.g., organic versus conventional farming (Power et al., 2016) and on landscape factors (Meyer et al., 2009; Power et al., 2016). This may explain why Syrphidae abundance showed no significant response to farming intensity and associated environmental variables at a local scale.

3.5.3. Community analysis

In addition to abundance/species richness, farming intensity also showed a significant effect on the similarity of both fly family communities as explained by MRPP analysis. Moreover, NMS analysis shows that extensive farms are characterised by specific environmental conditions that are likely to play important roles in shaping the community composition of each family. NMS analysis indicates that grassland quality score, % soil moisture and % organic matter are important environmental variables playing a role in shaping Syrphidae species assemblages. This is in line with previous studies that showed intensively managed fields with poor-quality habitats are unlikely to provide valuable resources to sustain insect pollinators (Cole et al., 2020) including Syrphidae (Schweiger et al., 2007). In addition, other studies have also demonstrated that drainage along with high stocking rates and fertilizer inputs in intensive farms result in reduced soil moisture and organic matter (Plantureux et al., 2005) with intensive grazing causing habitat loss through the removal of ground vegetation and organic matter as well as soil compaction (Yadamsuren et al., 2015). Practices such as these coupled with greater levels of agrochemical inputs in intensive farms have also been shown to limit resource availability for many invertebrates

(McMahon et al., 2012) and reduce plant and invertebrate species richness in general (Zechmeister et al., 2003; Klimek et al., 2007). On the other hand, Sciomyzidae communities were positively correlated with the grassland habitat score. Moreover, there were overlaps in Sciomyzidae species assemblages between all farm categories that can be explained by some extensive fields being located within a farm classified overall as intensive or intermediate. This demonstrates that the retention of extensive or wet grassland fields even within intensive farms can provide valuable habitats to sustain Sciomyzidae assemblages at small spatial scales. This agrees with the study by Carey et al., (2017a) which demonstrated that Sciomyzidae communities in grassland habitats can vary at small scales of up to 20 m.

3.5.4. Management implications

Overall, Syrphidae and Sciomyzidae species richness and community composition are shown to reflect a farming intensity gradient with significantly greater species richness for both families on extensive farms.This reinforces the importance of extensive farms for biodiversity conservation in general and is particularly important for dipteran conservation on farmland. Nevertheless, since the farm classification used in this study considers not only farm management but also a visual assessment of the size of fields and linear habitats, careful considerations should also be taken at smaller scales since different linear habitats within fields/farms seem to contribute differently to dipteran abundance and diversity. While it is known that dense continuous hedgerows are generally considered as good quality hedgerows with valuable resources for insect pollinators (Garratt et al., 2017; Volpato et al., 2020), other, less mobile aerial invertebrates with important ecosystem functions, appear to have different requirements. Hence, hedgerows, irrespective of perceived quality, and particularly those deemed 'gappy' adjacent to watercourses, appear, in this study, to be of value to biodiversity. This is particularly important in the context of current agricultural Environmental Impact Assessment (EIA) regulations in Ireland which allow for up to 500 m of boundary to be removed without assessment ((European Communities (Environmental Impact Assessment) (Agriculture) Regulations, 2011)). Under current regulations, therefore, hedgerows with significant value to biodiversity are likely to be lost if such regulations are not improved to protect these valuable habitats. Discussions, based on the scientific evidence of multiple studies, regarding advice to farmers in Ireland under the current Agri-Environment Scheme ((Green, Low-carbon Agri-Environment Scheme (GLAS)) to maintain dense hedgerows, will be required to inform future schemes under the new EU common agricultural policy (2021-2027) to facilitate those less mobile species (including those with conservation value) adversely affected through habitat loss and resource decline (Graham et al., 2018). It is likely that consideration to supporting a mixture of both open and dense hedgerows adjacent to watercourses is required (linear habitat heterogeneity within and between habitats), with particular attention given to spatial scales and management heterogeneity over both time and space (Graham et al., 2018).

3.6. Conclusions

Our results indicate that linear habitats irrespective of perceived quality, particularly those hedgerows deemed 'gappy' adjacent to watercourses, are of significant value to biodiversity. This could have important implications for future design and implementation of agri-environment schemes by considering the heterogeneity of linear habitats (i.e., not only dense hedgerows but also a diverse range of boundary types) across different farming intensities. In addition, our results show that farmland intensity is an important driver of overall pattern and community composition of both dipteran families investigated in this study. Nevertheless, enhancing habitat quality within and between farms appears to be a key message for conservation of dipteran diversity in farmland and in supporting their ecosystem functions. Thus, future agri-enviroment schemes should also incentivise low intensity farming since it is likely to generate favourable conditions to promote habitat quality and subsequently support invertebrate diversity in agricultural lands.

3.7. Acknowledgements

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Chapter 4 An investigation of how local and landscape variables can be used to incorporate hoverfly functional groups in rapid farmland biodiversity assessments

Karzan S. D. Ahmed[1,2], Sara Ruas[3], Stephanie Maher[4], Alessio Volpato[3], Christopher Williams[5], Daire Ó hUallacháin[6], James Moran[3], Simon Hodge[7], Jane C. Stout[3], Blanaid White[8], Michael J. Gormally[1,2]

[1]Applied Ecology Unit, School of Natural Sciences, National University of Ireland Galway, Ireland; [2]Ryan Institute, National University of Ireland Galway, Ireland; [3]Department of Natural Sciences, Galway Mayo Institute of Technology, Galway, Ireland; [4]School of Natural Sciences, Trinity College Dublin, Dublin, Ireland; [5]School of Biological and Environmental Sciences, Liverpool John Moores University, UK; [6]Teagasc Environment, Soils and Land use Department, Johnstown Castle, Co Wexford, Ireland; [7]School of Agriculture and Food Sciences, University College Dublin, Dublin, Ireland; [8]School of Chemical Sciences, Dublin City University, Dublin, Ireland.

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4.1. Abstract

While farm biodiversity is frequently assessed in agri-environmental payment schemes on the basis of habitat type, condition, and/or plant indicator species, less attention is generally given to invertebrates. In this study, we used hoverflies (Diptera: Syrphidae) and their associated functional groups to assess the value of different farm habitats to an insect family that delivers multiple ecosystem services. The aim of the study was to determine those habitat features that best reflect hoverfly requirements and their associated ecosystem functions at local and landscape scales with a view to incorporating them in rapid assessments as part of future agri-environmental payment schemes. The study was conducted in two regions of Ireland (Co. Sligo and Co. Wexford) with contrasting topographic, climatic, and farming conditions where farms in Co. Sligo and Co. Wexford consist of predominantly livestock farms and tillage/mixed farms, respectively. Within each study region, fifteen farms across a gradient of farming intensity (extensive, intermediate, and intensive) were selected and pan traps were employed to capture hoverflies from 15th of May to 16th of August 2019. In addition, a range of environmental variables were collected using a combination of field surveys and satellite imagery. At local scale, field habitat quality score was the most important driving factor positively affecting abundance and species richness of non-predatory hoverflies (both counties combined) as well as abundance of predatory hoverfly species (Co. Wexford). At landscape scale, hedgerows, and tree lines (irrespective of county and farming conditions) were the most important driving factors positively affecting hoverfly abundance/richness, with other linear habitat features showing variables results within and between each study region. Hoverfly community composition and the driving factors varied significantly between the two counties where in Co. Wexford predatory species such as *Episyrphus balteatus* dominated and hoverfly communities were driven by altitude and grassy margins. However, in Co. Sligo where non-predatory species dominated, hoverfly communities were driven by field habitat quality score, drainage ditches and linear feature diversity. In relation to management practices, our results indicate that while intensive farming can benefit selected hoverfly species (such as generalist, predatory species *E. balteatus*), these systems result in lower total hoverfly abundance and diversity. With regard to agri-enviromental payment schemes, our results show that measuring total length of hedgerows, and treelines through satellite imagery are the best explanatory variables at landscape scale that can be incorporated in rapid assessments for national agri-environmental payment schemes since they positively affect hoverflies abundance and richness. While total length of other variables such as stonewalls and drainage ditches showed contrasting results between the two counties, they are unlikely to be beneficial for national agri-environmental payment schemes since their presence does not consistently support hoverfly abundance, richness and their communities. Further research, based on field surveys, is required to assess the value of these habitat features for hoverflies across different farming intensities and study regions.

Keywords: biodiversity, farmland habitats, agri-environmental schemes, hoverflies, functional groups, pan traps.

4.2. Introduction

Farmland biodiversity has undergone significant decline, particularly since World War II (Robinson & Sutherland, 2002), as a result of changes in the management of agricultural land including intensification, specialisation and abandonment (IPBES, 2019; Pardo et al., 2020). Given that farmland biodiversity can play a pivotal role in providing multiple ecosystem services including biological pest control (Thies et al., 2011; Pywell et al., 2015), pollination (Carrié et al., 2012; Cole et al., 2020), and nutrient cycling (Sünnemann et al., 2021), developing new approaches to protecting and enhancing farmland biodiversity is of paramount importance in the context of developing sustainable agricultural systems.

Semi-natural habitats (hereafter SNHs) on farmland including hedgerows, semi-natural grasslands, drainage ditches and dry-stone walls, are important habitat features which support farmland biodiversity (Marshall & Moonen, 2002; Herzon & Helenius, 2008; Manenti, 2014; Schirmel et al., 2018). SNHs provide valuable food sources for both aquatic and terrestrial invertebrates, particularly in intensive agricultural lands where food sources can be scarce or lacking (Herzon & Helenius, 2008; Cole et al., 2017; Garratt et al., 2017; Dover, 2019; Larkin et al., 2019). These habitats are also vital to functionally important invertebrate taxa such as beneficial predators and saprotrophic species which deliver ecosystem services at different spatial scales (Garratt et al., 2017; Moquet et al., 2018; Schirmel et al., 2018; D.Ahmed et al., 2020). Thus, farmlands with a range of natural or semi-natural habitat types (i.e., habitat quantity), coupled with low intensity management, are generally perceived as “biodiverse farmlands” in Europe (Andersen et al., 2003; Anderson & Mammides, 2020). Despite the value of biodiverse farmlands, they are still highly threatened (Plieninger & Bieling, 2013) by processes including agricultural intensification (e.g., increased application of agrochemicals), removal of SNHs or land abandonment (Anderson & Mammides, 2020; Pardo et al., 2020).

The relationship between habitat quality and quantity is a complex one (Vickery et al., 2004) and it has been suggested that improving habitat quality at local scale [measured as “plant species richness or the abundance of plants needed for food and reproduction” - see Kleijn and Van Langevelde (2006)] is a more promising conservation strategy than simply creating new habitat patches (i.e., habitat quantity)

(Jeltsch et al., 2011). A recent study by Rotchés-Ribalta et al. (2020), based on plant species alone, indicates that retention of semi-natural habitats on farms (i.e., habitat quantity) does not necessarily guarantee greater quality of habitats. The implications of this for invertebrates are even less well known, but it is likely that small habitat patches of high habitat quality, depending on how quality is defined, could provide better resources for invertebrates than larger habitat patches of poor habitat quality (Haenke et al., 2009; Meyer et al., 2009).

Habitat selection by many dipteran species is determined, in part, by a wide diversity of feeding and breeding habits among phytophagous, saprophagous, predaceous, and parasitic groups in both adult and larval stages. Therefore, dipteran communities will be influenced by habitat modification and accompanying changes in plant composition and substrate quality (Taillefer & Wheeler, 2010); making them suitable candidates for habitat evaluation in agroecosystems (Williams et al., 2009; Maher et al., 2014; Carey et al., 2017a; Carey et al., 2017b). Among Diptera, hoverflies also known as syrphid flies (family Syrphidae) are one of the most important and diverse groups of true flies, with more than 930 known species across Europe and 185 species in Ireland (Speight, 2020). They have important ecosystem functions, in that adults of many hoverfly species act as crop pollinators (Stanley et al., 2013; Power et al., 2016; Moquet et al., 2018; Doyle et al., 2020; Speight, 2020) while larval stages of many species act as predators of crops pests. Indeed, hoverflies provide extensive pest control services where seasonal migrations of nearly 4 billion individuals between Britain/Ireland and continental Europe are known to consume approximately 6 trillion aphids (Wotton et al., 2019). Other hoverfly species contribute to dung breakdown and nutrient cycling (Moquet et al., 2018; Schirmel et al., 2018; Speight, 2020). Nevertheless, despite their importance, dipteran species in general, including hoverflies, receive less attention (Rader et al., 2016) due to their high abundance and diversity in comparison with other better-known taxa (Carey et al., 2017a).

Recent studies indicate that the favourable effects of habitat type and availability of food sources can vary among hoverfly functional groups, based on larval feeding habits (i.e., predacious vs non-predacious species) (Moquet et al., 2018; Schirmel et al., 2018; Gaytán et al., 2020). Hoverflies, therefore, require complementary resources for both larval and adult stages and their response to these resources vary at both local and landscape scales (Moquet et al., 2018). At local scale, accumulating evidence shows that abundance/richness of hoverflies is strongly related to high density of floral resources (i.e., high habitat quality) that provides pollen and nectar, thereby reducing hoverfly foraging time (Kleijn & Van Langevelde, 2006; Haenke et al., 2009; Meyer et al., 2009; Power et al., 2016; Garratt et al., 2017; Moquet et al., 2018). However, in a recent study of 35 agricultural landscapes in Switzerland and Germany, Schirmel et al. (2018) found that the abundance and richness of floral resources are poor predictors of hoverfly abundance, richness and community composition at a local scale. Thus, defining habitat quality based on floral resources alone may not fully reflect the needs of hoverflies. Evaluating the effects of habitat quality on hoverfly abundance/richness and their communities should, therefore, consider the effect of other quality measures such as those assessed by Rotchés-Ribalta et al. (2020) which includes not only floral resources but also other local structural and ecological elements important to habitat quality (see Rotchés-Ribalta et al., 2020).

Many hoverfly species are known as strong fliers (Dover, 2019; Speight, 2020) with their abundance/richness and communities being influenced by landscape context (i.e., %SHNs or the density of linear landscape features) (Kleijn & Van Langevelde, 2006; Power et al., 2016; Garratt et al., 2017; Schirmel et al., 2018). Landscape complexity can, therefore, promote hoverfly abundance/species richness and influence community composition by providing multiple resources for hoverflies (e.g., food and resting sites) which spill over into neighbouring fields (Kleijn & Van Langevelde, 2006; Schirmel et al., 2018; Rodríguez-Gasol et al., 2020). While the value of landscape context is relatively well known for hoverfly abundance, richness and their community composition, less is understood about the value of specific features at landscape scale (Schirmel et al., 2018) including non-woody linear features such as watercourses, stone walls and grassy margins. Thus, to fill the existing gaps in knowledge, studies at local scale should utilize quality metrics that are based not only on floral resources but also on other quality indicators for both linear and non-linear features (Rotchés-Ribalta et al., 2020) and determine how they reflect hoverfly requirements. In addition, it is crucial to understand how specific features at landscape scale reflect hoverfly requirements, and subsequently develop agri-environment schemes that reflect both plants and hoverflies with important ecosystem services.

This study examined how farming intensity and a range of environmental variables at local and landscape scale affect hoverfly communities and their associated functional groups. We follow this by determining which of these variables, best reflecting hoverfly requirements, could be incorporated in rapid assessments for agri-environmental payment schemes. In addition to their ecosystem function, we selected hoverflies as a focal group in this study since they are good indicators of other dipteran groups (Carey et al., 2017a) and they satisfy the important criteria used for the selection of invertebrates in nature conservation research as proposed by Speight (1986) in terms of ease of collection, identification and their ubiquity across a range of habitats. In this paper, we focus specifically on hoverfly abundance, species richness and larval feeding habits while the pollination services of hoverflies and bees combined will be published elsewhere (Maher et al., in prep).

With this in mind, this study aims to:

1- Determine which farmland habitats and other environmental variables at local and landscape scales best reflect hoverfly requirements and their associated functional groups.
2- Establish how farming intensity affects hoverflies and their functional groups.
3- Assess the impact of the above parameters on hoverfly assemblages.

The implications of this study are important in the context of developing rapid score cards used to highlight those farmland habitats of importance to hoverflies and their functional groups. Moreover, our study has implications to inform successful agri-environmental measures aimed at protecting and enhancing functional diversity on Irish farmland.

4.3. Methods

4.3.1. Study regions and site selection

The study was conducted on agricultural land within two representative study regions in Ireland: County Sligo in the north-west (54.1553° N, 8.6065° W) and County Wexford in the south-east (52.3369° N, 6.4633° W) (Fig. 3.1). These two regions were selected because of their contrasting climatic, topographic, and farming conditions (Rotchés-Ribalta et al., 2020). Grassland accounts for approximately 99% of the farmed area in the west of Ireland in which Co. Sligo is situated (www.cso.ie, accessed 18/01/2021). Cattle and sheep grazing is the dominant land-use (www.cso.ie, accessed 18/01/2021) with the lowlands more intensively managed than the uplands. Mean annual temperature and precipitation in Co. Sligo are 9.6 °C and 1260.1 mm (https://www.met.ie/, accessed 23/03/2020).

The study in Co. Wexford (generally representing more intensive farming than Co. Sligo; Rotchés-Ribalta et al. 2020) was conducted in the Blackstairs Mountains region, where farming systems range from extensive grassland in the foothills to intensive tillage systems in the southern half of the county. Grassland (including rough grazing) accounts for 82% of the farmed area in the south-east of Ireland in which Co. Wexford is situated (www.cso.ie, accessed 18/01/2021). The average annual temperature and precipitation in the Wexford region are 9.8 °C and 840.2 mm (https://www.met.ie/, accessed 23/03/2020).

Fifteen farms in each county were selected (30 in total) reflecting a range of farming intensities categorised by a Nature Value (NV) index using the on-line tool "Is your farm HNV" (Rotchés-Ribalta et al., 2020). Five each of extensive, intermediate, and intensive farms per study region were selected with NV indices of greater than 5, 3.5-5 and less than 3.5 respectively (Rotchés-Ribalta et al., 2020). All selected farms in Co. Sligo were livestock farms (e.g., sheep/cattle) whereas the Co. Wexford study sites included both livestock and tillage farms (see Appendix III.1 in the Appendices). Within each selected farm, detailed habitat quality surveys were undertaken in a representative parcel which consisted of 2 – 5 adjoining fields including the linear features of the parcel (internal and perimeter). Habitat quality surveys were by means of rapid assessment scorecards (see Rotchés-Ribalta et al., 2020). Each habitat (both fields and linear features) within the parcel was then assigned a weighted score from 0 (poorest quality) to 1 (best quality) after Rotchés-Ribalta et al. (2020). Field sizes in each parcel ranged from 0.5 ha – to 1.85 ha in Co. Sligo and from 0.7 ha to 3.9 ha in Co. Wexford (Appendix III.1).

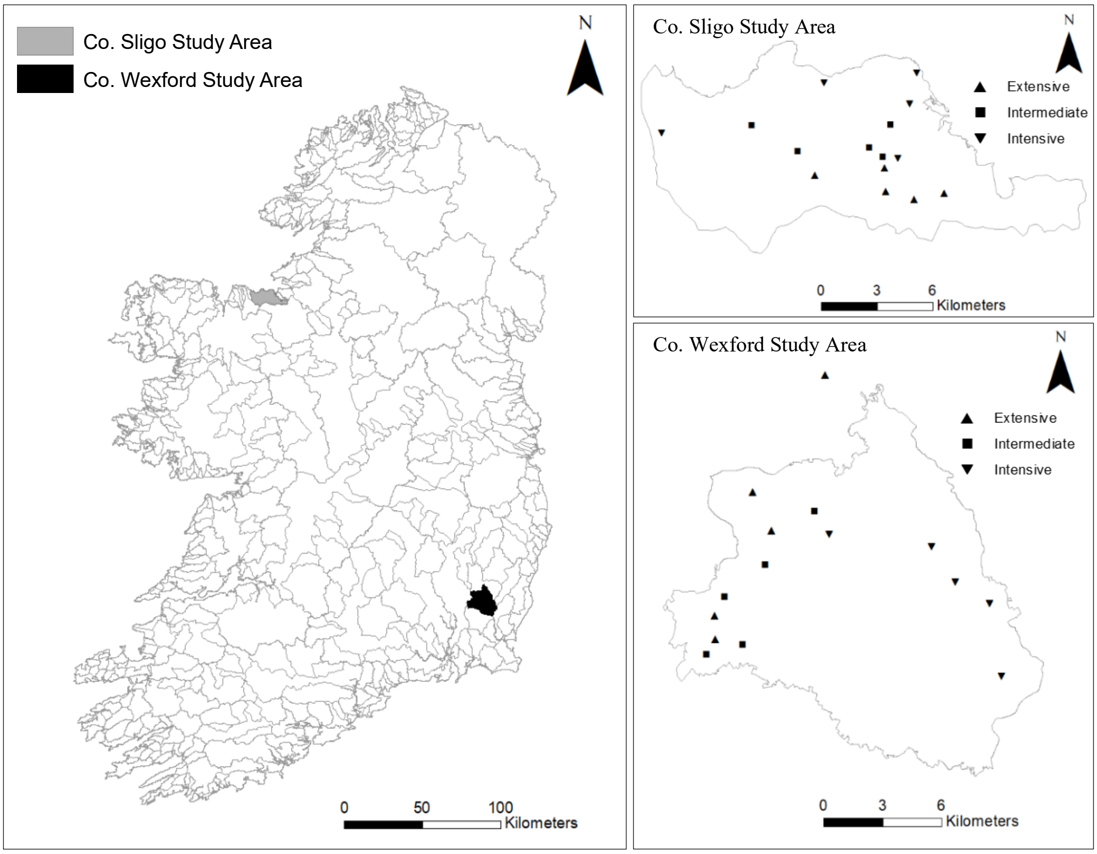


Figure 4.1. Map of Ireland showing location of the study regions for both counties (Sligo and Wexford) and farming intensity categories (Extensive, Intermediate, and Intensive).

4.3.2. Hoverfly sampling and identification

Within each parcel, three pan traps, each consisting of a central wooden stake to which was attached a yellow, white and blue trap, were set up 15 m apart immediately adjacent and parallel to a linear feature (Stanley et al., 2013; Power et al., 2016). Each end of the line of traps was a minimum distance of 10 m from the next nearest linear feature in that parcel. In addition, to avoid spatial autocorrelation, each group of traps was at least 1 km from another group of traps on a different farm. This was largely achieved with the exception of Sites 6 and 7 (Co. Sligo) where, due to the limited availability of farms meeting the criteria for this study, pan traps were 800 m apart (Fig. 4.1; Appendix III.1).

The rims of the pan traps were adjusted to the height of the surrounding vegetation (after Stanley et al. 2013) and all traps set in grazed fields were protected by electric fences which has been shown not to significantly affect insect movement (Wratten et al., 2003). Pan traps were partially filled with soapy water and the captured insects collected after 48 hours. Samples were collected once every three weeks from 15th of May to 15th of August 2019 and preserved in 70% ethanol for later identification. To account for spatiotemporal autocorrelation, all hoverfly samples were pooled for each sample site and sampling visit per site (Happe et al., 2018; Schirmel et al., 2018). This resulted in a total of 30 invertebrate samples for both counties combined (i.e., n = 15 per county).

Hoverflies were identified to genus and species level using Ball and Morris (2015) and Stubbs and Falk (2002) and compared with reference collections previously identified by taxonomic experts. For this study, we categorised the hoverflies into two functional groups based on larval food type (Power et al., 2016; Schirmel et al., 2018; Speight, 2020). Larval food preferences reflect pest suppression and nutrient cycling services (Table 4.1; Appendix III.2).

Table 4.1. Hoverflies and associated functional groups investigated in this study.

Response variable	Variable description
Total Hoverfly	Total hover fly species collected across all sites during the study
Larval Food	
Predacious	Species with larvae which feed on living animals
Non-Predacious	Species with larvae which feed on either decomposing organic matter (Saprotrophic) or living plants.

4.3.3. Selection of explanatory variables

Explanatory variables in this study included those reflecting local and landscape scale habitat features, as well as farming intensity and management as follows:

1) Local variables: Measured on site at or adjacent to insect sampling points (Table 4.2).

2) Landscape variables: Measured, using satellite imagery, within a 500 m buffer zone from the centre of the three traps per site (using ArcGIS10.5; ESRI, 2016). The 500 m buffer was selected to avoid spatial autocorrelation among the sites (Joshi et al., 2016) and is in line with previous studies which suggest that hoverflies are generally influenced by a landscape matrix within a radius of 500 – 1000 m (Kleijn & Van Langevelde, 2006; Meyer et al., 2009). These variables were selected to determine how less detailed habitat assessments (i.e., based on satellite imagery) reflect hoverfly abundance and species richness (Table 4.2).

3) Farm intensity and management (Table 4.2).

Since Nature Value Scores (NVS) were directly calculated for each farm (Rotchés-Ribalta et al., 2020), the NV score was also considered a potential explanatory variable. In addition, we included county as an additional explanatory variable.

Table 4.2. Description of explanatory variables and codes.

Explanatory variables	Variable Code	Type	Variable description
County	County	Categorical	Sligo or Wexford
Local Scale			
Altitude	Altitude	Continuous	Measured as meters above sea level (MSL)
Field Area	FA	Continuous	Area (ha) of field in which trap is placed.
Linear Habitat Quality Score	LHQS	Continuous	Using score cards that included vegetation structure, cover and abundance of positive plant species indicators and poaching levels (after Rotchés-Ribalta et al., 2020).
Field Habitat Quality Score	FHQS	Continuous	Using score cards that included vegetation structure, cover and abundance of positive plant species indicators and poaching levels (after Rotchés-Ribalta et al., 2020).
Shrub Flower Cover	SFC	Continuous	Percentage cover of flower heads for six main flowering shrub species in 30m transects (*Crataegus monogyna*, *Lonicera periclymenum*, *Rosa canina*, *Rubus fruticosus*, *Sambucus spp*, *Ulex europaeus)*
Landscape Scale (within 500m of trap)			
Stone walls and other stonework	BL1*	Continuous	Length of stone walls and other stonework (m)
Earth banks	BL2*	Continuous	Length of earth banks (m)
Drainage ditches	FW4*	Continuous	Length of drainage ditches (m)
Dry meadows & grassy verges	GS2*	Continuous	Length of dry meadows and grassy verges (m).
Hedgerows	WL1*	Continuous	Length of hedgerows (m).
Treelines	WL2*	Continuous	Lengths of treelines (m).

Combined Length of Linear Features	CLLF	Continuous	Combined length of linear features.
Linear Feature Richness	LFR	Continuous	Richness of linear features (up to a maximum of six linear features comprising BL1, BL2, FW4, GS2, WL1 and WL2).
Linear Feature Diversity	LFD	Continuous	Diversity of linear features (Shannon Index using linear feature lengths).
Farming Intensity / Management			
Farming intensity	Farming intensity	Categorical	Farm intensity designation (intensive, intermediate, and extensive) using Nature Value scores after Rotchés-Ribalta et al. (2020).
Nature Value Score	NVS	Continuous	Nature value score calculated for each selected farm based on area owned and farmed, stocking rate, the proportion of improved grasslands and size of fields and boundaries (after Rotchés-Ribalta et al. 2020).

* Habitat codes after Fossitt (2000).

4.3.4. Data analysis

For each site, abundance and species richness were calculated for total hoverflies and for each functional group (i.e., predacious, and non-predacious) as the total number of individuals and species per site, respectively. Given that hoverfly abundance data did not follow a normal distribution, we used Mann-Whitney U Tests to compare abundances of predacious and non-predacious hoverflies for both counties combined and separately. We used Generalized Linear Models (GLMs) in SPSS (V. 27) to examine the response of hoverfly abundance and richness (including functional groups) as response variables (Table 3.1) to the selected explanatory variables (Table 4.2). In addition, we included the interacting effects of county* altitude; county* field habitat quality score (FHQS); county* stonewalls (BL1); county* drainage ditches (FW4); county* combined length of linear features (CLLF) and county* linear feature diversity since these variables were different between the two counties (see Appendix III.3). We used Poisson error GLMs for species richness and negative binomial for abundance data due to overdispersion. Models were simplified by backwards selection using the Akaike information criterion (AIC) (Stanley et al. 2013; Schirmel et al. 2018). In addition, models were validated by checking model residuals for normality and heteroscedasticity. Model simplification and validation was undertaken using R (V.4.0.2) in conjunction with Brodgar (V.2.7.5; Highland Statistics Ltd.).

Given that the predatory species *E. blateatus* was dominant (representing 42% of the catch overall), this species was analysed separately and excluded in the GLM analysis for abundance data (after Schirmel et al. 2018) since its inclusion would have given rise to abundance models reflecting this rather than the other species with their associated functional groups. While *E. balteatus* did not follow a normal distribution (i.e., after transformations), we used the non-parametric Kruskal-Wallis test to determine differences in *E. balteatus* abundance across different levels of farming intensity. In addition, we used Spearman's correlation to determine correlations between the abundance of *E. balteatus* and the continuous explanatory variables (Table 4.2).

To investigate patterns in hoverfly community structure, and to identify the drivers of any differences in community composition between the two counties, an ordination analysis was performed in PC-ORD (V.6) using non-metric multidimensional scaling ordination biplot (NMS) and Sørenson as a distance measure (McCune et al., 2002; McCune & Mefford, 2011). The number of significant axes was determined through 250 runs of real data compared with 250 runs with randomised data. An orthogonal principal axis output was selected for each NMS to illustrate maximum community variation along axis 1. Explanatory variables were utilised as a second matrix and variables with Pearson R^2 values > 0.2 overlain as a biplot (McCune and Mefford, 2011).

4.4. Results

4.4.1. Hoverfly abundance and species richness across study regions

For both study regions combined, we collected a total 53 hoverfly species (1,165 individuals) representing 28.6% of all known Irish syrphid fauna (Speight 2020). All the recorded species are common in Ireland and in Europe (Speight 2020). Fifty percent of the entire catch consisted of *Episyrphus balteatus* (42%) and *Syrphus ribesii* (8%) (Fig. 4.2a). In Co. Wexford sites, 34 species and 703 individuals were recorded, with 68% of the catch consisting of *E. balteatus* alone (Fig. 4.2b). In contrast, four species made up almost 50% of the catch at the Co. Sligo sites where 45 species and 462 individuals were captured (*S. ribesii* (13%); *Helophilus pendulus* (10%); *Rhingia campestris* (10%); *Melanogaster hirtella* (9%); and *Platycheirus albimanus* (7%) - Fig. 4.2c).

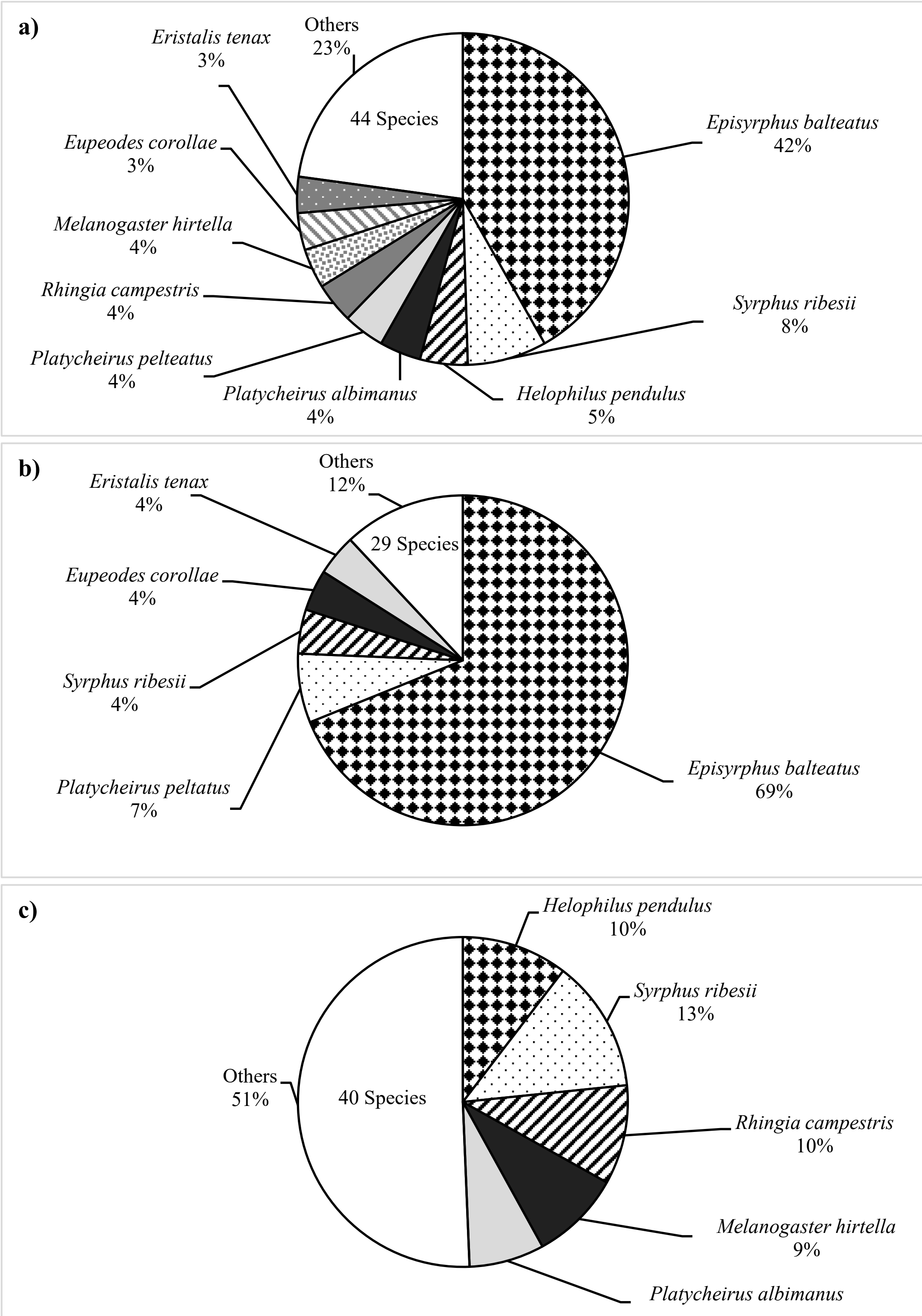


Figure 4.2. Percentage of the most abundant hoverfly species collected for: a) both counties combined; b) Co. Wexford; and c) Co. Sligo (each of the 40 species under "Others" constitutes < 5% of the overall catch).

The abundance of predacious hoverflies was significantly greater than non-predacious species overall and for Co. Wexford (U = 292.5, P = 0.020; U = 56.5, P = 0.019 respectively) while no significant differences were detected at the Co. Sligo sites (Fig. 4.3).

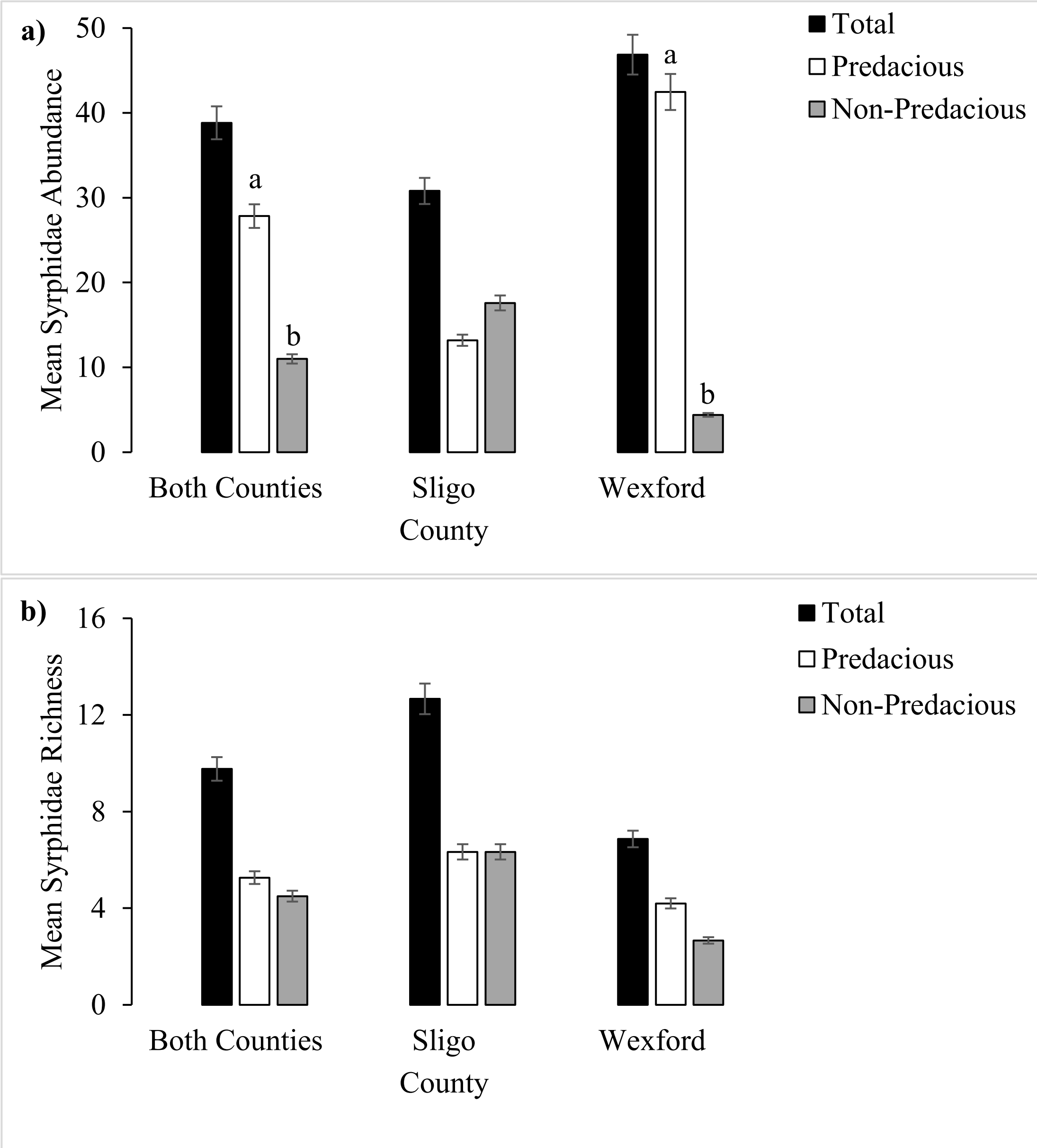


Figure 4.3. Mean abundance (a) and mean species richness (b) of total hoverflies with associated functional groups collected on 30 farms across the two study regions in Ireland. Bars with the different letters within the study regions indicate significant differences between predacious and non-predacious groups (Mann-Whitney U test; P<0.05).

4.4.2. Hoverfly response to variables at local scale

For both counties combined, altitude showed a significant negative effect on the species richness of predatory hoverflies. In addition, there was a significant interacting effect of altitude and county on total hoverfly abundance and richness and on the abundance of predacious and non-predacious species (Table 4.3; Fig. 4.4). In Co. Sligo, altitude showed a significant negative effect on the abundance and species richness of hoverfly (total) and on the abundance of predacious and non-predacious species. However, in Co. Wexford, altitude showed a significant negative effect on total hoverfly abundance only (Table 4.3; Fig. 4.4).

For both counties combined, field area (FA) showed a significant negative effect on the abundance of predacious species. Field habitat quality score (FHQS) showed a significant positive effect on the abundance and species richness of non-predacious hoverflies. There was also a significant interacting effect of FHQS and county on the abundance of predacious hoverflies with a significant positive effect in Co. Wexford in comparison with Co. Sligo where no significant effect was detected (Table 4.3; Fig. 4.4). Spearman's correlations (Table 4.4) between explanatory variables and the abundance of the dominant predatory species *E. balteatus* in Co. Wexford showed significant correlations with altitude (negative) and with LHQS (positive).

4.4.3. Hoverfly response to variables at landscape scale

At landscape scale, length of hedgerows (WL1) and treelines (WL2) were the most frequently significant factors (for both counties combined), showing a significant positive effect on total hoverfly abundance and abundance of hoverfly functional groups. In addition, WL2 showed a significant positive effect on total hoverfly species richness and species richness of hoverfly functional groups (Table 4.3; Fig. 4.4).

There was an interacting effect of length of stone walls (BL1) and county, and drainage ditches (FW4) and county, on the abundance of total hoverflies and hoverfly functional groups (Table 4.3; Fig. 4.4) with significant positive effects in Co. Sligo. Meanwhile, in Co. Wexford, both BL1 and FW4 showed a significant negative effect on total hoverfly abundance and abundance of predatory species, respectively. On the other hand, earth banks (BL2) showed a significant negative effect on the species richness of non-predacious species for both counties combined (Table 4.3; Fig. 4.4). Moreover, Spearman's correlation showed a significant positive correlation between the abundance of *E. balteatus* and BL2 for both counties combined (Table 4.4). In addition, *E. balteatus* also showed a significant positive correlation with length of grassy verges and dry meadows (GS2) for both counties combined and for Co. Wexford separately (Table 4.4).

The combined length of linear features (CLLF) showed a significant negative effect on the abundance of total hoverflies (both counties separately) and the abundance of predacious and non-predacious species (both counties combined; Table 4.3; Fig. 4.4). In addition, linear feature richness (LFR) showed a significant negative effect on the abundance and species richness of non-predacious species for both counties combined; while linear feature diversity (LFD) showed a significant positive effect on these hoverfly groups (Table 4.3; Fig. 4.4). Furthermore, there was interacting effect of LFD and county on the abundance of total and predacious hoverfly species with significant positive effects in Co. Wexford (Table 4.3; Fig. 4.4).

Table 4.3. (A-C) Results from final GLMs showing the effects local, landscape and farm management variables on the abundance and species richness of hoverflies collected across 30 farms in Co. Sligo and Co. Wexford. Numbers in bold indicate significant P values ($P < 0.05$).

A- Total Hoverflies

	Estimate	χ2	df	P
Abundance				
(Intercept)	4.117	9.423	1	**0.002**
CLLF* County	—	57.313	2	**<0.001**
CLLF*Sligo	-0.001	13.461	1	**<0.001**
CLLF* Wexford	-0.002	21.737	1	**<0.001**
SFC	0.004	2.039	1	0.153
LHQS	-0.072	0.005	1	0.943
FHQS* County	—	2.131	2	0.345
Field Area	-0.170	0.706	1	0.401
LFD* County	—	36.492	2	**<0.001**
LFD* Sligo	-0.936	1.761	1	0.185

LFD* Wexford	4.828	20.534	1	**<0.001**
NVS	0.050	0.207	1	0.649
WL1	0.001	12.085	1	**<0.001**
Altitude* County	—	13.894	2	**0.001**
Altitude* Sligo	-0.015	9.020	1	**0.003**
Altitude* Wexford	-0.006	5.614	1	**0.018**
FW4* County	—	13.707	2	**<0.001**
FW4* Sligo	0.001	12.051	1	**<0.001**
FW4* Wexford	-0.001	0.870	1	0.351
Farming Intensity	—	7.580	2	**0.023**
WL2	0.001	22.241	1	**<0.001**
BL1* County	—	19.853	2	**<0.001**
BL1* Sligo	0.001	10.519	1	**0.001**
BL1* Wexford	-0.002	10.913	1	**<0.001**
Richness				
(Intercept)	3.743	10.078	1	**0.002**
Field Area	-0.178	1.096	1	0.295
NVS	-0.093	0.707	1	0.400
LHQS	-0.969	1.216	1	0.270
Farming Intensity	—	1.081	2	0.582
FW4* County	—	7.222	2	**0.027**
FW4* Sligo	<0.001	3.162	1	0.075
FW4* Wexford	-0.001	1.09	1	0.296
FHQS	1.325	3.005	1	0.083
Altitude* County	—	8.897	2	**0.012**
Altitude* Sligo	-0.013	8.058	1	**0.005**
Altitude* Wexford	-0.002	0.732	1	0.392
LFD	0.389	0.438	1	0.508
BL1* County	—	0.611	2	0.737
WL1	<0.001	0.442	1	0.506
CLLF	<-0.001	1.476	1	0.224
WL2	0.001	10.267	1	**0.001**

B- Predacious hoverflies

	Estimate	$\chi 2$	df	P
Abundance				
(Intercept)	6.093	7.205	1	**0.007**
LHQS	-1.746	1.205	1	0.272
SFC	-0.002	0.373	1	0.541
Farming Intensity	—	3.002	2	0.223
LFD* County	—	16.078	2	**<0.001**

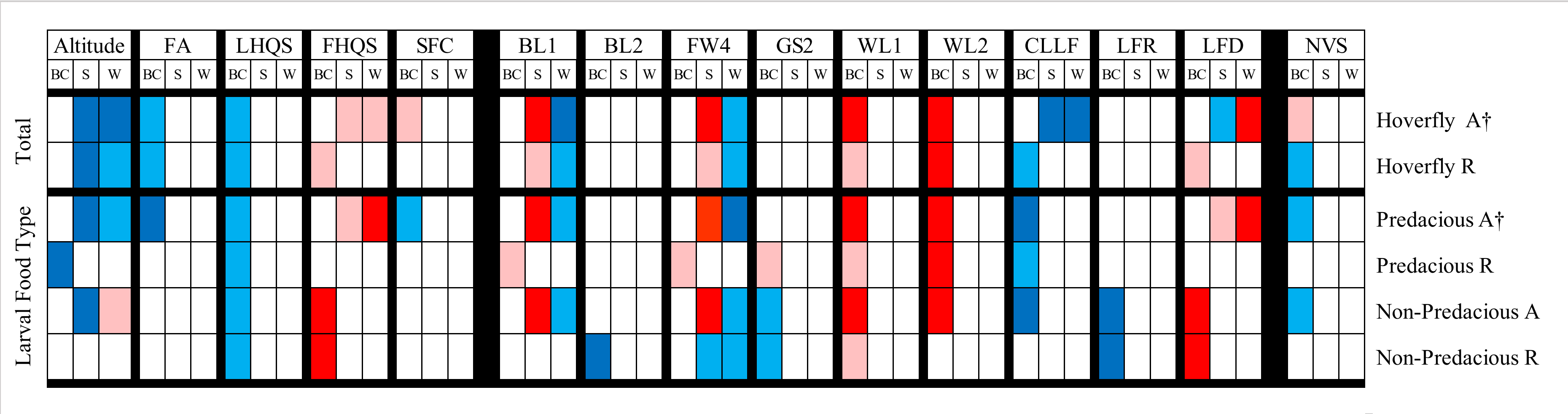
LFD* Sligo	1.120	1.666	1	0.197
LFD* Wexford	5.842	15.24	1	**<0.001**
NVS	-0.266	2.439	1	0.118
Altitude* County	—	6.987	2	**0.030**
Altitude* Sligo	-0.015	4.92	1	**0.027**
Altitude* Wexford	-0.008	3.546	1	0.060
FHQS* County	—	8.354	2	**0.015**
FHQS* Sligo	1.619	2.173	1	0.140
FHQS* Wexford	5.693	8.251	1	**0.004**
Field Area	-.719	5.852	1	**0.016**
WL1	0.002	8.409	1	**0.004**
BL1* County	—	8.479	2	**0.014**
BL1* Sligo	0.001	7.419	1	**0.006**
BL1* Wexford	-0.001	1.687	1	0.194
FW4* County	—	22.7	2	**<0.001**
FW4* Sligo	0.002	13.189	1	**<0.001**
FW4* Wexford	-0.004	10.942	1	**<0.001**
WL2	0.002	20.836	1	**<0.001**
CLLF	-0.002	12.851	1	**<0.001**
Richness				
(Intercept)	2.886	20.396	1	**<0.001**
GS2	<0.001	0.372	1	0.542
LHQS	-1.199	1.607	1	0.205
BL1	<0.001	0.3	1	0.584
WL1	<0.001	0.685	1	0.408
FW4	<0.001	1.238	1	0.266
CLLF	<0.001	1.074	1	0.300
WL2	0.001	4.416	1	**0.036**
Altitude	-0.004	9.608	1	**0.002**

C- Non-predacious hoverflies

	Estimate	χ2	df	P
Abundance				
(Intercept)	3.842	13.245	1	**<0.001**
GS2	<0.001	0.158	1	0.691
LHQS	-1.584	1.084	1	0.298
LFR	-.691	9.542	1	**0.002**
NVS	-.135	0.975	1	0.323
LFD	2.309	8.422	1	**0.004**
WL2	<0.001	5.400	1	**0.020**
Altitude* County	—	16.391	2	**<0.001**

Altitude* Sligo	-.020	16.221	1	**<0.001**
Altitude* Wexford	<0.001	0.003	1	0.958
FW4* County	—	20.839	2	**<0.001**
FW4* Sligo	0.001	9.608	1	**0.002**
FW4* Wexford	-0.001	3.192	1	0.074
BL1* County	—	10.315	2	**0.006**
BL1* Sligo	0.001	7.138	1	**0.008**
BL1* Wexford	-0.001	1.084	1	0.298
Farming Intensity	—	3.607	2	0.165
FHQS	2.241	8.792	1	**0.003**
WL1	0.001	10.870	1	**<0.001**
CLLF	-0.001	10.418	1	**0.001**
Richness				
(Intercept)	2.522	4.830	1	**0.028**
Altitude* County		5.105	2	0.078
LHQS	-1.768	1.477	1	0.224
FHQS	1.028	6.145	1	**0.013**
BL1* County	—	4.373	2	0.112
WL1	<0.001	2.578	1	0.108
GS2	-0.001	3.357	1	0.067
BL2	-0.001	4.545	1	**0.033**
FW4* County	—	5.537	2	0.063
LFR	-0.522	5.621	1	**0.018**
LFD	2.075	7.354	1	**0.007**

Acronyms: FA= Field Area; LHQS = Linear Habitat Quality Score; FHQS = Field Habitat Quality Score; SFC = Shrub Flower Cover; BL1 = Stone Walls and Other Stonework; BL2 = Earth Banks; FW4 = Drainage Ditches; GS2 = Dry Meadows & Grassy Verges; WL1 = Hedgerows; WL2 = Treelines; CLLF = Combined Length of Linear Features; LFR = Linear Feature Richness; LFD = Linear Feature Diversity; NVS= Nature Value Score.



† Excluding the dominant species *E. balteatus*

Figure 4.4. Heat map showing estimated coefficients from GLM analysis for total hoverflies, and hoverfly functional groups (A: Abundance; R: Richness) and explanatory variables (n = 30). Red and blue indicate positive and negative effects with dark red and dark blue indicating significant positive and negative effects, respectively.

Acronyms: FA= Field Area; LHQS = Linear Habitat Quality Score; FHQS = Field Habitat Quality Score; SFC = Shrub Flower Cover; BL1 = Stone Walls and Other Stonework; BL2 = Earth Banks; FW4 = Drainage Ditches; GS2 = Dry Meadows & Grassy Verges; WL1 = Hedgerows; WL2 = Treelines; CLLF = Combined Length of Linear Features; LFR = Linear Feature Richness; LFD = Linear Feature Diversity; NVS= Nature Value Score; BC = Both Counties, S = Co. Sligo and W = Co. Wexford.

Table 4.4. Spearman's correlation (rs) between the abundance of dominant species *Episyrphus balteatus* and the explanatory variables for both counties combined and separately. Numbers in bold indicate significant correlations (*P<0.05; **P<0.01).

	Both Counties		Co. Sligo		Co. Wexford	
	r_s	P	r_s	P	r_s	P
Altitude	0.000	0.999	-0.343	0.210	$-.781^{**}$	**0.001**
FA	0.314	0.092	0.058	0.837	0.391	0.150
LHQS	$.449^{*}$	**0.013**	0.403	0.136	$.730^{**}$	**0.002**
FHQS	$-.392^{*}$	**0.032**	-0.147	0.602	-0.457	0.087
SFC	-0.227	0.229	0.156	0.579	-0.454	0.090
BL1	$-.630^{**}$	**0.000**	-0.459	0.085	$-.618^{*}$	**0.014**
BL2	$.440^{*}$	**0.015**	0.174	0.536	0.453	0.090
FW4	-0.177	0.349	0.471	0.076	-0.079	0.778
GS2	$.449^{*}$	**0.013**	-0.277	0.318	$.693^{**}$	**0.004**
WL1	0.172	0.364	0.447	0.095	-0.139	0.621
WL2	0.026	0.893	-0.379	0.163	0.488	0.065
CLLF	-0.138	0.468	0.140	0.619	0.126	0.653
LFR	0.080	0.676	-0.090	0.751	0.377	0.166
LFD	-0.064	0.738	-0.459	0.085	0.435	0.105
NVS	$-.414^{*}$	**0.023**	-0.317	0.250	$-.619^{*}$	**0.014**

Acronyms: FA= Field Area; LHQS = Linear Habitat Quality Score; FHQS = Field Habitat Quality Score; SFC = Shrub Flower Cover; BL1 = Stone Walls and Other Stonework; BL2 = Earth Banks; FW4 = Drainage Ditches; GS2 = Dry Meadows & Grassy Verges; WL1 = Hedgerows; WL2 = Treelines; CLLF = Combined Length of Linear Features; LFR = Linear Feature Richness; LFD = Linear Feature Diversity; NVS= Nature Value Score.

4.4.4. Hoverfly response to farm management type

GLMs followed by pairwise comparisons (LSD) showed significant differences in the abundance of total hoverfly in relation to farming intensity (Table 4.3) with significantly more hoverfly individuals in extensive farms in comparison with intermediate (P = 0.018) and intensive farms (P = 0.008; Fig. 4.5). In addition, the abundance of total hoverflies was also significantly greater in intermediate farms in comparison with intensive farms (P = 0.018; Fig. 4.5). The non-parametric Kruskal

Wallis test also showed significant differences (P < 0.05) in the abundance of the dominant species *E. balteatus* in relation to farming intensity. Significantly more individuals of *E. balteatus* were detected in intensive farms in comparison with extensive (P < 0.05) and intermediate (P < 0.05) farms (see Appendix III.4). Moreover, *E. balteatus* showed a significant negative correlation with Nature Value Score (NVS) for both counties combined and Co. Wexford (Table 4.4). No significant effects of farming intensity on species richness were detected (see Appendix III.4 for mean values).

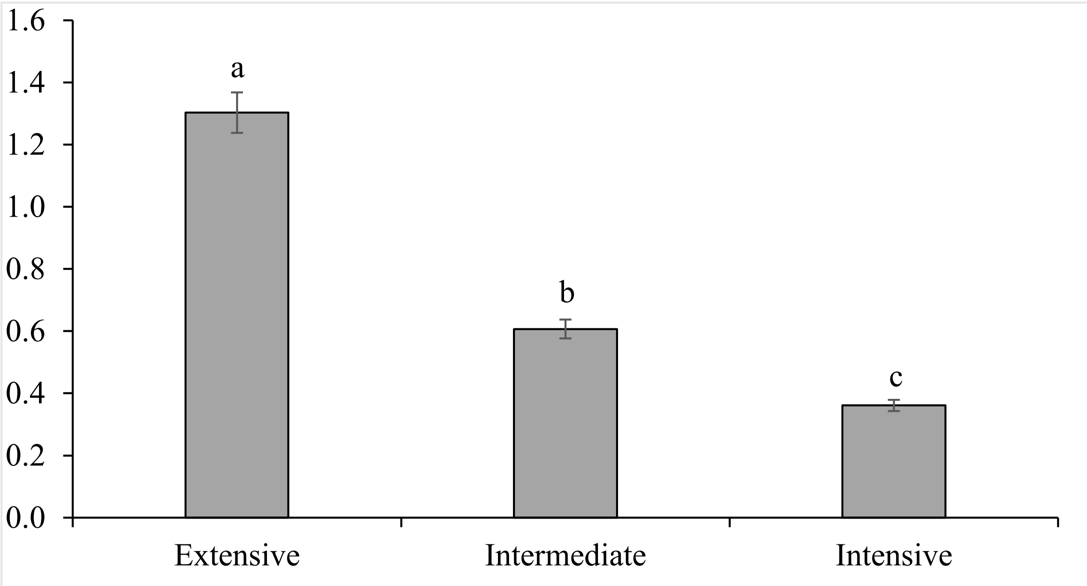


Figure 4.5. Mean (LS-means + SE) abundance of total hoverflies (without *E. balteatus*) collected across a range of farming intensities (n = 10 for each farming intensity). Different letters above the bars indicate significant differences across the three levels of farming intensity (GLMs and LSD pairwise comparison; P < 0.05 - see Appendix III.4 for raw means).

4.4.5. Community analysis

Multiple response permutation procedure (MRPP) showed significant differences in the community composition of hoverflies between the two counties (A = 0.06, P <0.001) as has been visualised by nonmetric multidimensional scaling (NMS) analysis (Fig. 4.6). Most of the species in Wexford sites were predatory species, in particular *Episyrphus balteatus*, *Platycheirus peltatus* and *Eupeodes corollae* (Fig. 4.6; Appendix III.2).

At local scale, the community composition of hoverflies was driven mainly by altitude in Co. Wexford while it was driven by habitat quality score of fields (FHQS) in which traps were placed in Co. Sligo. Furthermore, the diversity, and availability of certain linear features at a landscape scale were the main driving factors affecting hoverfly community composition, with linear feature diversity (LFD) and drainage ditches (FW4) being the most important driving factors in Co. Sligo while dry meadows and grassy verges (GS2) were the main driving factors in Co. Wexford (Fig. 4.6).

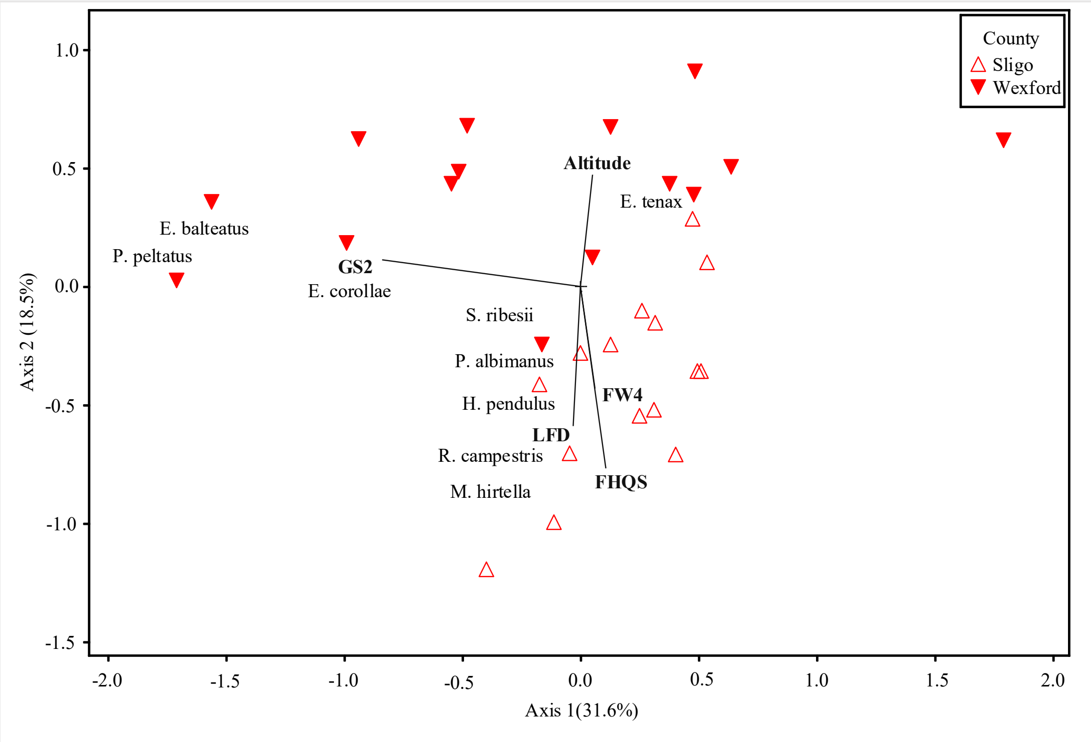


Figure 4.6. Non-metric multi-dimensional scaling ordination of hoverfly samples collected across 30 farms on Irish farmland. Only the most abundant hoverfly species (≥ 4%) for each study region are shown. The first two axes explain 50.1% of the variation (31.6% axis 1 and 18.5% axis 2) with an orthogonality of 100%. County is a significant grouping variable (A= 0.06, P<0.001; MRPP chance-corrected within-group agreement A). Acronyms: GS2 = Dry Meadows and Grassy Verges; FW4 = Drainage Ditches; LFD = Linear Feature Diversity and FHQS = Field Habitat Quality Score.

4.5. Discussion

4.5.1. Hoverfly response to variables at local scale

At local scale, field habitat quality score (FHQS) showed a significant positive effect on non-predacious species richness and abundance for both counties combined. This could potentially be explained by the benefits of fields of high habitat quality providing valuable floral resources and heterogenous vegetation structure (see Rotchés-Ribalta et al., 2020). These high-quality farm habitats (with more resources) have been shown to increase the abundance and species richness of hoverflies (including non-predatory species) at local scale (Power et al., 2016; Lucas et al., 2018). In addition, fields with good habitat quality for aerial invertebrates often provide better structural condition due to the presence of taller vegetation (Ryder et al., 2005), low management pressure and frequently, wetter, more organic soils, particularly important elements for saprophytic larvae. On the other hand, while predatory hoverflies are known to supplement their diet with nectar and pollen as well as aphids (Rotheray and Gilbert, 2011; Lucas et al., 2018), fields with high habitat quality can provide continuous, multiple resources given that they harbour more flowering plant species and heterogenous vegetation structure in comparison to fields of low habitat quality (Rotchés-Ribalta et al., 2020). The significant positive effect of fields with better habitat quality in Co. Wexford (where farming is generally more intensive than in Co. Sligo) on the abundance of the dominant functional group i.e. predacious hoverflies, can also be seen in this study. This is further supported by the significant negative effect of field area on predacious hoverfly abundance (excluding *E. balteatus*) for both counties combined where larger fields are generally associated with more intensive farming practices (Lucas et al., 2018). While intensive tillage farms may provide some resources for hoverfly larval stages such as aphids, hoverfly food and other resources are generally known to be either ephemeral or scarce in field crops (Garratt et al., 2017; Schirmel et al., 2018; Dover, 2019; Cole et al., 2020) in comparison to plant species-rich, structurally heterogenous grasslands (Langlois et al., 2020).

It is interesting to note that the dominant hoverfly species *Episyrphus balteatus*, a predator, showed a significant negative correlation with field habitat quality score. However, *Episyrphus balteatus* is a highly mobile, migratory species which is found in a wide range of habitats including cereal field crops and is more strongly driven by local aphid density than floral resources (Schirmel et al., 2018). A significant positive correlation with the linear habitat quality score for *E. balteatus* for both counties combined, however, demonstrates their importance (in particular hedgerows) in providing floral resources, shelter and prey for pest control species which spill over into neighbouring fields (Haenke et al., 2009; Haenke et al., 2014). This is supported by studies which show significantly greater abundances of *E. balteatus* in woody semi-natural habitats (e.g., hedgerows) than in open fields (Schirmel et al., 2018) wherein *E. balteatus* (a weakly sclerotized species) avoids dehydration, preferring a humid environment, in particular woody linear habitats, for oviposition and larval development (Schirmel et al., 2018). Further investigations for other predatory species are required given that different species may respond differently to drivers of biodiversity decline (Newbold et al., 2020). Nevertheless, it is important to further develop agri-environmental (AE) measures that can improve both field and linear habitat quality scores at a local scale to sustain different hoverfly species that deliver multiple ecosystem services,

In contrast to FHQS, there was a significant negative effect of altitude on the species richness of predatory hoverflies for both counties combined. Altitude showed a significant negative effect on total hoverfly abundance for each county while the abundance of predacious/non-predacious species and total species richness was affected by altitude in Co. Sligo. Our results agree with the review by Hodkinson (2005) which has shown that altitudinal gradients in general negatively affect terrestrial insect species and their communities, including an overall decline in the species richness of plants and their insect pollinator systems. Other studies have shown significant negative correlations between altitude and arthropod/parasitoid family abundance/richness (Poveda et al., 2012) including hoverflies (De Groot & Vrezec, 2019).

4.5.2. Hoverfly response to variables at landscape scale

At landscape scale, hedgerows (WL1) and treelines (WL2) were the most frequently significant factors showing a significant positive effect on the abundance of total hoverflies and their associated functional groups (for both counties combined). Treelines also showed a significant positive effect on the species richness of total and predacious species for both counties combined. Overall, the results in this study show that woody semi-natural habitats such as hedgerows and treelines (irrespective of differences in climatic, topographic and farming conditions between the two counties) can play a significant role in sustaining hoverfly species richness and/or abundance. They provide valuable resources for both adult and larval stages as demonstrated by previous studies on hoverflies as well as bees (Garratt et al., 2017; Schirmel et al., 2018; Volpato et al., 2020). Hedgerows can provide valuable resources for pest control species which spill over into neighbouring fields (Haenke et al., 2009; Haenke et al., 2014) with significantly more predatory hoverfly species having been recorded in woody semi-natural habitats (e.g., hedgerows/treelines) than in open fields (Schirmel et al., 2018). In addition to providing hoverfly food sources either in the form of prey or floral resources, hedgerows can provide additional resources including, shelter, breeding sites and protection from pesticides or harsh weather conditions particularly in intensive farms (Garratt et al., 2017; Dover, 2019). The results in this study, therefore, highlight the need to develop measures that can increase woody semi-natural habitats on farmland at landscape scale to sustain hoverflies and the associated multiple ecosystem services they deliver.

While both drainage ditches (FW4) and stone walls (BL1) showed significant positive effects on the abundance of total hoverfly and both functional groups in Co. Sligo, a negative pattern for these habitat features was detected in Co. Wexford where BL1 and FW4 showed a significantly negative effect on the abundance of total and predacious hoverflies, respectively. It is acknowledged that excessive input of agrochemicals negatively affect drainage ditch habitat quality (Verdonschot et al., 2012). However, in this study it is not clear whether these different effects between the two counties are due to differences in habitat quality of these features. Given that it is not possible to measure habitat quality score of these features at landscape scale using satellite images, further research based on field surveys is required to provide evidence on the effects of farming practices on the habitat quality of these features and their influence on hoverflies. Therefore, unlike hedgerows and treelines which appear to be important irrespective of farming conditions and can be quantified by employing satellite/aerial images, using these non-woody habitat features alone might not be a suitable measure to use in agri-environmental schemes at a national scale as evidence for payment for the provision of multiple ecosystem services.

Linear feature diversity (LFD) also showed a significant positive effect on the abundance and species richness of non-predacious species for both counties combined as well as positive effects on the abundance and species richness of total and predacious hoverflies for Co. Wexford. While landscape complexity is known to sustain hoverfly abundance and diversity in general (Haenke et al., 2009; Haenke et al., 2014; Schirmel et al., 2018), less is understood about the role of specific linear features at landscape scale to invertebrates with different ecosystem services (Power et al., 2016; Schirmel et al., 2018). The positive effects of LFD on hoverfly richness and abundance are in line with previous studies which have shown that conservation of hoverflies requires different resources provided by semi-natural habitats at landscape scale and local scale (Moquet et al., 2018; Schirmel et al., 2018). However, given that diversity considers the proportion of each linear feature in relation to the combined length of the linear features, it is likely that the positive effects of linear feature diversity are due to the dominance of hedgerows and treelines (i.e., woody SNHs) rather than other non-woody SN linear features particularly in Co. Wexford. This may explain why LFR showed significant negative effects on the abundance and species richness of non-predacious hoverflies for both counties combined since LFR does not consider the proportion of each linear feature alone, thereby reducing the positive influence of hedgerows and treelines. This may explain further the negative effects of the combined length of linear features (CLLF) on the abundance of total hoverflies (both counties separately) and their functional groups (both counties combined) since different linear features may contribute differently to hoverfly abundance. Thus, given that other linear features (e.g., BL1 and FW4), showed variable results between the two counties, the quantity of certain linear features alone, as well as in combination, does not appear to be a good measure to quantify hoverflies and their associated ecosystem services at a landscape scale.

4.5.3. Hoverfly species composition across study regions and driving factors

As expected, the two counties showed significant differences in their hoverfly species composition as indicated by MRPP analysis. In addition, NMDS analysis revealed distinct differences in the driving factors affecting hoverfly species composition between the two counties. This difference in hoverfly species composition reflects, in general, the contrasting climatic, topographic, and farming conditions (Rotchés-Ribalta et al. 2020) where farms in Co. Wexford have better soils for agricultural production with larger field sizes. In general, the predatory species (in particular *E. balteatus)* were the most abundant species for Co. Wexford sites which may reflect farm type where four of the five intensive farms were either tillage (3) or mixed farms with tillage (1) in contrast to Co. Sligo where all the farms were livestock farms. It is hypothesised that predatory hoverflies such as *E. balteatus* can cope relatively well with intensive tillage farms, given that they provide higher prey densities, an important larval food source (Meyer et al., 2009; Haenke et al., 2014; Schirmel et al., 2018). Prey density (aphids) may also provide additional food sources in the form of honeydew, an important food source for adult hoverflies especially when flower resources are scarce in intensive field crops, thereby reducing adult foraging time (van Rijn et al., 2013; Pinheiro et al., 2015). *Episyrphus baltatus* is a highly mobile species which may explain the greater abundances of *E. balteatus* in Co. Wexford where they are ubiquitous and found in a wide range of habitats including cereal field crops (Speight, 2020) where their immature stages can overwinter (Raymond et al., 2014).

On the other hand, the most abundant species for Co. Sligo sites (e.g., *Syrphus ribesii*, *Helophilus pendulus*, *Rhingia campestris*, *Melanogaster hirtella* and *Platycheirus albimanus*) were relatively similar in their abundance despite having different larval food types. This is likely due to more complex landscapes in Co. Sligo as a result of smaller field sizes, the absence of tillage farms and a higher proportion of semi-natural linear habitats. These complex landscapes (with SNHs) can sustain hoverfly abundance and diversity (Haenke et al., 2009; Haenke et al., 2014; Schirmel et al., 2018) by providing multiple resources for larval development particularly for non-predatory species (Schirmel et al., 2018). This may explain why greater abundances of non-predacious species were found in Co. Sligo in agreement with the findings of Schirmel et al. (2018) who found greater abundances of non-predacious hoverflies in complex landscapes in comparison with more simple landscapes.

The most significant driving factors in Co. Wexford was altitude and dry meadows & grassy margins (GS2). Given that Co. Wexford has higher altitudes than Co. Sligo, the greater effects of altitude on hoverfly communities in Co. Wexford is not surprising since higher altitudes negatively affect a range of terrestrial insect species and their communities as been reviewed by Hodkinson (2005). Moreover, while the length of GS2 was greater in Co. Wexford in comparison with Co. Sligo, it must be noted that it was easier to distinguish GS2 using satellite images in Co. Wexford where many were immediately adjacent to tillage crops in comparison to Co. Sligo where they could sometimes blend in with immediately adjacent grassland fields (see Schirmel et al., 2018). It is, therefore, likely that the effects of GS2 on hoverfly communities are similar to those of grassland fields in Co. Sligo since all these farms were grasslands. Conversely, in Co. Wexford, measuring GS2 length was relatively easier due to better differences between field margins and crop fields. Thus, grassy semi-natural habitats (e.g., GS2) may affect hoverfly communities in Co. Wexford by providing adult (pollen, nectar and honeydew) as well as larval food sources particularly aphids important for predatory species (Schirmel et al., 2018). On the other hand, the most important driving factors for Co. Sligo at local scale was field habitat quality score which may reflect the overall higher field habitat quality in Co. Sligo in comparison with Co. Wexford (Table S3) as a result of lower farming intensity and/or farm types (grassland). Therefore, it appears that low intensity farming practices may generate favourable conditions that increase field habitat quality score such as positive plant indicator species or more flowering plants, low poaching levels and diversity in vegetation structure (see Rotchés-Ribalta et al., 2020) which are important for sustaining hoverfly communities.

The results in this study indicate that the diversity of linear features (e.g., landscape complexity) at landscape scale (particularly diversity of woody semi-natural habitats), is important in sustaining hoverfly communities in agreement with a recent study by (Schirmel et al., 2018). These results also show that specific linear features such as watercourses (FW4) appear to be important in sustaining hoverfly communities in Co. Sligo since watercourses are known to provide valuable habitats as well as food sources to both aquatic and terrestrial invertebrates, including rare species (Herzon & Helenius, 2008). FW4 plays an important role in habitat connectivity within wider landscapes and their function in regulating water flow and nutrient retention is likely to depend on the biological communities of watercourses (Herzon & Helenius, 2008). This is particularly important when drainage ditches are associated with woody semi-natural habitats such as hedgerows (D.Ahmed et al., 2020) where they can increase hoverfly abundance by 30% (Speight, 2008).

4.6. Conclusions and management implications

The results of this study indicate that at landscape scale, hedgerows and tree lines (irrespective of county and farming conditions) are the most important driving factors positively affecting hoverfly abundance/richness, with other linear habitat features showing variables results. Since hedgerows, and treelines appear to best reflect hoverflies that deliver multiple ecosystem services, our results have important implications for national agri-environmental payment schemes where farmers can be awarded based on the total length of hedgerows and treelines on their farms. These habitat features can be rapidly measured through satellite imagery rather than extensive, time-consuming field surveys. The results also show that at local scale, field habitat quality score is an important driving factor positively affecting abundance of non-predatory hoverflies as well as hoverfly communities. This reinforces the importance of biodiversity friendly farming practices to increase field habitat quality (e.g., through low intensity farming) to sustain more hoverfly communities with their associated ecosystem services.

In relation to management practices, the results here indicate that intensive farming can benefit selected hoverfly species (such as generalist, predatory species *E. balteatus),* by providing ephemeral or scarce food sources in adjacent field crops. However, these systems result in lower total hoverfly abundance and diversity. Thus, future agri-environment measures should support farmers to maintain or improve field habitat quality to sustain hoverfly communities with multiple ecosystem services. Such schemes should consider heterogeneity of habitats in terms of quality and quantity and low intensity/mixed farms to maximize biodiversity and associated ecosystem services on farmland.

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Chapter 5 General Discussion

5.1. Discussion

Results-based agri-environment payment schemes (RBAPSs) reward farmers primarily for improving local habitat quality rather than the creation of new features and habitats (European Commission, 2021). Despite the promising biodiversity outputs from such schemes, there are still gaps in knowledge in relation to the tools that could be used for assessing the quality of farm habitats. Assessing the biodiversity value of farm habitats based on plant indicator species alone is not a sufficient measure to determine high habitat quality since it does not consistently reflect terrestrial invertebrates including rare species (Maher et al., 2014; Williams et al., 2014).

There is a growing interest in the use of bioindicators (including invertebrates) for habitat evaluation, thereby minimising the time and effort required for intensive and expensive sampling procedures, and playing a crucial role in the identification and protection of biodiverse farmlands (Boyle et al., 2015; Power et al., 2016; Carey et al., 2017a & b; Schirmel et al., 2018; Volpato et al., 2020). However, while using invertebrates for rapid assessment of the quality of farm habitats is a promising tool (that could be incorporated into RBAPSs), much focus to date has been on iconic invertebrates in particular pollinators such as wild bees and butterflies (Carvell et al., 2007; Pywell et al., 2011; Volpato et al., 2020). Other insect groups, particularly Diptera, are often less studied. Therefore, to maximize biodiversity gain both quantitatively (e.g., pollinators abundance) and qualitatively (e.g., multiple ecosystem services abundance and richness), it is crucial to conserve wider biodiversity including other invertebrate species with different ecosystem services. The immature stages of certain species of the dipteran family Sciomyzidae which feed on pestiferous gastropods, for example, deliver important ecosystem services that are not provided by bees and hoverflies.

With this in mind, the studies presented in this thesis aim firstly to fill gaps in the knowledge regarding the ecosystem function of the slug killing fly *Tetanocera elata* (Diptera:Sciomyzidae), an understudied taxon found across a range of farm habitats along with other better known dipteran taxa such as hoverflies. The viability of *T. elata* as a biocontrol agent of pestiferous slugs used either in isolation or in combination with another commercially available biocontrol agent of pestiferous slugs is discussed. Secondly, since sciomyzid and syrphid species are known indicators of dipteran diversity (Williams et al., 2009ba & b; Knutson & Vala, 2011; Maher et al., 2014; Carey et al., 2017a & b), the contribution of a range of linear farm habitats to their respective abundances and diversity is discussed in the context of farm-based assessments for agri-environmental payment schemes. Finally, Syrphidae species have multiple ecosystem services which include not only pollination but also pest control and nutrient cycling services in the larval stages (Dunn et al., 2020; Speight, 2020) in addition to which they have been considered as environmental and functional bioindicators (Sommaggio, 1999; Sommaggio & Burgio, 2014; Carey et al., 2017a). The value of features for the maintenance of these ecosystem functions at both local and landscape level, with a view to their incorporation in rapid assessments for agri-environmental payment schemes, is also examined. The habitat requirements of hoverflies and their functional groups and how that fits with the habitat requirements of Sciomyzidae are also explored.

The overall findings of this thesis are important in the context of raising concerns about conservation of not only pollinators but other important and less studied taxa (and/or their functional groups) that deliver multiple ecosystem services (i.e., pest control and nutrient cycling). A roadmap for developing measures to conserve and enhance wider biodiversity and their associated multiple ecosystem services on farmland at both local and landscape scales is presented. The major findings of this thesis and their implications are summarised and discussed below:

5.2. Viability of *T. elata* as a biocontrol agent of pestiferous slugs

For an invertebrate to be an effective biocontrol agent (BCA), its biology and ecology need to be fully understood (see Bielza et al., 2020). While recent studies have contributed significantly to our knowledge of the biology and ecology of *T. elata* (Hynes et al., 2014a, b & c; Bistline-East et al., 2018; Bistline-East et al., 2020a & b), there are still gaps in the knowledge regarding:

1- Adult egg laying potential (important for mass rearing);

2- Neonate larval survival in the absence of suitable prey;

3- Host range of neonate larvae;

4- Prey location mechanisms of parasitoid neonate and predatory third instar larvae

5- Combined effects using *T. elata* with a commercially available biocontrol agent (Nemaslug©) which is used on farmland for pestiferous slug control

5.2.1. Egg laying potential of T. elata: Implications for mass rearing

Overall, the number of eggs laid by adult *T. elata* varies for both laboratory-reared and field-caught females with the latter generally shown to lay fewer eggs than the laboratory-reared females. This could be due to the fact that some of these females commence egg laying in the field prior to capture (Knutson et al., 1965). There are also a range of other contributing factors affecting *T. elata* oviposition rates in field caught females such as the habitat type in which they were captured and/or environmental stresses experienced in the field compared to more regulated laboratory conditions. These may include fluctuating temperatures under field conditions or changes in humidity or nutrition quality/quantity.

While the optimum diet for maximum egg production by *T. elata* has not yet been fully determined, *T. elata* adults have been reared successfully under laboratory conditions by feeding them on a 3:1 mixture of honey to brewer's yeast or on aphid produced honeydew which is likely to be an adult food source in the wild (Knutson et al., 1965; Bistline-East et al., 2020). In this study, a laboratory reared *T. elata* adult (fed on a 3:1 mixture of honey to brewer's yeast) produced a maximum of 487 eggs, more than 27% higher than previously maximum recorded number (Knutson et al., 1965). This gives an indication of the potential maximum number of eggs that could be laid by female *T. elata* and has important implications for mass rearing of *T. elata* for biocontrol by augmentation trials. In addition, the relatively high maximum observed egg hatch of 86%, as observed by Beaver (1973) compared to a maximum of only 52% by Hynes et al. (2014b) is likely explained by the conditions under which the eggs were maintained in each study. Knutson et al. (1965) and Beaver (1973) both maintained eggs in rearing jars at 19 - 24 °C and 20 - 25 °C, respectively with 1.27cm of dampened moss at the base. However, Hynes et al. (2014b) maintained *T. elata* eggs in 1cm (approx.) of water in square petri dishes which, while proven as suitable conditions for eggs for the more aquatic sciomyzid species, may not suit the fully terrestrial *T. elata* which has no aquatic larval stages (Knutson & Vala, 2011).

5.2.2. Larval feeding strategy and survival to pupariation

One of the factors that makes an arthropod biocontrol agent effective is its ability to survive during periods of low prey availability or prey absence. In addition, acceptance of less preferred prey, particularly in the field when the essential food source is lacking or scarce, is an important factor that increases the survival of a biocontrol agent despite these alternative food sources often being less nutritious than essential prey (Bielza et al., 2020). Knutson et al. (1965) were the first authors to report that neonate *T. elata* larvae can demonstrate a waiting strategy by staying motionless near the empty egg membranes, until touched by a passing slug host. Similarly, in this study, more than 50% of neonate larvae showed a waiting strategy.

While neonate *T. elata* is known to be host specific to *Deroceras reticulatum* and *Deroceras laeve* only (Knutson et al., 1965), the first evidence that neonate larvae can also feed on another pestiferous slug species (i.e., *Deroceras invadens*) has been presented in this study. However, we found that larval survivorship to pupariation was significantly reduced, and larval development was considerably longer when *D. invadens* was the host compared to when *D. reticulatum* was the host. Neonate *T. elata* larvae appear to use *D. reticulatum* as an "essential" host with all larvae feeding on this slug species in this study pupariating successfully. However, the ability of *T. elata* to feed (albeit less successfully) on alternative food sources when the "essential" host is not present, makes *T. elata* a potentially good biocontrol agent against pestiferous slugs. Nevertheless, further research is needed to confirm the food choice of neonate *T. elata* on different pestiferous slug species and determine whether these are due to differences in nutritional value, in particular slug mucus which is the main *T. elata* food source during the first instar and the early part of the second instar (Knutson et al., 1965).

Another important factor for a viable arthropod biological control agent is its potential foraging/search capacity to find its host (Bielza et al., 2020). While third instar *T. elata* feed as predators (Hynes et al., 2014a & b), the majority (i.e., >60%) of *T. elata* larvae have been shown to use an active searching and attacking strategy without the requirement for previous contact with a slug (Hynes et al., 2014a). However, since low number of replicates (n=8) were used in Hynes et al., (2014a), additional experiments were undertaken in this study (n=27) using a similar approach to deliver robust conclusions. The results of this study reflect those of Hynes et al. (2014a) where almost 60% of third instar larvae showed a searching and attacking strategy further indicating their potential viability as biocontrol agent against pestiferous slugs.

Overall, the maximum egg laying potential by laboratory reared *T. elata* as recorded in this study (487eggs); the maximum observed hatch rate at 86% (Beaver, 1973); all larval stages surviving to pupariation as recorded in this study when fed control *D. reticulatum*; and the maximum recorded number of eleven slugs killed (Knutson et al., 1965) results in the theoretical potential to remove 4,607 individual slugs from an environment under laboratory conditions. However, given that field conditions will be less optimal than laboratory conditions, the total number of slugs killed will be lower due to a range of factors including prey availability and mortality due to predation/disease as well as a range of sub-optimal habitat conditions. While the overall results from the laboratory-based experiments are promising, further research is needed to investigate correlations between *D. reticulatum* abundance and abundance of adult *T. elata* under field conditions to fully determine efficacy of *T. elata* as a biological control agent of pestiferous slugs in the field. In addition, determining best habitats in the field that can sustain *T. elata* abundance which predate pestiferous slugs will support an important ecosystem function in the context of the EU recent banning of methiocarb slug pellets. In this regard, it is interesting to note that the presence of *T. elata* in our study (and other Sciomyzidae) was associated with open hedgerows adjacent to watercourses and showed a significant positive correlation with grassland habitat quality score, in particular good vegetation structure in adjacent grassland fields (i.e., >50% of the field having a heterogeneous vegetation structure). In addition, 95% of adult *T. elata* were found in sites with soil moisture above 30% while only 5% of adult *T. elata* were found in sites with soil moistures below 30%. This is likely due to sites with higher moistures having greater amounts of decaying vegetation that provide desirable shelter and alternative food sources for their prey *D. reticulatum* (Bistline-East et al., 2020). Nevertheless, these findings support the study by Bistline-East et al. (2020) which found that adult *T. elata* is associated with the proximity of hedgerows (near a river) and tall, dead vegetation (likely to be associated with very low stocking densities). This is further supported by previous studies where Sciomyzidae (including *T. elata* as well as other dipteran families of grassland) have been shown to demonstrate positive correlations with vegetation height (Ryder et al., 2005; Williams et al., 2009a; Maher et al., 2014). In this study, heterogeneous vegetation structure is likely to be a result of the less intensively managed, wetter fields carrying lower stocking densities than the more improved fields with drier soils.

5.2.3. Effect of Phasmarhabditis hermaphrodita on the viability of T. elata as a biocontrol agent

Phasmarhabditis hermaphrodita is the only commercial nematode biocontrol agent available in the European market (under the trade name Nemaslug®). However, it has been shown to have a limited success in controlling pestiferous slugs under laboratory conditions with approximately 60% success rate when applied onto the body of the slug *Deroceras reticulatum* (Dankowska, 2006). In addition, its success rate on other pestiferous slug species is lower (Dankowska, 2006) and is likely to be lower under field conditions. While *T. elata* shows potential as an alternative biocontrol agent against pestiferous slugs, determining the potential of the two biocontrol agents to be used synergistically for pest control is of great interest in the context of maximizing biological control of pestiferous slugs.

In this regard, this study is the first to investigate the effects of pestiferous slugs which are exposed to *P. hermaphrodita* on the survival of *T. elata* larvae under laboratory conditions. In the no-food choice experiments, 70% of neonate larvae reached third instar when fed continuously on nematode-exposed slugs but with only 20% pupariating successfully compared to 100% pupariation using control slugs. Similarly, in the food choice experiments, more than 70% of third instar larvae initially selected nematode-exposed slugs with 75% surviving to pupariation when fed continuously on nematode exposed slugs compared to 100% pupariation using control slugs. This study also presents the first evidence that continuous feeding of neonate *T. elata* larvae on nematode-exposed slugs reduces the chances of survival to pupariation to below 20%. However, further research is needed to determine the effects of *P.*

hermaphrodita on the nutritional quality of nematode-exposed slugs and how that affects the larval stages of *T. elata* and their survival to pupation.

While both *P. hermaphrodita* and *T. elata* have a biocontrol potential against pestiferous slugs, in particular *D. reticulatum*, careful consideration should be taken when using *P. hermaphrodita* in areas where *T. elata* is naturally occurring. To minimize any adverse effects on biodiversity, it might be advisable at present, to not apply *P. hermaphrodita* in areas and during the times where *T. elata* larvae are active in the field (May to September; Knutson et al., 1965). Nevertheless, further research is needed on both biocontrol agents to fully determine the synergistic or antagonistic effects of both biocontrol agents when used in combination against pestiferous slugs.

5.3. Contribution of farm habitats to dipteran abundance and diversity: one size does not fit all

Currently, dense continuous hedgerows that are diverse in woody species and floral resources are generally considered to be 'good quality' linear habitats that can sustain pollinators with strong mobility such as bees (Garratt et al., 2017; Volpato et al., 2020) and hoverflies (Garratt et al., 2017). However, their value for pest control species with lower mobility (such as sedentary sciomyzid flies and hymenopteran parasitoids) is less well known (Williams et al., 2010; Dover, 2019). Therefore, the "one size fits all" approach for hedgerow quality (i.e., dense, diverse) might not fulfil the habitat requirements of certain invertebrate species that deliver important pest control services. This "one size fits all" approach is a matter of concern when designing AE schemes to protect wider biodiversity, since advice on best practice for hedgerow maintenance recommends keeping the shrub layer dense (Hedgelink leaflet, 2013 - www.hedgelink.org.uk) and hedgerows are often assessed as being less favourable based on increased "gappiness" (Foulkes et al., 2013). To explore this more fully, two dipteran families, Sciomyzidae and Syrphidae, were selected for further study. Both taxa deliver different ecosystem services, and are known as good indicators of farm habitat types and wider dipteran diversity in wet grassland habitats (Carey et al., 2017a).

The two taxa showed different responses to linear farm habitats, in particular the family Sciomyzidae which were significantly more abundant and species rich in open hedgerows with adjacent watercourses in comparison with dense hedgerows only (i.e., without watercourses) or open hedgerows only. While the results have been discussed in detail in Chapter 3, the major finding is that the “one size fits all” approach of agri-environmental schemes does not appear to cater for all invertebrate species, in particular sedentary species such as Sciomyzidae which are likely to be negatively affected by encouraging farmers to maintain intact and thick hedgerows without any gaps. Therefore, to maximize dipteran diversity and their multiple ecosystem services, management requirements of linear habitats such as hedgerows should reflect not only the needs of pollinator species with strong mobility, but also the needs of wider biodiversity associated with hedgerows including species with pest control services. Moreover, in the context of future AE schemes under the new EU common agricultural policy (post 2023), it is likely that considering a range of linear habitats on the farm (i.e., linear habitat heterogeneity within and between habitats), with various management options, could be a potential measure to maximize wider biodiversity on farmland since different taxa may have heterogeneous requirements for different habitat types (Graham et al., 2018; Dover, 2019). However, spatial scales and management heterogeneity over time and space should also be taken into consideration since they play a significant role in the way how we assess farmland biodiversity (Carey et al., 2017b; Graham et al., 2018).

5.4. Hedgerows and treelines promote hoverflies with dual ecosystem services at landscape scale: implications for agri-environment schemes

While Chapter 3 showed evidence that different linear habitats, and variability within habitats, in particular hedgerows (including less favourable, gappy hedgerows), can be important for sustaining dipteran abundance and diversity with multiple ecosystem services, the implications of this for a national agri-enviroment scheme is worth further investigation. This can play a significant role in the identification of biodiversity hotspots on farmland at larger scales using high resolution satellite imagery rather than time consuming, extensive field surveys.

With this in mind, Chapter 4 further investigates the effect of habitat quality at local scale and different linear habitats (quantity and diversity) at landscape scale on hoverflies and their functional groups within two contrasting study regions in Ireland (i.e., Co. Sligo and Co. Wexford). We selected hoverflies since they deliver multiple ecosystem services (Dunn et al., 2020) and reflect biodiversity at both local and/or landscape scale (Power et al., 2016; Moquet et al., 2018; Schirmel et al., 2018). Interestingly, length of hedgerows and treelines separately were the most important driving factors of hoverfly abundance/richness at landscape scale showing a significant positive effect on the abundance and/or species richness of hoverflies and their functional groups regardless of farming conditions and study regions. While hoverflies are known to reflect wider dipteran diversity at least in wet grasslands (Carey et al., 2017a), the implications of this study are particularly important in developing measures at larger scales (targeted using high resolution satellite imagery) to support invertebrates with multiple ecosystem services. Therefore, when evaluating different linear farm habitats for biodiversity at local and landscape scale, length of hedgerows (and treelines) appears to be the most important explanatory variables that can reflect wider dipteran diversity on farmland. This in agreement with previous studies who have shown the value of woody semi-natural habitat features for biodiversity (Garratt et al., 2017; Schirmel et al., 2018; Volpato et al., 2020).

Given that the value of other linear habitat features such as dry-stonewalls (BL1) and drainage ditches (FW4) varied between the two study regions (i.e., Co. Sligo and Co. Wexford), using these habitat features to quantify biodiversity at a national scale does not appear to be a suitable measure at present (at least in this study). In addition, neither do the combined length of different linear features and linear feature richness appear to be a suitable measure to quantify biodiversity at a national scale since the value of certain habitat features (i.e., non-woody) varied between the two study regions. Since these non-woody linear habitat features (BL1& FW4) showed a significant positive effect on the abundance of hoverflies and their functional groups in Co. Sligo, they are likely to be more suitable when assessing farm habitats at county level rather than a national level. The study by Kleijn and Van Langevelde (2006) showed that the species richness and abundance of hoverflies were not significantly correlated with drainage ditch length and flower abundance at any spatial scale (500-1000m) in intensively managed grassland and arable lands in the Netherlands. However, others have shown that excessive input of agrochemicals change drainage ditch habitat quality from a mesotrophic system containing a species-rich mosaic of submerged, emergent and floating plant species to a hypertrophic state dominated by Lemnaceae or phytoplankton (Verdonschot et al., 2012). Given that it is difficult to determine the habitat quality score of these features at landscape scale using satellite images, more research at local scale which are based on field survey is needed to better assess the value of these non-woody habitat features for biodiversity. This is particularly important across different farming intensities and study regions which may have important implications for quantifying different invertebrates with multiple ecosystem services at landscape scale including pollinators, biological pestcontrol and nutrient cycling services. Studying the effect of other quality metrics such as those assessed by Rotchés-Ribalta et al. (2020) which includes not only floral resources but also other local structural and ecological elements such as number and cover of positive/negative plant indicator species, linear feature shape, height, and width can help to deepen our understanding of the values of these non-woody habitat features for biodiversity.

5.5. General Conclusions

- *Tetanocera elata* shows potential as a viable biological agent against pestiferous slugs with a maximum egg laying of 487 eggs/individual female under laboratory conditions and room temperatures of 19-20°C.
- Neonate *T. elata* larvae can survive in the absence of their first prey for up to two weeks and can feed on another pestiferous slug (i.e., *Deroceras invadens*) but have a greater chance of survival when they feed on their essential slug prey *Deroceras reticulatum* throughout larval stage.
- Most neonate larvae show a waiting strategy to find their prey while most third instar larvae use an active searching and attacking strategy to find and kill it is prey.
- Neonate and third instar larvae of *T. elata* have reduced chances of survival to pupariation when feeding on the alternative prey *D. invadens* or nematode-exposed slugs while all larvae can survive to pupariation when feeding on nematode-free *D. reticulatum*.
- Linear habitat heterogeneity can play a key role for sustaining wider biodiversity including species with pollination and pest control services However, habitat requirements of Sciomyzidae (including *T. elata*) differ from those of Syrphidae in relation to linear habitat types and the "one size fits all" approach such as dense and thick hedgerows does not appear to benefit Sciomyzidae that deliver important pest control services which are not provided by hoverflies and bees.
- Hedgerows and treelines at landscape scale are the most important driving factors positively affecting abundance/richness of hoverflies and their associated functional groups (regardless of farming conditions and study regions) and have important implications for biodiversity and ecosystem services at a national scale.
- Other linear habitat features such as dry-stonewalls and drainage ditches show positive/negative effects on hoverflies and their functional groups abundance/richness at landscape scale across different study regions and may

be more appropriate for use as explanatory variables at county level rather than a national scale.

- Abundance, species richness and community composition of Sciomyzidae and Syrphidae reflect low intensity farming practices and high habitat quality with significantly more species of both families in extensive farms when using Malaise traps as a sampling strategy.

5.6. Recommendations for future research

- To fully assess *Tetanocera elata* as a viable biocontrol agent against pestiferous slugs, further research is required particularly regarding ways of maximising egg laying and hatching rates and increasing larval survival at each instar under different laboratory conditions.

- Further research is needed to confirm whether the food choice of neonate *T. elata* on different slug species are due to differences in nutritional value in particular slug mucus which is the main *T. elata* food source during the first instar and the early part of the second instar.

- While research regarding larval development and survivorship at different constant temperatures has been undertaken, additional research regarding maximising hatch and survival rates at varying temperatures should be undertaken.

- Further research regarding interactions between *T. elata* larvae and parasitic nematodes should be undertaken to better understand how, if possible, they can be used in tandem as a biological mollusc control method.

- While there is a variety of information regarding the biology of sciomyzid flies that predate on and/or parasitise molluscs, the understanding of their role in mollusc population dynamics is still relatively limited and future studies in the field should assess the correlations between sciomyzidae abundance/richness and their mollusc hosts.

- While the value of different linear farm habitats varied between Sciomyzidae and Syrphidae at local scale, similar research but with more replicates are needed to fully assess the value of different linear farm habitats for wider biodiversity and their associated ecosystem services particularly in intensive farms and across different study regions.

- Hedgerows in general and treelines appear to support hoverflies with dual ecosystem services at landscape scale but further research using various spatial scales, and other important taxa (bees and other understudied dipteran taxa) is needed to confirm their value for wider biodiversity and their associated ecosystem services.

- Given that the 2018 summer heatwave impacted on Syrphidae, and the subsequent impact on the Malaise trapping, comparing Malaise and pan traps within the same time period under the same site conditions would be worthwhile to quantify the effectiveness of Malaise and pan traps for Syrphidae assessment. This has important implications for a more reliable assessment of farm habitats.

- While only few aphid individuals were observed when using pan traps in our study, future research into the correlation between aphid abundance and the visitation of Syrphidae at pan traps could prove insightful into pan trap effectiveness.

- Future studies should further investigate the impact that floral communities have on the attractiveness to Syrphidae of pan traps across different farming intensities. This has important implications for selecting reliable sampling strategies for rapid assessment of farm habitats that can be incorporated in result-based agri-environment schemes.

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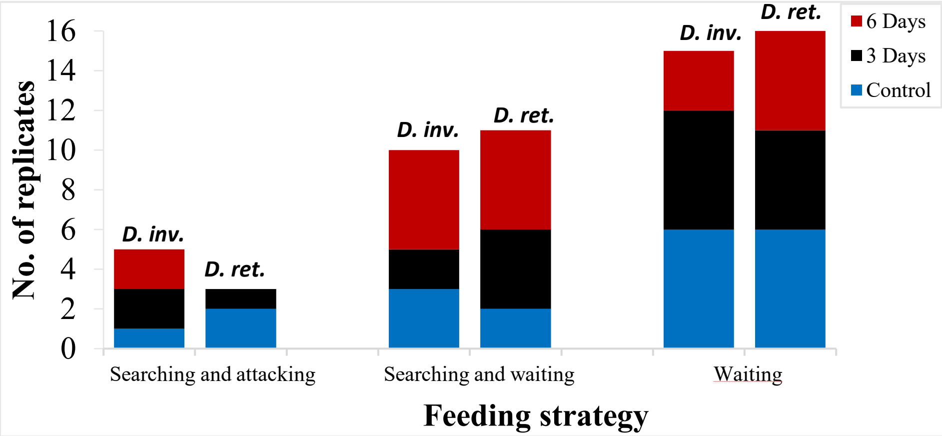
Appendices

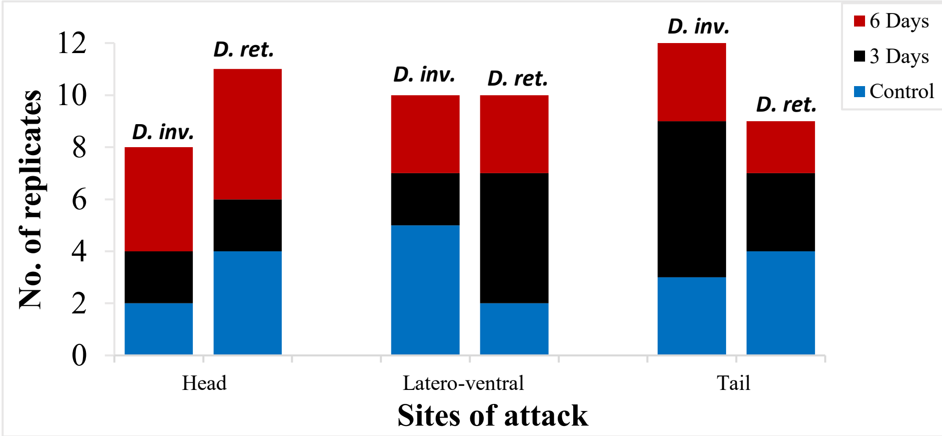
Appendix I Supplemental Information for Chapter 2

Appendix I.1

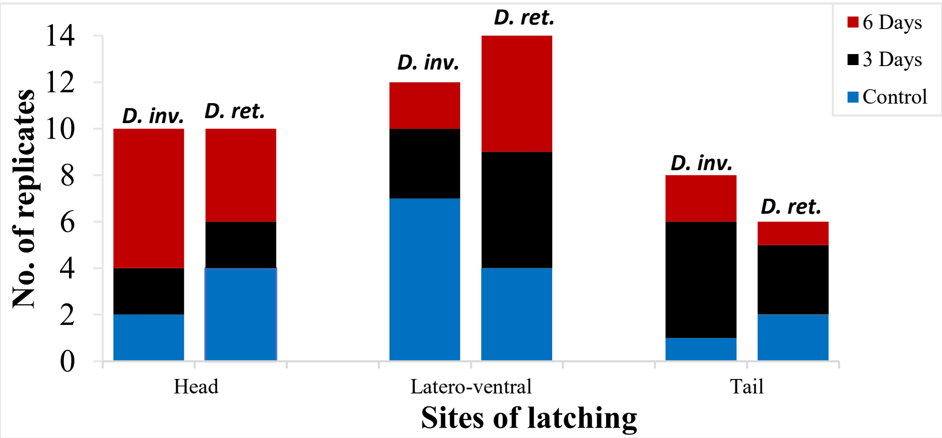
Tetanocera elata neonate Larval behavior.



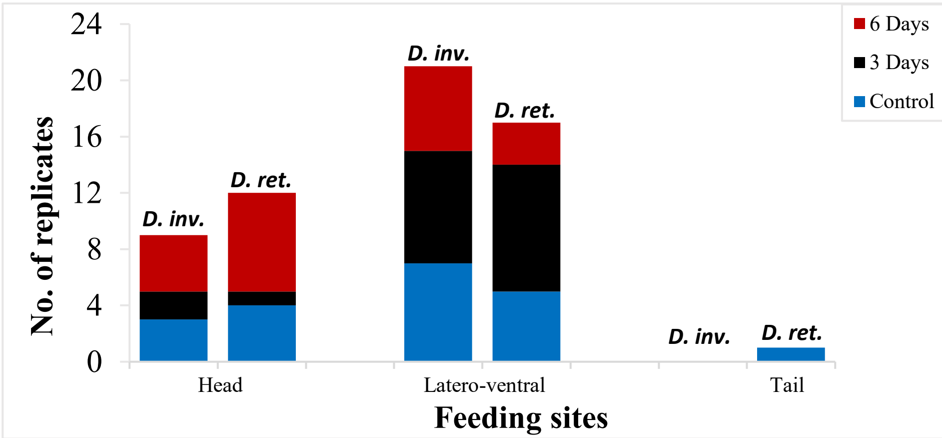
a) Feeding strategy of neonate *T. elata* larvae on both slug species



b) Sites of neonate *T. elata* larvae attack on both slug species



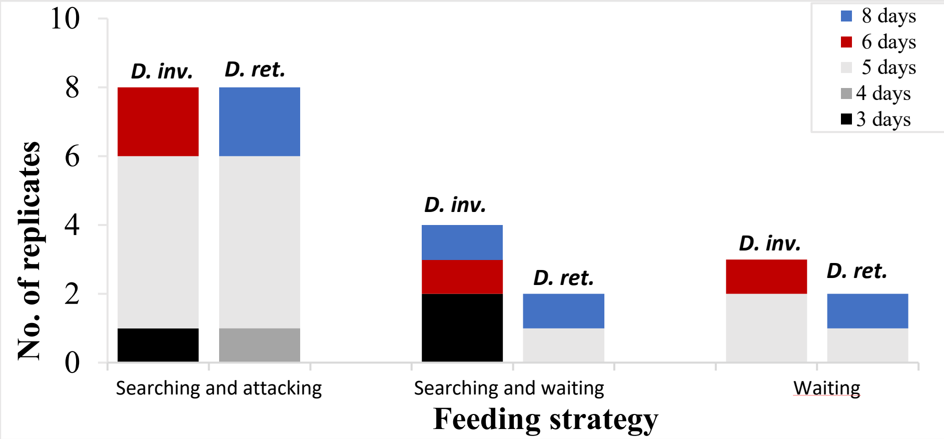
c) Latching sites on both slug species by neonate *T. elata* larvae



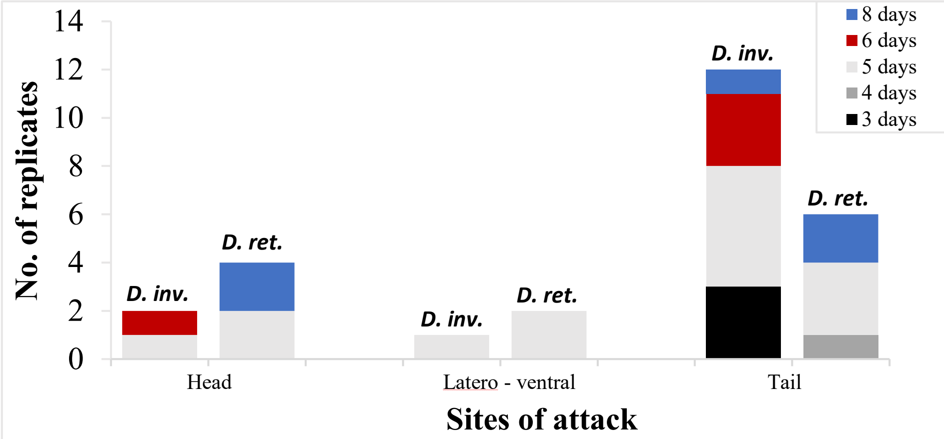
d) Feeding sites by neonate *T. elata* larvae on both slug species

Appendix I.1 – Continued

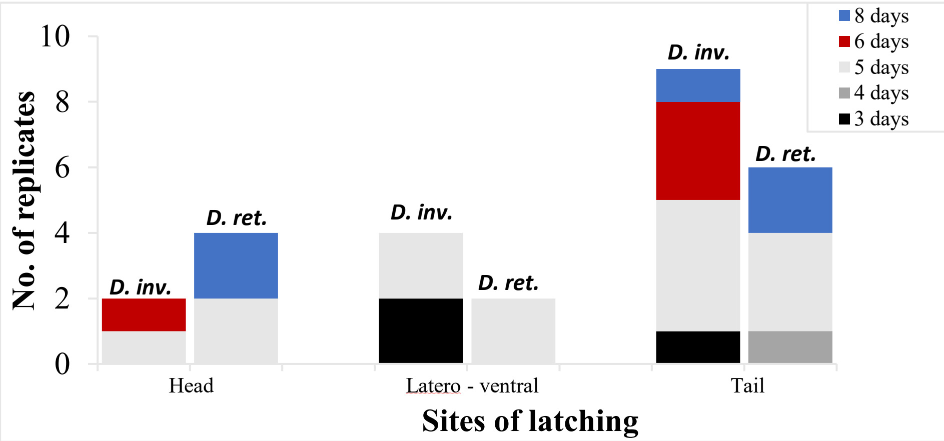
T. elata third instar larval behavior.



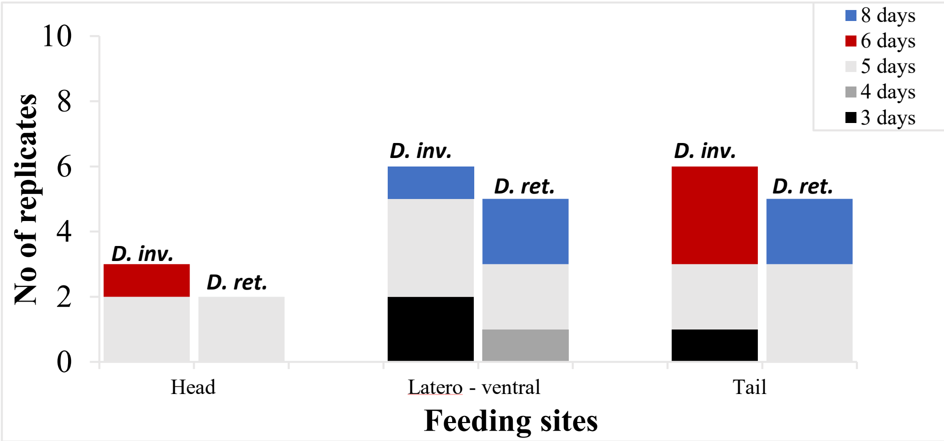
e) Feeding strategy of third instar T. elata on both slug species



f) Sites of third instar T. elata attack on both slug species



g) Latching sites on both slug species by third instar T. elata larvae

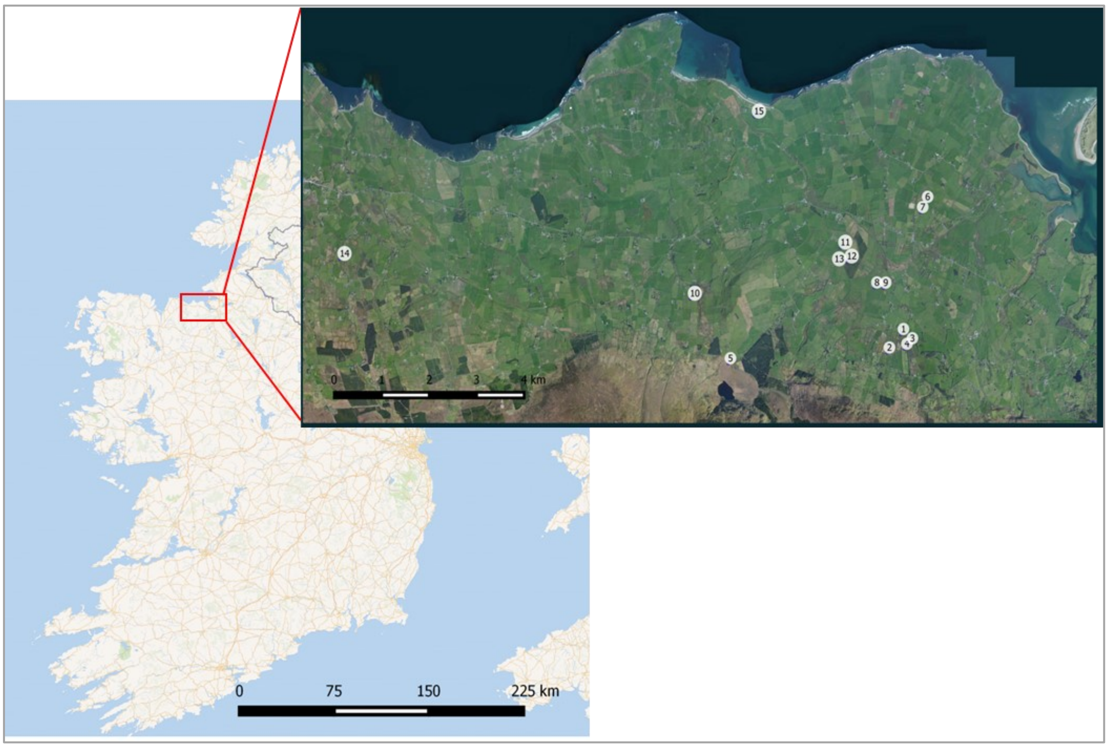


h) Feeding sites by third instar T. elata larvae on both slug species

Appendix II Supplemental Information for Chapter 3

Appendix II.1

A map showing location of the study sites in County Sligo, Northwest of Ireland (left). Farming intensity categories are denoted with different numbers (right). Extensive = 1-5; Intermediate = 6 -10; and Intensive = 11-15.



Appendix II.2

Site description and classification based on farming intensity (extensive, intermediate, intensive) and linear habitat types[1]. The overall habitat quality scores (0 - 1) are calculated for both linear and adjacent grassland habitats based on several variables including physical structure (e.g., width, height), vegetation structure (profile, how many layers of vegetation, cover of trees, shrubs), management pressure (grazing pressure, poaching) and number and cover of plant species indicators.

Site no.	Parcel area (ha)	LU/ha	NV score	Linear habitat type	Linear habitat score	Grassland habitat score
Extensive						
1	18.3	0.59	6.9	DH	0.37	0.65
2	3.12	0.24	7.5	DHW	0.55	0.80
3	3.91	0.24	7.5	OH	0.36	0.70
4	7.83	0.59	6.9	OHW	0.32	0.60
5	45.05	0.36	8.2	W	0.11	0.70
Intermediate						
6	12.31	1.18	3.8	DH	0.51	0.00
7	7.73	0.78	4.1	DHW	0.62	0.50
8	7.47	1	4.6	OH	0.43	0.40
9	1.95	1.18	3.8	OHW	0.66	0.15
10	15.61	0.74	3.9	W	0.73	0.95
Intensive						
11	3.05	0.75	3.4	DH	0.39	0.00
12	10.4	0.75	3.4	DHW	0.71	0.60
13	9.22	1.02	3.3	OH	0.48	0.15
14	5.38	1.11	3.3	OHW	0.51	0.35
15	10.4	0.75	3.4	W	0.32	0.60

[1]Linear habitat types are categorised as: Dense hedgerow (DH); Open hedgerow (OH); Dense hedgerow with adjacent watercourse (DHW); Open hedgerow with adjacent watercourse (OHW); and watercourse only (W).

Appendix II.3

Bi-directional Malaise traps used throughout the study.

Appendix II.4

Syrphidae species recorded on farms in Co. Sligo, Ireland.

Species	Total Abundance	% Total Abundance
Anasimyia contracta Claussen & Torp, 1980	2	0.02
Anasimyia lineata (Fabricius, 1787)	8	0.09
Arctophila superbiens (Müller, 1776)	3	0.03
Baccha elongata (Fabricius, 1775)	29	0.33
Chalcosyrphus nemorum (Fabricius, 1805)	18	0.21
Cheilosia albipila Meigen, 1838	1	0.01
Cheilosia albitarsis (Meigen, 1822)	14	0.16
*Cheilosia sp**	1	0.01
Chrysogaster cemiteriorum (Linnaeus, 1758)	3	0.03
Chrysotoxum bicinctum (Linnaeus, 1758)	28	0.32
Chrysotoxum festivum (Linnaeus, 1758)	1	0.01
Dasysyrphus albostriatus (Fallén, 1817)	1	0.01
Dasysyrphus venustus (Meigen, 1822)	4	0.05
Epistrope eligans (Harris, 1780)	50	0.57
Epistrophe nitidicollis (Meigen, 1822)	1	0.01
Episyrphus balteatus (De Geer, 1776)	183	2.09
Eristalis abusiva Collin, 1931	3	0.03
Eristalis arbustorum (Linnaeus, 1758)	6	0.07
Eristalis horticola (De Geer, 1776)	11	0.13
Eristalis intricaria (Linnaeus, 1758)	41	0.47
Eristalis nemorum (Linnaeus, 1758)	41	0.47
Eristalis pertinax (Scopoli, 1763)	94	1.07
Eristalis tenax (Linnaeus, 1758)	16	0.18
Eupeodes corollae (Fabricius, 1794)	342	3.90
Eupeodes latifasciatus (Macquart, 1829)	889	10.13
Eupeodes luniger (Meigen, 1822)	12	0.14
Helophilus hybridus Loew, 1846	41	0.47
Helophilus pendulus (Linnaeus, 1758)	1160	13.22
Lejogaster metallina (Fabricius, 1781)	83	0.95
Leucozona lucorum (Linnaeus, 1758)	9	0.10
Melangyna lasiopthalma (Zetterstedt, 1843)	6	0.07
*Melangyna spp**	4	0.05
Melanogaster hirtella (Loew, 1843)	11	0.13
*Melanostoma [melanic]**	10	0.11
Melanostoma mellinum (Linnaeus, 1758)	679	7.74
*Melanostoma scalare (*Fabricius, 1794)	778	8.87

*Meligramma sp**	1	0.01
Meliscaeva cinctella (Zetterstedt, 1843)	3	0.03
Meliscaeva auricollis (Meigen, 1822)	1	0.01
Myathropa florea (Linnaeus, 1758)	2	0.02
Neoascia obliqua Coe, 1940	3	0.03
Neoascia podagrica (Fabricius, 1775)	62	0.71
Neoascia tenur (Harris 1780)	15	0.17
Orthonevra nobilis (Fallén, 1817)	1	0.01
Parasyrphus punctulatus (Verrall, 1873)	2	0.02
Parhelophilus versicolor (Fabricius, 1794)	1	0.01
*Pipiza sp**	1	0.01
Pipiza noctilucaa (Linnaeus, 1758)	3	0.03
*Platycheirus [melanic]**	2	0.02
Platycheirus albimanus (Fabricius, 1781)	421	4.80
Platycheirus angustatus (Zetterstedt, 1843)	238	2.71
Platycheirus clypteatus (Meigen, 1822)	1071	12.21
Platycheirus granditarsus (Forster, 1771)	939	10.70
Platycheirus manicatus (Meigen, 1822)	1	0.01
Platycheirus peltatus (Meigen, 1822)	13	0.15
Platycheirus rosarum (Fabricius, 1787)	94	1.07
Platycheirus scambus (Staeger, 1843)	1	0.01
Platycheirus scutatus (Meigen, 1822)	14	0.16
Rhinga campestris Meigen, 1822	698	7.96
Riponnensia splendens (Meigen, 1822)	21	0.24
Scaeva pyrastri (Linnaeus, 1758)	50	0.57
Sericomyia silentis (Harris, 1776)	212	2.42
Sphaerophoria interrupta (Fabricius, 1805)	38	0.43
Sphaerophoria scripta (Linnaeus, 1758)	5	0.06
Sphaerophoria philanthus (Meigen, 1822)	4	0.05
Syritta pipiens (Linnaeus, 1758)	3	0.03
Syrphus torvus Osten Sacken, 1875	1	0.01
Syrphus ribesii (Linnaeus, 1758)	48	0.55
Syrphus vitripennis Meigen, 1822	4	0.05
Trichopsomyia flavitarsis (Meigen, 1822)	20	0.23
Tropidia scita (Harris, 1780)	142	1.62
Volucella bombylans (Linnaeus, 1758)	17	0.19
Volucella pellucens (Linnaeus, 1758)	10	0.11
Xylota jakutorum Bagachanova, 1980	3	0.03
Xylota segnis (Linnaeus, 1758)	21	0.24
Xylota sylvarum (Linnaeus, 1758)	5	0.06

* Identified to genus only since females could not be identified or distinguished from each other.

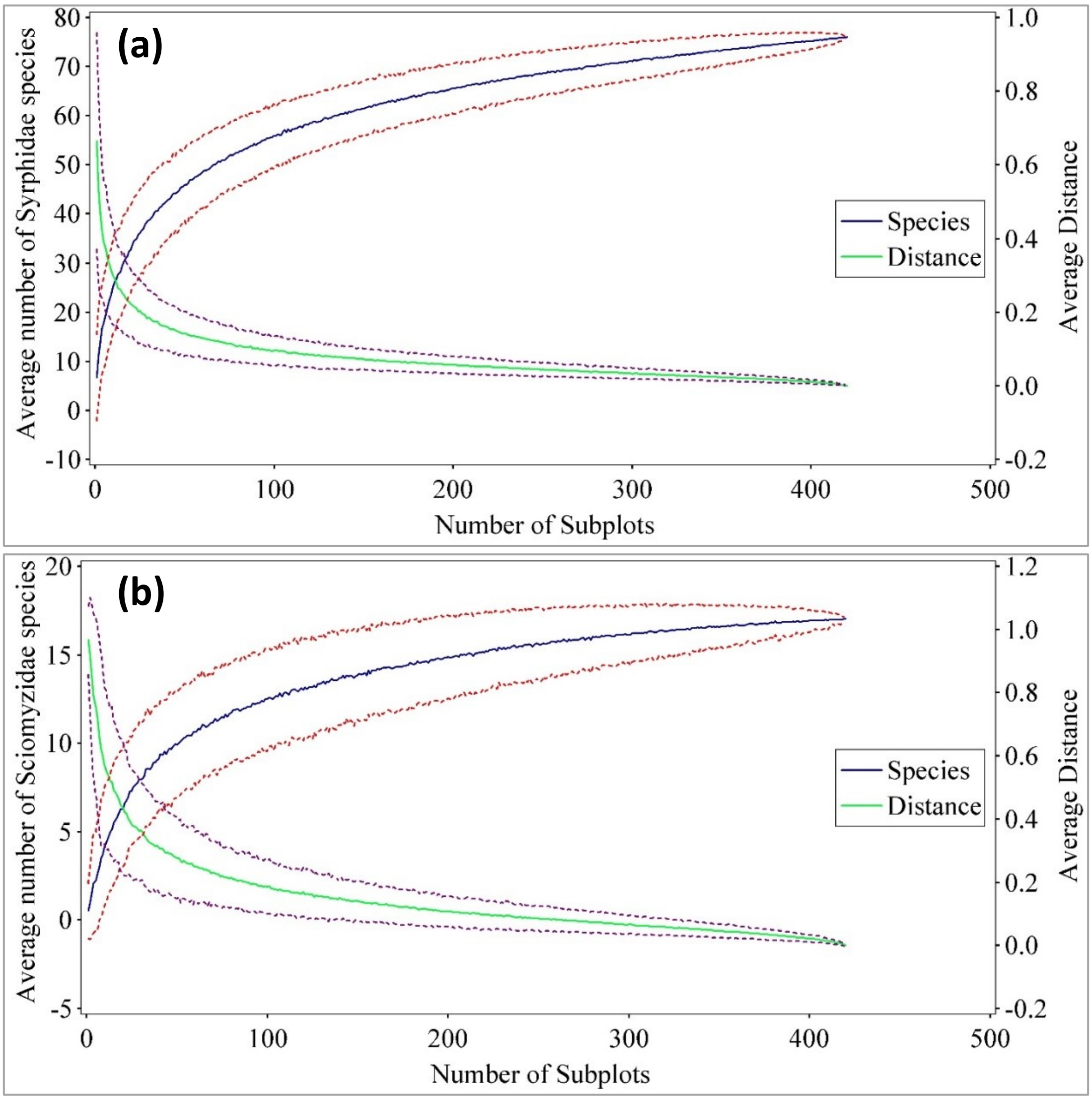
Appendix II.5

Sciomyzidae species recorded on farms in Co. Sligo, Ireland.

Species name	Total abundance	% Total abundance
Coremacera marginata (Fabricius, 1775)	11	4.0
Elgiva cucularia (Linnaeus, 1767)	1	0.4
Ilione albiseta (Scopoli, 1763)	4	1.5
Ilione lineata (Fallen, 1820)	19	7.0
Limnia paludicola Elberg, 1965	4	1.5
Limnia unguicornis (Scopoli, 1763)	2	0.7
Pherbina coryleti (Scopoli, 1763)	4	1.5
Renocera pallida (Fallén, 1820)	50	18.3
Renocera striata (Meigen, 1830)	2	0.7
Sepedon spinipes (Scopoli, 1763)	1	0.4
Tetanocera arrogans Meigen, 1830	57	20.9
Tetanocera elata (Fabricius, 1781)	40	14.7
Tetanocera ferruginea Fallén, 1820	35	12.8
Tetanocera fuscinervis (Zetterstedt, 1838)	13	4.8
Tetanocera hyalipennis Roser, 1840	11	4.0
Tetanocera robusta Loew, 1847	17	6.2
Trypetoptera punctulata (Scopoli, 1763)	2	0.7

Appendix II.6

Species area curves for Syrphidae (a) and Sciomyzidae (b). Dotted lines represent ±2SDs. First-order jackknife estimates of total species richness were 90.75 (Syrphidae) and 18.9 (Sciomyzidae).



Appendix II.7

P values for the mean species richness and abundance of Sciomyzidae collected for each linear habitat type[1]. Numbers in bold indicate significant P values (GLMM followed by LSD pairwise comparisons, $P < 0.05$).

Boundary type	Sciomyzidae species richness			Sciomyzidae abundance		
	d.f	t	P	d.f.	t	P
DH × DH_W	4	1.40	0.17	4	1.81	0.08
DH × OH	4	0.04	0.97	4	1.03	0.31
DH × OH_W	4	2.56	**0.01**	4	2.83	**0.01**
DH ×W	4	2.51	**0.02**	4	3.22	**<0.001**
DH_W × OH	4	1.44	0.16	4	0.84	0.40
DH_W × OH_W	4	1.19	0.24	4	1.26	0.21
DH_W × W	4	1.14	0.26	4	1.72	0.09
OH × OH_W	4	2.60	**0.01**	4	2.02	**0.04**
OH × W	4	2.54	**0.01**	4	2.45	**0.02**
OH_W × W	4	0.06	0.95	4	0.49	0.63

[1]Linear habitat types are categorised as: Dense hedgerow (DH); Open hedgerow (OH); Dense hedgerow with adjacent watercourse (DH_W); Open hedgerow with adjacent watercourse (OH_W); and watercourse only (W).

Appendix II.8

P values for the mean species richness and abundance of Syrphidae and Sciomyzidae collected per site at each of the three-farming intensities. Numbers in bold indicate significant P values (GLMM followed by LSD pairwise comparisons, P < 0.05).

Parameter	Farming intensities			Linear habitat types		
	d.f	t	P	d.f	t	P
Flowering plant species richness/linear habitat	2	1.86	0.39	4	3.22	0.52
Flowering plant species richness/grassland habitat	2	3.29	0.19	4	2.73	0.60
Linear habitat quality score	2	5.47	0.07	4	3.97	0.41
Grassland habitat quality score	2	5.60	0.06	4	5.48	0.24
% Soil moisture	2	8.66	**0.01**	4	3.60	0.46
% Soil Organic matter	2	7.02	**0.03**	4	4.53	0.34
Soil pH	2	1.82	0.40	4	7.47	0.11

Appendix II.9

A- Results of Kruskall-Wallis test to determine differences between environmental variables across three farming intensities and linear habitat types. Numbers in bold indicate significance.

Farming intensity	Syrphidae					
	Species richness			Abundance		
	d.f	t	P	d.f	t	P
Extensive × Intermediate	2	3.25	**<0.01**	2	1.31	0.20
Extensive × Intensive	2	4.19	**<0.001**	2	1.81	0.08
Intermediate × Intensive	2	0.67	0.51	2	0.53	0.60
Farming intensity	Sciomyzidae					
	Species richness			Abundance		
	d.f	t	P	d.f	t	P
Extensive × Intermediate	2	3.86	**<0.001**	2	4.66	**<0.001**
Extensive × Intensive	2	4.69	**<0.001**	2	5.26	**<0.001**
Intermediate × Intensive	2	0.91	0.37	2	1.13	0.26

B- Pairwise comparisons for % soil moisture and soil organic matter between the three farming intensity categories using Bonferroni correction for multiple comparisons. Numbers in bold indicate significance ($P < 0.05$).

Comparison	**% Soil moisture**		**% Soil organic matter**	
	t	P	t	P
Extensive × Intermediate	7.40	**0.03**	5.40	1.69
Extensive × Intensive	7.00	**0.04**	7.20	**0.03**
Intermediate × intensive	-0.04	1.00	1.80	1.00

Appendix III Supplemental Information for Chapter 4

Appendix III.1

Study sites details for both study regions (1-15= Sligo; 16-30=Wexford). NVS = Nature Value Score per Farm.

Site No.	Site Name	Intensity level	Field Area (ha)	NVS
1	CF4	Intensive	1.77	3.30
2	CB4	Intensive	1.59	3.30
3	GL9	Intensive	1.22	2.20
4	MG	Intensive	1.15	2.40
5	SK	Intensive	0.67	3.3
6	JK	Intermediate	1.16	4.10
7	EA	Intermediate	0.51	3.60
8	MB	Intermediate	1.03	3.80
9	EJ	Intermediate	1.01	4.60
10	EF	Intermediate	1.35	3.60
11	MM	Extensive	1.70	5.90
12	BM	Extensive	1.85	8.70
13	HC	Extensive	1.70	7.50
14	MC	Extensive	1.40	8.20
15	JC	Extensive	0.78	8.40
16	MK_3	Intensive	1.20	NONE
17	MD_17	Intensive	0.70	NONE
18	RK_9	Intensive	3.90	NONE
19	JM3_15	Intensive	2.10	1.90
20	AK_17	Intensive	3.80	2.50
21	WB_8	Intermediate	1.20	4.70
22	NO_5	Intermediate	1.20	4.80
23	MO2_10	Intermediate	1.20	4.20
24	LF_1	Intermediate	1.10	4.30
25	PL_8	Intermediate	2.70	4.30
26	AO_3	Extensive	0.92	5.60
27	KN_5	Extensive	1.10	8.10
28	JG_7	Extensive	1.20	6.30
29	MM_5	Extensive	2.10	5.00
30	MP_4	Extensive	1.80	5.00

Appendix III.2

Abundance of hoverfly species and their functional groups recorded for both study regions.

Species	Sligo	Wexford	Larval food
Chrysotoxum bicinctum (L.), 1758	3	2	Predacious
Dasysyrphus albostriatus (Fallen), 1817	2	2	Predacious
Dasysyrphus venustus (Meigen), 1822	0	1	Predacious
Episyrphus balteatus (De Geer), 1776	9	478	Predacious
Eupeodes corollae (Fabricius), 1794	14	28	Predacious
Eupeodes latifasciatus (Macquart), 1829	1	1	Predacious
Eupeodes luniger (Meigen), 1822	1	0	Predacious
Melangyna lasiophthalma (Zetterstedt), 1843	0	1	Predacious
Melanostoma mellinum (L.), 1758	8	1	Predacious
Melanostoma scalare (Fabricius), 1794	16	9	Predacious
Meliscaeva auricollis (Meigen), 1822	4	5	Predacious
Meliscaeva cinctella (Zetterstedt), 1843	4	0	Predacious
Platycheirus albimanus (Fabricius), 1781	34	13	Predacious
Platycheirus clypeatus (Meigen), 1822	1	3	Predacious
Platycheirus granditarsis (Forster), 1771	22	2	Predacious
Platycheirus manicatus (Meigen), 1822	1	1	Predacious
Platycheirus peltatus (Meigen), 1822	0	47	Predacious
Platycheirus scutatus (Meigen), 1822	2	0	Predacious
Sphaerophoria interrupta (Fabricius), 1805	1	0	Predacious
Sphaerophoria scripta (L.), 1758	0	5	Predacious
Syrphus ribesii (L.), 1758	60	30	Predacious
Syrphus torvus Osten-Sacken, 1875	13	1	Predacious
Syrphus vitripennis Meigen, 1822	2	4	Predacious
Volucella bombylans (L.), 1758	0	2	Predacious
Volucella pellucens (L.), 1758	0	1	Predacious
Chrysogaster solstitialis (Fallen), 1817	1	0	Non-Predacious
Eristalinus sepulchralis (L.), 1758	0	1	Non-Predacious
Eristalis arbustorum (L.), 1758	14	5	Non-Predacious
Eristalis horticola (De Geer), 1776	4	0	Non-Predacious
Eristalis intricaria (L.), 1758	3	0	Non-Predacious
Eristalis nemorum (L.), 1758	6	3	Non-Predacious
Eristalis pertinax (Scopoli), 1763	7	9	Non-Predacious
Eristalis tenax (L.), 1758	13	28	Non-Predacious
Ferdinandea cuprea (Scopoli), 1763	2	0	Non-Predacious
Helophilus hybridus Loew, 1846	2	0	Non-Predacious
Helophilus pendulus (L.), 1758	48	6	Non-Predacious
Lejogaster metallina (Fabricius), 1781	3	0	Non-Predacious

Melanogaster hirtella (Loew), 1843	41	4	Non-Predacious
Neoascia podagrica (Fabricius), 1775	14	1	Non-Predacious
Rhingia campestris Meigen, 1822	45	1	Non-Predacious
Riponnensia splendens (Meigen), 1822	3	0	Non-Predacious
Sericomyia silentis (Harris), 1776	17	1	Non-Predacious
Sericomyia superbiens (Muller), 1776	5	0	Non-Predacious
Syritta pipiens (L.), 1758	1	4	Non-Predacious
Tropidia scita (Harris), 1780	2	0	Non-Predacious
Xylota jakutorum Bagatshanova, 1980	2	0	Non-Predacious
Xylota segnis (L.), 1758	10	2	Non-Predacious
Xylota sylvarum (L.), 1758	1	0	Non-Predacious
Cheilosia albitarsis (Meigen), 1822	10	0	Non-Predacious
Cheilosia antiqua (Meigen), 1822	2	0	Non-Predacious
Cheilosia bergenstammi Becker, 1894	1	0	Non-Predacious
Cheilosia pagana (Meigen), 1822	0	1	Non-Predacious
Cheilosia variabilis (Panzer), 1798	7	0	Non-Predacious
Total abundance	**462**	**703**	
Total Species	**45**	**34**	

Appendix III.3

Environmental variables (mean ± SE) measured throughout the study for both study regions (Co. Sligo and Co. Wexford). Numbers in bold indicate significant differences between the two counties (Mann-Whitney U test; P<0.05).

	Co. Sligo	Co. Wexford		
	Mean ± SE	Mean ± SE	Mann-Whitney U	P value
Altitude	47.6 ± 8.3	159.2 ± 23.5	24.00	**<0.01**
FA	1.3 ± 0.1	1.7 ± 0.3	86.00	0.29
LHQS	0.5 ± 0	0.4 ± 0	108.50	0.87
FHQS	0.3 ± 0.1	0.1 ± 0	43.50	**<0.01**
SFC	63.7 ± 10.5	47.5 ± 9.7	83.00	0.23
BL1	2182.2 ± 501.5	268.9 ± 120.6	25.00	**<0.01**
BL2	93.9 ± 41	351.8 ± 149	70.00	0.08
FW4	1869.6 ± 364	500.7 ± 108.4	39.00	**<0.01**
GS2	13.9 ± 13.9	324.7 ± 171.4	89.00	0.35
WL1	5747.2 ± 658.3	6264.3 ± 526	99.00	0.60
WL2	508.5 ± 117.2	456.9 ± 147.6	87.00	0.31
CLLF	10415.4 ± 529.3	8167.2 ± 582	50.00	**0.01**
LFR	4.3 ± 0.2	4.1 ± 0.2	107.00	0.84
LFD	1 ± 0	0.7 ± 0.1	50.00	**0.01**
NVS	4.9 ± 0.6	3.8 ± 0.6	102.00	0.68

Acronyms: FA= Field Area; LHQS = Linear Habitat Quality Score; FHQS = Field Habitat Quality Score; SFC = Shrub Flower Cover; BL1 = Stone Walls and Other Stonework; BL2 = Earth Banks; FW4 = Drainage Ditches; GS2 = Dry Meadows & Grassy Verges; WL1 = Hedgerows; WL2 = Treelines; CLLF = Combined Length of Linear Features; LFR = Linear Feature Richness; LFD = Linear Feature Diversity; NVS= Nature Value Score.

Appendix III.4

Mean ± SE abundance and species richness of total hoverflies with associated functional groups collected across a range of farming intensities (n = 10 for each farming intensity).

	Extensive	Intermediate	Intensive	Total
Abundance				
Total†	25.5 ± 10.87	17.9 ± 4.97	24.4 ± 6.45	22.6 ± 4.41
Predacious†	9.6 ± 3.06	6.8 ± 1.84	18.4 ± 6.68	11.6 ± 2.6
Non-Predacious	15.9 ± 8.82	11.1 ± 3.33	6 ± 1.26	11 ± 3.15
Episyrphus balteatus	1.2 ± 0.49	4.1 ± 2.49	43.4 ± 27.75	16.23 ± 9.65
Species Richness				
Total	10.1 ± 2.3	9.4 ± 1.9	9.8 ± 0.63	9.77 ± 0.98
Predacious	5.1 ± 0.99	4.3 ± 0.92	6.4 ± 0.45	5.27 ± 0.49
Non-Predacious	5 ± 1.5	5.1 ± 1.04	3.4 ± 0.52	4.5 ± 0.63

† Excluding the dominant species *E. balteatus*.

Appendix IV

Appendix IV.1

Maximizing biodiversity conservation through an optimal sampling strategy: Malaise vs pan traps

Developing an effective and reliable sampling strategy for invertebrates used in conservation research is of paramount importance, since the lack of an appropriate sampling strategy may affect how we evaluate species-rich farm habitats and has important implications for result-based agri-environment payment schemes. Given that both Malaise and pan trapping have been used as an effective sampling strategy for the assessing hoverfly populations (Stanley et al., 2013; Power et al., 2016; Carey et al., 2017a & b; Schirmel et al., 2018), evaluating the performance of Malaise and pan traps observed in Co. Sligo survey during 2018 and 2019 survey, respectively, is interesting and worthy of discussion. Although certain aspects of the two sampling periods and trap types may explain the results obtained in our study, to our knowledge, there is a lack of information regarding the comparison of Malaise and pan traps for sampling hoverflies. Given that the comparison of the two sampling methods was not one of the main objectives of the present study, this discussion aims specifically to provide future research pathways to develop an optimal sampling strategy for hoverflies in future similar studies.

Overall hoverfly abundance and species richness were greater in Malaise traps (8,774 specimens and 76 species) than in pan traps (462 specimens and 45 species) similar to the results obtained by Geroff et al. (2014) for wild bees where Malaise traps had a significantly greater bee species richness and abundance than pan traps in a west-central Illinois restored tallgrass prairie. A potential factor that may have impacted on the results of our study was that traps were tested in slightly different, albeit similar, settings. Although both studies took place in similarly classified farmlands in Co. Sligo, the slight difference between some sites (i.e., some different farm locations for 2018 and 2019 survey) may have impacted on data since the two surveys were part of a larger project and sites were selected in 2019 with a focus on pollinators only, the results of which will be published elsewhere (Maher et al., in prep.).

An important factor in this comparison of trap performance is that the two trap types were deployed in different years (Malaise traps were deployed in 2018 and pan traps in 2019). Not surprisingly, Malaise traps captured greater hoverfly species richness and abundance, even in 2018, which was deemed a poor year for hoverfly sampling possibly due to the summer heatwave (Dipterists Forum, 2019). In addition, it is interesting to note that only one sciomyzid individual was collected in pan traps while a total of 273 individuals and 17 species was collected in Malaise traps. However, the sampling periods varied considerably for each trap type, with Malaise traps having an additional sampling period. The 2018 Malaise trapping ran from May to September with 7 sampling periods, while the 2019 pan trapping took place from May to September and only had 6 sampling periods. Having an additional sampling period resulted in (266 more hoverfly individuals) but did not result in any additional species found. Furthermore, the length of time each trap activated on site varied during the two sampling periods. The Malaise traps were operational for two weeks at a time while the pan traps were operational for only two days (see methods section in Chapter 3 & 4). This difference in sampling time is typical of each trap type involved (Stanley et al., 2013; Power et al., 2016; Carey et al., 2017a & b). While there were some variations in the sampling period, trap activation time and sites, it would be worthwhile to compare the efficacy of the two sampling methods under uniform time constraints.

Unlike Malaise traps, pan traps are designed to represent entomophilous flowers (i.e., flowers that can be accessed by pollinators) with abundance/richness of hoverflies are strongly related to high density of floral resources that provides pollen and nectar and thereby reducing foraging time (Meyer et al., 2009; Power et al., 2016; Garratt et al., 2017; Moquet et al., 2018). Therefore, it is not clear whether the poor performance of pan traps in comparison with Malaise traps are also due to floral community of the area that may reduce the attractiveness of pan traps to hoverflies. Cane et al. (2000) found that pan traps failed to accurately represent certain bee species that were known to be present at flowers, suggesting that flowers may draw certain species away from pan traps (Cane et al., 2000; Westerberg et al., 2021). A recent study by Westerberg et al. (2021) has shown that the effect of flower frequency on pan traps performance in coniferous forest and meadows vary among different taxa. The greatest negative effect of flower frequency on pan trap catches was detected on

Vespoidea and Lepturinae while no effect was detected on solitary Apoidea. However, hoverflies along with Cetoniidae, social Apoidea showed intermediate bias towards flower frequency and pan trap catch. Földesi and Kovács-Hostyánszki (2014) suggested that pan traps are less efficient when setup in close proximity to hedgerows since they are likely to interact with flowers, but they are more effective in crop fields (e.g., wheat) due to the absence of flowers that may attract hoverflies. This can have important implications for future AE studies when assessing farmland habitats for invertebrates by considering for example, malaise traps over pan traps when evaluating the value of linear habitats such as hedgerows for invertebrates while employing pan traps in crop fields where flowers are either lacking or may have minimal effects on hoverflies. The scent given off by a flower may also increase the attractiveness of hoverflies to flowers over pan traps since hoverflies have been shown to be attracted to odour (Laubertie et al., 2006). Nevertheless, it known that hoverflies benefit from aphids either in the form of larval diet for predatory species (Schirmel et al., 2018; Dunn et al., 2020; Rodríguez-Gasol et al., 2020), or honeydew from aphids as adult diet for predatory (Rodríguez-Gasol et al., 2020) and non-predatory species (van Rijn et al., 2013). Since only a few aphid individuals were collected in our study compared to Schirmel et al. (2018), the attractiveness of aphids to pan traps and their effect on hoverfly visitation to pan traps is worth further investigation.

Given that the comparison of the two sampling methods was not the main scope of our study, no firm conclusion could be drawn directly from our comparison. However, future studies should specifically address the efficacy of the two sampling method for hoverflies (and bees) using the same sampling period, and habitat type.

Appendix IV.2

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Appendix V Disseminations from FARM-ECOS Project

Appendix V.1

Publications in Internationally Peer Reviewed Journals

1. Ruas, S., Rotchés-Ribalta, R., O´hUallacháin, D., D. Ahmed, K.S., Gormally, M., Stout, J., White, B., Moran, J., 2021. Selecting appropriate plant indicator species for Result-Based Agri-Environment Payments schemes. Ecological Indicators, 126, 107679.
2. Ruas, S., Kelly, R., D. Ahmed, K.S., Maher, S., O'Hora, E., Volpato, A., O hUallacháin, D., Gormally, M., Stout, J., and Moran, J., 2021. Does landscape structure affect the presence of woodland specialist's pollinators in farmland? Contributions for AES design. Biology and Environment (in review).
3. Rotchés-Ribalta, R., Ruas, S., D.Ahmed, K.S., Gormally, M., Moran, J., Stout, J., White, B. and Ó hUallacháin, D., 2020. Assessment of semi-natural habitats and landscape features on Irish farmland: New insights to inform EU Common Agricultural Policy implementation. Ambio, pp.1-14.
4. Volpato, A., D.Ahmed, K.S., Williams, C.D., Day, M.F., O'Hanlon, A., Ruas, S., Rotchés-Ribalta, R., Mulkeen, C., Huallachain, D.O. and Gormally, M.J., 2020. Using Malaise traps to assess aculeate Hymenoptera associated with farmland linear habitats across a range of farming intensities. Insect Conservation and Diversity, 13(3), pp.229-238.
5. D.Ahmed, K.S., Volpato, A., Day, M.F., Mulkeen, C., O'Hanlon, A., Carey, J., Williams, C., Ruas, S., Moran, J., Rotchés-Ribalta, R., Ó hUallacháin, D., Stout, J., Hodge, S., White, B., Gormally, M., 2020. Linear habitats across a range of farming intensities contribute differently to dipteran abundance and diversity. Insect Conservation and Diversity. https://doi.org/10.1111/icad.12455.
6. D.Ahmed, K.S., Stephens, C., Bistline-East, A., Williams, C.D., Mc Donnell, R.J., Carnaghi, M., Huallacháin, D.Ó. and Gormally, M.J., 2019. Biological control of pestiferous slugs using Tetanocera elata (Fabricius) (Diptera: Sciomyzidae): larval behavior and feeding on slugs exposed to Phasmarhabditis hermaphrodita (Schneider, 1859). Biological Control, 135, pp.1-8.

Appendix V.2

Poster Presentations

1. D.Ahmed, K.S., Stephens, C., Bistline-East, A., Williams, C.D., Mc Donnell, R.J., Carnaghi, M., Ó hUallachain, D., & Gormally, M.J., 2019. Controlling pest slugs without chemicals the story of the fly Tetanocera elata and the worm Phasmarhabditis hermaphrodita. BES Annual Meeting, Belfast, UK, December 2019.
2. Day, M. F., D.Ahmed, K.S., Volpato, A., O'Hanlon, A., Hughes, A., Ó hUallacháin, D, Gormally, M.J., 2019. Malaise trap by-catch: How can it contribute to our understanding of invertebrate diversity in farmed landscapes? BES Annual Meeting, Belfast, UK, December 2019.
3. D.Ahmed K.S., Volpato A., Mulkeen C., O'Hanlon A., Carey J., Williams, C., Ruas, S., Moran, J., Rotches-Ribaltes, R, Ó hUallacháin, D., Stout, J., White, B., Lee, A., and Gormally, M., 2019. Assessing habitat quality of farm linear habitats across a range of farming intensities in northwest of Ireland using novel bioindicators. Ryan Institute Research Day, Galway, May 2019
4. Volpato, A., D.Ahmed, K.S., Williams, C.D., Day, M. F., O'Hanlon, A., Ruas, S., Rotches-Ribalta, R., Mulkeen, C., Ó hUallacháin, D. and Gormally, M.J., 2019. Using Malaise traps by catch to assess non focal invertebrate groups: the case for aculeate Hymenoptera associated with farmland linear habitats. Ryan Institute Research Day, Galway, May 2019
5. Ruas, S., Rotches-Ribalta, R., Ó hUallacháín, D.Ahmed, K.S., Gormally, M., Ryan, M., Stout, J., White, B., Lee, A. and Moran, J., 2019. Assessing multiple ecosystem services in Irish farmland – scaling up payments for ecosystem services (PES). 2nd Ecology and Evolution Conference, Galway, January 2019.
6. Rotches-Ribalta, R., Ruas, S., Moran, J., Gormally, M., D. Ahmed, K.S., Stout, J., and Ó hUallacháín, D., 2019. An assessment of semi-natural habitats across a gradient of farming intensities. 2nd Ecology and Evolution Conference, Galway, January 2019.
7. D.Ahmed, K.S., Stephen, C., Bistline-East, A., Williams, C.D., McDonnell, R.J., Carnaghi, M., Ó hUallachain, D., and Gormally, M.J., 2019. Laboratory

investigations on the effects of prey type and availability on the larval stages of the slug killing fly, Tetanocera elata (Diptera: Sciomyzidae). 2nd Ecology and Evolution Conference, Galway, January 2019.

8. Sara Ruas, James Moran, Daire Ó hUllacháin, Roser Rotches, Mary Ryan, Mike Gormally, Karzan D.Ahmed, Jane Stout and Blánaid White, 2018. Farm-ECOS: Enhancing farmland biodiversity and the provisioning of ecosystem services through connectivity at the landscape scale. 16th Annual Law and the Environment Conference, Cork, Ireland, April 2018.

Appendix V.3

Oral Presentations

1. D.Ahmed, K.S., Volpato, A., Day, M.F., Mulkeen, C., O'Hanlon, A., Carey, J., Williams, C., Ruas, S., Moran, J., Rotchés-Ribalta, R., Ó hUallacháin, D., Stout, J., Hodge, S., White, B., Gormally, M., 2022. Assessing biodiversity value of farm linear habitats across different levels of farming intensity in agricultural grasslands using Diptera as bioindicators. ICE, Helsinki, Finland, July 2022 (accepted).
2. Maher, S., Kelly, R., Ruas, S., D.Ahmed, K.S., Lee, A., White, B., Gormally, M., Moran, J., Ó hUallacháin, D., Hodge, S., Stout, J., 2021. Evaluating the quantity and quality of habitat features for pollinators in Irish agricultural landscapes. AAB Association of Applied Biologists, Slough, UK, June – July 2021.
3. D.Ahmed, K.S., Volpato, A., Day, M.F., Mulkeen, C.J., O'Hanlon, A., Carey, J., Williams, C., Ruas, S., Moran, J., Rotchés-Ribalta, R. and ÓhUallacháin, D., 2021. Evaluating different linear farm habitats for selected dipteran families: Implications for diversity and associated ecosystem functions. 3rd Ecology and Evolution Conference, Galway, January 2021.
4. Ó hUallacháín, D., Moran, J., Ruas, S., Gormally, M., D.Ahmed, K.S., Stout, J., Maher. S., Kelly, R., Lee, A., White, B., Rotches-Ribalta, R., 2020. Farming and Natural Resources: Measures for Ecological Sustainability. CIEEM, Dublin, Ireland, April 2020.
5. D.Ahmed, K.S., Stephens, C., Bistline-East, A., Williams, C.D., Mc Donnell, R.J., Carnaghi, M., Ó hUallachain, D., & Gormally, M.J., 2019. Laboratory investigations on the effects of prey type and availability on the larval stages of the slug killing fly, *Tetanocera elata* (Diptera: Sciomyzidae). Irish Ecological Association, 2nd Ecology and Evolution Ireland conference, January 2019, Galway, Ireland.