EVALUATING FIELD MARGINS FOR WILD BEE CONSERVATION AT THE

FARM- AND LANDSCAPE-SCALE IN THE AGRICULTURAL LAND RESERVE

OF DELTA, BRITISH COLUMBIA

by

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B.Sc., Zurich University of Applied Sciences, 2011

A THESIS SUBMITTED IN PARTIAL FULFILLMENT OF

THE REQUIREMENTS FOR THE DEGREE OF

MASTER OF SCIENCE

in

THE FACULTY OF GRADUATE AND POSTDOCTORAL STUDIES

(Integrated Studies in Land and Food Systems)

THE UNIVERSITY OF BRITISH COLUMBIA

(Vancouver)

October 2017

Abstract

Wild bees provide essential pollination service to both agricultural crops and wild flowering plant species. The decline of wild bee species has been associated with a number of different threats, primarily the loss of natural habitat. The Delta Farmland & Wildlife Trust (DF&WT), a non-profit conservation organization, incentivizes farmers to plant hedgerows consisting of native shrubs and trees on the edge of their production fields, mainly to create habitat for wildlife in the Agricultural Land Reserve (ALR) of Delta, British Columbia. In this study, the value of DF&WT's planted hedgerows was evaluated as foraging habitat for wild bees at both the farm and landscape-scale. During the summers of 2015 and 2016, I surveyed bees and flowers in planted hedgerows, as well as the two other most dominant field margin habitats, remnant hedgerows and grass margins. The relationship between floral resources and bees, as well as bee-flower visitations was analyzed and compared among these three habitat types. These empirical data were then used to parameterize the Conefor model, to evaluate the network of field margin patches within the agricultural landscape for their relative importance in landscape connectivity for wild bees.

Overall, wild bees collected from flowers and pan traps were significantly more abundant, species rich and diverse in grass margins compared to planted and remnant hedgerows. While the strongest relationship was found between floral abundance and bee abundance, it did not explain the differences between habitat types alone. Bee-flower visitation records revealed a preference for herbaceous species mostly found in grass margins while only few recommended plant species for hedgerow plantings were visited. The results indicate that grass margins could be a valuable alternative conservation approach or addition to woody hedgerows if properly planned and managed. Connectivity indices generated by

Conefor identified four grass margin patches that most contributed to overall landscape connectivity for bees with different dispersal abilities. These results can be used to help improve field edge management and the spatial targeting of activities by the DF&WT to improve the conservation of wild bee species.

Lay summary

Declines in wild bee populations have highlighted the need for research on the ecological requirements of wild bees. In this study, I evaluated the value of different field margin types as foraging habitat for wild bees at both the farm- and landscape-scale in the Agricultural Land Reserve (ALR) of Delta, British Columbia. In the summers of 2015 and 2016, bees and flowers were surveyed in three habitats: planted hedgerows, remnant hedgerows and grass margins. Field data were incorporated into a landscape connectivity model that identified important field margin patches for the provision of floral resources and the maintenance of habitat connectivity in the landscape for wild bees. I found that grass margins were surprisingly attractive foraging habitats for wild bees that revealed a preference for non-native, herbaceous plant species. Scaling these results to the landscape, I identified field margin patches that most contributed to floral resource availability and habitat connectivity.

Preface

Martina Clausen, thesis author, Dr. Sean Smukler, research supervisor, and Christine Terpsma, Delta Farmland & Wildlife Trust, initiated the research project. Dr. Sean Smukler and Dr. Elizabeth Elle provided guidance in overall field sampling design and data collection.

Christine Terpsma, organized permission for field sampling sites and maintained communication with farmers and landowners in Delta. René McKibbin was the contact person from Environment Canada and guaranteed permission for field sampling within the Alaksen Wildlife Reserve.

Martina Clausen, thesis author, was responsible for managing field sampling days and sites, the collection and analysis of all data. Paula Porto assisted regularly in field sampling, data collection and data analysis.

Dr. Elizabeth Elle and Lincoln Best provided training for bee species identification and verification of identified species. James Strange, research entomologist at the United States Department of Agriculture provided his expert opinion on photographs of several specimens of the *Bombus* genus with an unusual colour variation.

Karen Needham provided the work space and lab equipment at the Beaty Biodiversity museum for species preparation and advised on the proper display of bee specimens for museum collections.

Dr. Sarah Gergel provided guidance for the landscape connectivity analysis using the Conefor software package.

John Petkau and Rashedul Hoque from the UBC Department of Statistics provided assistance for statistical analysis design and coding assistance for R statistical analysis software.

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AES: Agri-Environment Schemes

ALR: Agricultural Land Reserve

BC: British Columbia

CV: Coefficient of Variation

PH: Planted hedgerow

RH: Remnant hedgerow

GM: Grass margin

COSEWIC: Committee on the Status of Endangered Wildlife in Canada

DF&WT: Delta Farmland and Wildlife Trust

Ha: hectares

IPBES: Intergovernmental Science-Policy Platform on Biodiversity and

                    Ecosystem Services

PC: Probability of Connectivity

EC(PC):   Equivalent connectivity for PC

dPC: Change in PC

dPCintra: patch contribution in terms of intrapatch connectivity

dPCflux: patch contribution in terms of flux dispersal

dPCconnector: patch contribution in terms of topological location

a: Habitat attribute

d: Dispersal distance

Acknowledgements

I am grateful to all of the people who surrounded and supported me during my time at UBC. To the SAL Lab, my ISLFS friends and fellow bee enthusiasts– Madeleine, Susanna, Tori, Marika, Ellie, Claudia, Paula and Michelle– your beautiful spirits inspired and encouraged me.

I feel fortunate to have had a supervisor, Dr. Sean Smukler, who in the first place gave me the opportunity to start this journey as a graduate student on the other side of the world; thank you for challenging me and helping me grow as a scholar. I would also like to thank my committee members, Dr. Sarah Gergel and Dr. Elizabeth Elle, this research would not have been possible without their willingness to share their expertise. I would like to acknowledge the financial support and recognition I received from the UBC Faculty of Land & Food Systems and the Noel Roddick Service Award.

Thank you to my parents, Irene and Willy, for offering me the world, and to my sisters, Alexandra and Laura, for the unconditional love and support from overseas.

And finally, a special thanks to my partner, Bern, for cheering me on and being proud of the hard work I put into my research. Your humour and patience through stressful times made this endeavour possible.

To women in science

Chapter 1: Introduction

1.1   General introduction

Worldwide, the transformation of wild habitats to agricultural production has resulted in the loss of biodiversity, and endangers species persistence as well as ecosystem functioning (Kleijn et al., 2015). Biodiversity increases the magnitude and/or the stability of ecosystem functioning with most species contributing in some ways (Kleijn et al., 2015). The benefits that humans gain from functioning ecosystems are known as ecosystem services and have become a supporting argument for the importance of biodiversity conservation (Kleijn et al., 2015). Pollination of flowering plants is a vital ecosystem service for human wellbeing (Garibaldi et al., 2014; Gill et al., 2016). As much as 75% of agricultural crop species are directly or indirectly dependent on animal pollination, with bees being considered the most important contributors to crop pollination services (Klein et al., 2007). In addition to global crop yield enhancement, pollination of wild plants provides crucial food sources for wildlife (Ollerton et al., 2011).Without these pollinators, many plants would be reduced in their reproduction capability; food webs and therefore whole ecosystems could collapse (Kearns et al., 1998).

The western or European honey bee *Apis mellifera,* which was introduced to North America by colonists from Europe (Kremen et al., 2002) is generally regarded as the most economically valuable pollinator of agricultural crops worldwide (Buczek, 2009; Carreck and Williams, 1999). Honey bees can be managed in easily transportable boxes and placed for targeted pollination of many agricultural crops (Ministry of Agriculture, 2012). Over recent years honey bee populations have been in decline, and colony losses due to 'Colony Collapse Disorder' (Buczek, 2009) have resulted in costly hive rental fees and supply

shortages for growers in North America (Park et al., 2015). This trend combined with the rapid increase in pollinator-reliant crops for food production which has grown by 300 % over the last five decades, illustrate the risks of a strategy to rely on a single species for pollination for the human food supply (Goulson, 2003; IPBES, 2016).

In addition to honey bees, there are over 25,000 species of wild bees in the world, with many species yet to be identified (The University of Arizona, 2015). Winfree et al. (2007) investigated whether wild bees will buffer potential declines in agricultural production because of honey bee losses. They found that wild bees alone were sufficient to pollinate the watermelon crop fully at > 90% of the 23 farms studied in New Jersey and Pennsylvania, USA. A study by Button and Elle (2014) in the Lower mainland of British Columbia (BC), Canada, found that bumble bees (*Bombus spp.*) were more effective blueberry pollinators than honey bees, due to their ability to buzz pollinate flowers. Buzz pollination describes the pollen release by vibration of the flight muscles of bumble bees at an appropriate resonant frequency (Moisset and Buchmann, 2010). Honey bees do not have the ability buzz pollinate flowers (Moisset and Buchmann, 2010). Thus, increased abundance of wild bumble bees lead to higher blueberry yields (Button and Elle, 2014). Other economically important crops that require this kind of buzz pollination include tomatoes, blueberries, and cranberries (Michener, 2000; Moisset and Buchmann, 2010). For early blooming fruit crops, mason bees (*Osmia spp.*) have been shown to be much more efficient pollinators than honey bees, especially under poor weather conditions (Bradbear, 2009; Pfiffner and Müller, 2014; Van Westendorp and McCutcheon, 2001). For example only a few hundred females of the species *Osmia cornuta* are needed to pollinate a hectare of apple

or almond trees whereas tens of thousands of honey bee workers would be needed to accomplish the same (Bosch and Kemp, 2001; Vicens and Bosch, 2000).

A number of scientists have suggested new practices for integrated management of both honey bees and diverse wild insect assemblages that will lead to the highest quality of ecosystem services (Button and Elle, 2014; Garibaldi et al., 2014). Mounting evidence suggests that with greater bee diversity the quantity and quality of pollination in agro-ecosystems as well as in natural ecosystems increases more than with managed honey bees alone (Garibaldi et al., 2014; Klein et al., 2009; Kremen et al., 2002; Slagle et al., 2009). Bees with different functional traits (e.g. tongue length) complement or synergize pollination services, therefore increase the proportion of flowers setting fruits (or seeds) and product quality (e.g. fruit size and shape) (Garibaldi et al., 2014). Furthermore, interspecific differences in response traits to climate and land-use change can increase resilience of pollination services (Brittain et al., 2013; Williams et al., 2010; Winfree et al., 2009). Free pollination services provided by diverse wild bee communities potentially provide a pollination insurance policy against the on-going decline of honey bee stocks (Winfree et al., 2007). Natural or semi-natural areas near farmland have been shown to lead to more abundant and diverse wild bees in crops compared to homogenous crop areas (Garibaldi et al., 2011; Morandin and Kremen, 2013; Ricketts et al., 2008). Field edges, such as grass margins or hedgerows offer the potential as a means to integrate conservation of wild bees and associated pollination services to crops and wildflowers in surrounding habitats (Morandin and Kremen, 2013; Orford et al., 2016). It is therefore important to evaluate the diversity of wild bees in field margins, including species that are potentially valuable pollinators of adjacent cropland (Hannon and Sisk, 2009; Kevan, 1999).

1.2   Status and trends for wild bees

Wild pollinators have declined in abundance and diversity, with varying magnitude among taxa, at local and regional scales in North West Europe and North America (IPBES, 2016). The International Union for Conservation of Nature (IUCN) Red List assessments estimate that 16.5 % of vertebrate pollinators are threatened with global extinction (IPBES, 2016). While insect pollinator assessments on the global level are missing, regional and national assessments indicate that often more than 40 % of bee species are threatened (IPBES, 2016). The Committee on the Status of Endangered Wildlife in Canada (COSEWIC) currently lists seven wild bee species at risk (COSEWIC, 2017). Four out of these are either endangered, threatened or of special concern in the province of BC, all of which are bumble bees (COSEWIC, 2017). Of the wild bees, bumble bee declines are the most well-known, whereas trends for other species are unknown or only known for a small part of the species' distribution (IPBES, 2016).

Numerous wild bee species are threatened by a land-use change, intensive agricultural management and pesticide use, environmental pollution, invasive alien species, pathogens and climate change (IPBES, 2016). A meta-analysis of studies evaluating these threats by Winfree et al. (2009) found that the loss of natural habitat was most significantly adverse to the abundance and diversity of bees. A study by Koh et al. (2015) that modeled trends in wild bee abundance between 2008 and 2013 indicated a decline of 23% across the United States. This decline was generally associated with the loss of natural habitat due to conversion to row crops. Many features of current intensive agricultural practices endanger bees and their pollination services (IPBES, 2016). Large fields of monocultures, particularly consisting of wind-pollinated crops such as wheat and corn, constrict bee populations to

remnant semi-natural habitats such as field margins and roadsides (Kevan 1999). The conjoined isolation of bee populations from one another reduces genetic diversity and dispersal across the landscape (Roulston and Goodell, 2011). The impairment and elimination of semi-natural habitats removes food and nesting resources that are crucial, especially when crops are not in bloom or when the crop is not insect-pollinated (Kevan 1999). Moving towards more sustainable agriculture and reversing the simplification of agricultural landscapes by protecting, restoring and connecting patches of natural and semi-natural habitats offer strategic responses to risks associated with pollinator decline (IPBES, 2016). These strategies may concurrently mitigate the impacts of land-use change, land management intensity, pesticide use and climate change on pollinators (IPBES, 2016).

1.3   Habitat restoration for wild bee conservation

By restoring natural habitat within agricultural landscapes the two primary goals of conserving biodiversity and restoring ecosystem services could be met (Morandin and Kremen, 2013). A recent study by Kremen & M'Gonigle (2015) showed that even within highly intensive agricultural landscapes, small-scale habitat restoration could be a successful conservation management tool for pollinators, given that the habitat enhancements are tailored to promote certain species. Several countries have promoted various Agri-Environment Schemes (AES) that encourages the creation of flower-rich habitat in form of cover crops, field border plantings, buffer strips and hedgerows (Wratten et al., 2012). According to Nicholls & Altieri (2013) the presence of diverse flowering plant species helps to maximize bee diversity. In addition, flower abundance significantly affects reproductive success of wild bees (Pfiffner and Müller, 2014). A study in the central Swiss plateau by

Tschumi et al. (2016) demonstrated that diverse wildflower strips implemented through AES not only benefits farmland biodiversity, but also promoted biological pest control and crop yield.

The efficacy of such measurements can be enhanced if implemented from farm- to landscape-scales that correspond with pollinator mobility, thus assuring connectivity among natural or semi-natural habitats (IPBES, 2016). In order to sustain functional connectivity among subpopulations to facilitate dispersal and minimize mortality risk, areas that are most critical for habitat connectivity and habitat availability need to be identified and protected (Saura and de la Fuente, 2016).

1.3.1   Delta Farmland and Wildlife Trust (DF&WT)

The Delta Farmland and Wildlife Trust (DF&WT) is a non-profit organization established in 1993. Their goal is to promote the preservation of farmland and wildlife habitat on the lower Fraser River delta through co-operative land stewardship with local farmers. Currently they run six Stewardship Programs, which incorporate research, education, and financial incentives to establish wildlife habitat and/or invest in soil fertility on farmland (DFWT, 2015). Their Hedgerow Stewardship Program provides funding to establish new hedgerows on farmland in Delta and helps absorbing the costs for land taken out of agricultural production. Hedgerows are linear barriers of trees, shrubs and perennial vegetation usually associated with agricultural field boundaries (Hannon and Sisk, 2009). Within agricultural landscapes, hedgerows support wildlife with food, shelter, breeding sites and can function as corridors connecting habitat fragments (DFWT, 2015). The DF&WT recommends selecting a diverse mix of native trees and shrubs to improve the hedgerow

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habitat value for wildlife. To provide farmers with hedgerow planning guidance, the

DF&WT published a fact sheet that identifies a number of plant species suitable for planting

and outlines some of the wildlife benefits provided by them as suggested through

consultation with local biologists (Terpsma, pers. communication). Grouped into mammals,

birds and insects, the list offers general recommendations and is not targeted at specific

species within these groups. While there have been studies that have evaluated the hedgerow

program for impacts on avian species (DF&WT, 2015a) and its contribution to climate

change mitigation (Thiel et al., 2015), it is unclear how well current hedgerow plant species

support insects communities, or more specifically, wild bees (Terpsma, pers.

communication).

1.4   Research objectives and hypotheses

The overarching goal of this research was to provide DF&WT guidance for

optimizing habitat conservation and restoration investments for wild bees in the ALR in

Delta. The study described in Chapter 2 does this at the farm-scale, while the study in

Chapter 3 utilizes these data to model habitat at the landscape-scale.

In Chapter 2, the presence of wild bee species in Delta and the Lower Fraser Valley is

assessed, contributing important information to the limited data on BC's pollinators,

including the approximately 430 species of bees (Sheffield and Heron, 2017). An aim of this

data collection was to help identify what it is that we are trying to conserve (O'Toole, 2002).

Without long-term international and national monitoring, it is difficult to determine how

different species respond to land-use alteration, available plant resources or climate change

(IPBES, 2016). Furthermore, understanding the conservation needs of wild bees is important

because they represent alternative pollinators to honey bees (O'Toole, 2002). Discussing the role and floral characteristics of field margins as a place where wild bees can thrive and adding information to bee-flower visitation records helps improving restoration measurements for a targeted group of bee species (Kevan, 1999; O'Toole, 2002). This study is the first to investigate the efficacy of DF&WT planted hedgerow program for wild bee conservation. The research objectives for Chapter 2 were to:

Obj. 1: Assess the abundance and diversity of wild bee communities found in natural and semi-natural habitats in Delta and the Lower Fraser Valley

Obj. 2: Estimate and compare wild bee abundance, species richness and diversity between three different types of field margins: planted hedgerows (PH), remnant hedgerows (RH) and grass margins (GM)

Obj. 3: Better understand the relationship between floral resources and bee abundance, species richness and diversity.

The hypotheses I tested in this study were:

H$_1$: PH consisting of a diverse mix of native trees and shrubs have a more abundant and diverse wild bee populations than RH and GM

H$_2$: Floral abundance, species richness and diversity are significant predictor variables for bee abundance, species richness and diversity in farm field margins

Four plots in each of three habitat types were monitored in the summers of 2015 and 2016. Field survey data was then integrated into model parameterization to conduct a landscape connectivity analysis using the software package Conefor 2.6 (Saura and Torné, 2009) in Chapter 3. Through this analysis the amount of reachable resources for small and

large wild bee species provided by field margins was assessed and areas most critical for habitat connectivity and availability identified. The aim of this analysis was to shift the focus of protection and restoration activities to the landscape-scale, so that the combined threats of habitat loss and fragmentation can be addressed. The research objectives for Chapter 3 were to:

> **Obj. 4:** Determine the season with the lowest habitat resource availability for the parameterization of field margin habitat attributes for the Conefor landscape connectivity analysis

> **Obj. 5:** Compare connectivity indices calculated with different dispersal distances for small and large bees to assess the effects of uncertainty in dispersal distance determination on connectivity indices

> **Obj. 6:** Identify areas that are most critical for habitat connectivity and habitat availability for small and large bees

In this study the hypotheses I tested were:

> **H$_3$:** Habitat resources provided by field margins differ between Early-, Mid- and Late-season

> **H$_4$:** Connectivity indices calculated within estimated ranges of dispersal distances for small and large bees are highly correlated

Chapter 2: Evaluating hedgerow establishment in farm field margins for wild bee conservation in Delta, British Columbia

2.1 Introduction

Agriculture is the world's largest land use, accounting for 40–50% of terrestrial land around the globe (Kremen and M'Gonigle, 2015). Given the global dominance of agriculture, the role and management of semi-natural habitats within these types of human-modified landscape is critical for some wildlife. Semi-natural habitats in and around farmland are particularly important given that agricultural intensification has led to a strong decline in both wild bee abundance and diversity due to much reduced provision of food and nesting resources as well as increasing habitat fragmentation (Pfiffner and Müller, 2014).

The vast majority of wild bee species in North America are solitary nesters with a short individual flight season of only 3–6 weeks and different species flying in spring, early summer and late summer respectively (Packer et al., 2007; Wilson and Carril, 2016). Wild bees can also be social and live in colonies that are active for extended periods and so require food sources from spring to fall (Michener, 2000). Bumble bees are social and, like most social species, are generalists, and often the first bees out in spring and the last bees seen in fall (Michener, 2000). While the majority of wild bees in Canada are generalists that can be found foraging on pollen on a wide variety of plants, there are some specialists that forage on pollen from a single plant species or family. However, specialists may collect nectar from a wider range of flowers in addition to the ones they visit for pollen (Packer et al., 2007; Roulston and Goodell, 2011). Therefore a continuous provision of diverse floral resources during the whole season is essential to maintain species diversity (Oertli et al., 2005). Results

from Roulston and Goodell (2011) showed that floral resource abundance and diversity was the most clearly the limiting factor for bee populations and that nesting resources were likely to correlate with floral resources.

Semi-natural habitats in agricultural landscapes can offer additional floral resources to complement mass-flowering crops that are often blooming for a short period of time (Menz et al., 2011; Todd et al., 2016). Protecting or restoring uncultivated field edge habitat, in particular, is an important strategy to enhance biodiversity on farmland without taking land out of production (Garbach and Long, 2016). The installation of hedgerows of shrubs and trees, wildflower strips, and perennial grass plantings along field boundaries provides many benefits including water quality protection, pest control, and habitat for wild bees that enhance pollination in adjacent crops (Garbach and Long, 2016; Morandin and Kremen, 2013; Tschumi et al., 2016). Morandin and Kremen (2013) not only observed enhanced pollinator populations in restored hedgerow sites, but also greater abundances of wild bees up to at least 100 m into fields. Their data suggest that hedgerows are net exporters rather than neutral or concentrators of pollinators and that the additional provision of floral resources do not draw pollinators away from crop plants as many farmers might fear (Morandin and Kremen, 2013). In the same study, Morandin and Kremen (2013) compared floral and nesting characteristics and populations of pollinators between restored native perennial plant hedgerows and weedy, relatively unmanaged field margins. They found enhanced pollinator populations over multiple seasons in hedgerow sites as well as a significantly greater abundance of uncommon bee species compared to the control sites (unmanaged field margins). Not surprisingly, planting hedgerows along farm edges is the

most widely used ecological restoration technique to promote pollinators in agricultural landscapes (M'Gonigle et al., 2015).

The Delta Farmland & Wildlife Trust (DF&WT), a non-profit conservation organization, incentivizes farmers to plant hedgerows on the edge of their production fields, mainly to create habitat for wildlife. To date, it is not known whether these plantings enhance wild bee communities specifically.

To better understand how the DF&WT's Hedgerow Stewardship Program impacts wild bee pollinations, I conducted a field survey during the summers of 2015 and 2016. In this study I: 1. Investigated whether field margins planted with hedgerows consisting of native shrubs and trees in the ALR in Delta, BC increased wild bee abundance, species richness, diversity and floral resources, in comparison to two common, unmanaged field margin types: remnant hedgerows and grass margins; 2. Analyzed the relationship between floral resources and bees found in each field margin type to identify floral characteristics most likely to ensure bee abundance, species richness and diversity and; 3. Developed concrete recommendations based on this data analysis to help DF&WT improve their hedgerow program.

2.2 Material and methods

2.2.1 Site selection

The study was conducted in the ALR in Delta, BC in the summers of 2015 and 2016. The nearest Environment Canada Weather station data shows a mean annual temperature for 1981-2010 of 10.6° C with 1262.4 mm of precipitation and a seasonal average temperature for April to August of 15.2 ° C with 44.5 mm of precipitation (Government of Canada,

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2017). The seasonal average was 19.6° C with 44.0 mm of rainfall in 2015 and 17.8° C with

25.3 mm of rainfall in 2016 (World Weather Online, 2017) (Table 2.1).

Table 2.1: Canadian Climate Normals 1981-2000 for Delta, BC as recorded by Canada's National Climate Archive (Government of Canada, 2017) compared to average temperatures and rainfall in 2015 and 2016 (World Weather Online, 2017)**.**

Year	Month	April	May	June	July	August
1981-2000	Temperature Average (° C)	10.4	13.3	16.0	17.9	17.9
	Rainfall (mm)	67.9	52.2	42.6	30.5	28.7
2015	Temperature Average (° C)	12	19	22	24	21
	Rainfall (mm)	81.4	25.2	5.9	15.8	92.1
2016	Temperature Average (° C)	14	16	18	20	21
	Rainfall (mm)	23.5	16.5	55.9	22.0	8.4

The main field crop types produced on the 6,344 ha of cultivated land in Delta's ALR

are vegetables (38%), forage/pasture (34%) and berries (19%) (Ministry of Agriculture,

2012b). Four planted hedgerow plots (100 m x 2 m) adjacent to crop fields across Delta's

ALR were selected each year from the 15 hedgerows that have been established by the

DF&WT in the region, based on accessibility and availability of nearby control plots. For

each planted hedgerow, matching control plots of the same size in relatively unmanaged

remnant hedgerows and weedy margins were chosen within 500 m distance in order to

ensure similar environmental conditions (Morandin and Kremen, 2013) (Figure 2.1).
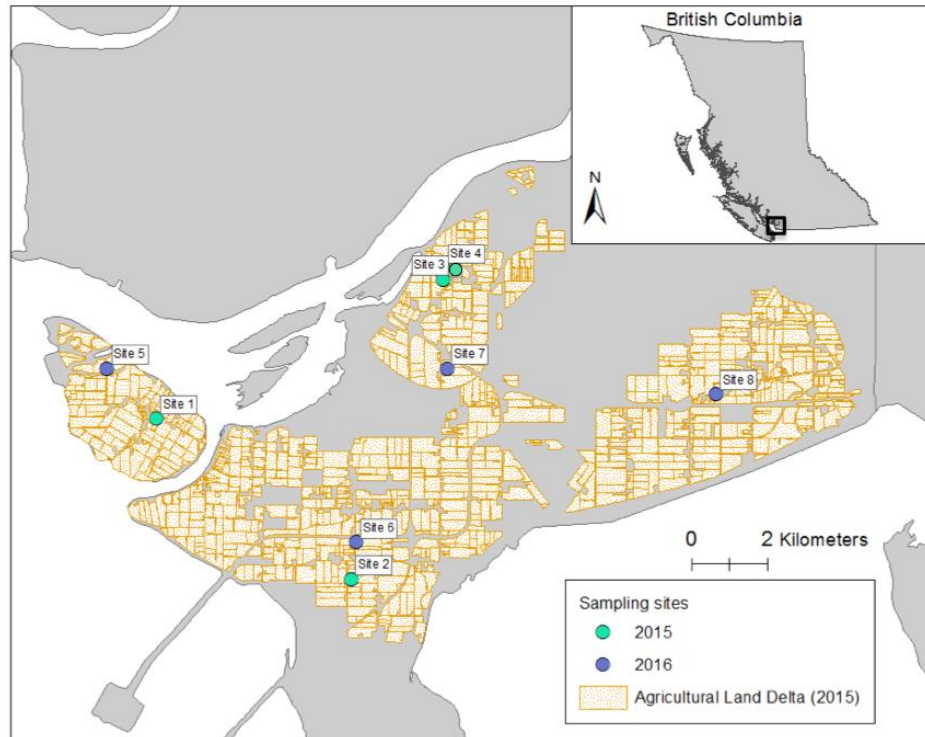
Figure 2.1: Sampling sites from 2015 and 2016 in the ALR in Delta, BC, Canada. Four planted hedgerows were matched with corresponding control sites each year.

Planted hedgerows were between three and 20 years of age and consist of a diverse mix of native trees and shrubs outlined by the DF&WT and modified by the farmer's specific needs (DF&WT, 2015b), often bordered by a small strip of perennial grasses. Due to the limited number of DF&WT planted hedgerows, a 40-year-old hedgerow in the Alaksen National Wildlife Area was chosen as one of the planted hedgerows sites that was not planted by the DF&WT in 2016. However, this hedgerow contained a very similar mix of native trees and shrubs and was comparable to all other planted hedgerows. Planted hedgerows were compared to remnant hedgerows and weedy field margins because these are the most common field margin types for crops in the study area. Remnant hedgerows were often dominated by invasive, non-native species and were found to have a lower plant

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species richness and diversity than planted hedgerows in previous surveys (Thiel et al., 2015). Grass margins consisted of herbaceous vegetation and differed widely in their management. Some were mowed or even ploughed during the sampling season. Wherever possible, planted hedgerows and corresponding control plots were sampled on the south-or east-side due to the influence of sunlight exposure on bee activity (Sardiñas et al., 2016; Sheffield, 2009). However, this was not possible in all cases due to access limitations.

2.2.2   Bee sampling with nets and pan traps

Bees in planted hedgerows and control plots were sampled monthly with nets and pan traps in the summers of 2015 and 2016. Bees were sampled alternately with pan traps and aerial nets approximately bi-weekly. The two sampling methods were applied to avoid sampling bias associated by the sampler's netting and observational skills described by Roulston et al. (2007) and in order to characterize the local bee fauna. In 2015 a total of eight sampling rounds were conducted, consisting of five sampling rounds with nets from May to August and three sampling rounds with pan traps from June to August. Based on observations in the first year, sampling in 2016 began a month earlier. Thus in 2016 a total of nine sampling rounds were conducted, consisting of five sampling rounds with nets from April to August and four sampling rounds with pan traps from May to August (Table 2.2).

Table 2.2: Overview of sampling rounds with nets and pan traps in 2015 and 2016.

Year	Sampling Method	Early Season		Mid-season		Late season	Total
		April	May	June	July	August	
2015	Net Rounds	-	1	1	1	2	5
	Pan Trap Rounds	-	-	1	1	1	3
	Vegetation Surveys		1	2	2	3	8
2016	Net Rounds	1	1	1	1	1	5
	Pan Trap Rounds	-	1	1	1	1	4
	Vegetation Surveys	1	2	2	2	2	9

Collections were made at a planted hedgerow plot and its corresponding control plots on the same day, all sampling sites were collected within 7 ± 1 day. Due to bee's sensitivity to weather conditions, bee sampling was only conducted on days with a minimum temperature of 15 °C, low wind activity and sunny or slightly overcast conditions (Evans, 2016; Feltham et al., 2015; Russo et al., 2015).

Aerial netting capturing bees visiting flowers in each plot was conducted for 15 min, not including the time processing each specimen. Collectors walked at a consistent pace along the 100 m x 2 m plots and back, collecting at patches of blooming flowers. All plots were sampled between 9.30 AM and 15.30 PM in a stratified random approach that ensured the sampling order was altered and collectors among survey locations were rotated. Bee specimens were terminated using ethyl acetate killing tubes plus the application of 100% ethanol on the specimen's face to ensure the bee was definitely put to death. Subsequently specimens were stored in the freezer until they were prepared and pinned for identification (Droege, 2012).

Bees were collected by pan traps using a protocol adapted from the Canadian Pollination Initiative (Sheffield, 2009) and the United States Geological Survey Native Bee Inventory and Monitoring Lab (Droege, 2012). A total of 30 pan traps consisting of ten each of blue, yellow, and white coloured plastic bowls (12 oz.) were laid out along the 100 m x 2 m plots between 9 AM and 11 AM for a total amount of four to five hours for the middle part of the day (Droege, 2012). Each pan trap was filled up to ¾ with unscented soap water to reduce surface tension and laid out at each site in a random colour order at approximately 3.3 m intervals. Tall vegetation was pushed aside, so that the bowls were clearly visible from above. Once specimens were retrieved, they were washed and stored in in 70% ethanol

before putting into the freezer until they were prepared and pinned for identification (Droege, 2012).

2.2.3 Vegetation survey

At each bee sampling round, vegetation surveys were conducted in each sampling plot for a total of eight times in 2015 and nine times in 2016 (Table 2.2). To quantify the availability of floral resources, five $1m^2$ quadrats were randomly placed, one each within a 20 m section along the 100 m x 2 m sampling plots in grass margins. Within those quadrats the number of open flowers were counted by species. Flower units were categorized as a single flower or inflorescence for densely clustered floral heads (e.g. Asteraceae, Fabaceae, Plantaginaceae) for the purposes of this study (Dicks et al., 2015). Inflorescences were identified as the largest consistent floral unit among individuals within a species. To incorporate differences in the vertical distribution of flowers in hedgerow plots, two $1m^2$ quadrats were stacked at the five locations and flower counts were averaged back to $1 \ m^2$ to get a comparable measurement between different habitat types. To track floral species richness, all plants in bloom within sampling plots were identified and recorded. Plants were either identified to species in the field or samples and photos were taken to identify in the lab.

2.2.4 Bee identification

Bee species within family or genus can differ in their habitat requirements for nesting and floral resources (Michener, 2000). Therefore, responses to changes in the environment, such as the loss of habitat might differ within family or genus and some trends may only

become apparent when analyzing species separately (Gonzalez et al., 2013; Williams et al., 2010). Whenever possible bees were identified to species using keys (Gibbs, 2010; Roberts, 1973; Sheffield et al., 2011; Stephen, 1957) and comparisons with reference collections from the Beaty Biodiversity Museum at the University of British Columbia and the Pollination Ecology Lab at Simon Fraser University in Vancouver. Following initial identifications to genus using (Packer et al., 2007), all specimens were brought to Dr. Elizabeth Elle's Pollination Ecology Lab for species identification or for confirmation of identifications. Species level identification was not possible in all cases. Species of the genus *Lasioglossum*, other than *Lasioglossum zonulum*, were subdivided into sub-genus *Dialictus* and *Evylaeus*. In each the following genera, *Andrena*, *Bombus*, *Melissodes*, *Osmia* and *Sphecodes,* one bee species could not be identified and was labeled sp.1. Unidentified male specimens of the same genus were labeled with sp.a and were not included in the species richness analysis, in case they were the same species as sp. 1 females. Most specimens will be added to the Spencer Entomological Collection at the Beaty Biodiversity Museum.

2.2.5   Bee community variables

Bee communities were characterized using measures of abundance, species richness and species diversity. Bee abundance was calculated as the total number of bees collected at each sampling plot summarized over the collection period. Species richness was measured as the number of species encountered over the collection period. Species diversity was measured using the Shannon-Wiener Index in the vegan package (Oksanen et al., 2016) in R (R Core Team, 2016). It is defined as $H' = -\sum_{i=1}^{R} p_i \ln p_i$ where p_i is the proportional abundance of all detected species i and ln is the natural log (Oksanen et al., 2016). The index

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incorporates both components of biodiversity; species richness and abundance and can be easily compared between sites and studies. The Shannon-Wiener index increases as both the richness and the evenness of the community increase and decreases as some species are much more abundant, presumably dominating the community (Magurran, 2005).

Analysis of similarities between communities sampled with nets and pan-traps were conducted to assess if data should be analyzed separately (Morandin and Kremen, 2013). A Permutational Multivariate Analysis of Variance Using Distance Matrices was applied using the adonis function with a Bray-Curtis distance distribution in the vegan package (Oksanen et al., 2016). A significant difference was found between the sampling techniques ($p < 0.05$) therefore the data was analyzed separately. Data was also separated by sampling year due to differences in weather pattern and sampling intensity. The non-native European honey bee was separated for most analyses as it is a managed species and thus its abundance mostly depends on the placement of hives within the region (Kremen and M'Gonigle, 2015; Moritz et al., 2010). However, honey bee colonies may benefit from native plants in grass margin and hedgerow sites, as they provide additional floral resources in otherwise intensively managed agricultural landscapes (Morandin and Kremen, 2013).

2.2.6   Floral resources

Floral resources were characterized similarly to bee communities using measures of abundance, species richness and species diversity. Floral abundance was calculated as the average count of open flowers per square meter for each sampling plot at each sampling round. Due to differences in dimensions between grass margins and the two hedgerow types, floral abundance per plot was also calculated by multiplying the average count of open

flowers per square meter by the total volume estimated for each habitat type. For the two-dimensional grass margins floral abundance per plot was simply defined as length*width. For three-dimensional hedgerows, the height per habitat type was estimated from five height measurements taken at each hedgerow plot and multiplied by length and width of the plots.

Species richness was measured as the number of flowering plant species encountered along the sampling plot over the collection period. Species diversity was measured using the Shannon-Wiener Index in the vegan package (Oksanen et al., 2016) in R, incorporating floral abundance and species richness found within quadrats thus using a reduced species richness measure, as not all flowering plant species were found within the quadrats.

2.2.7 Bee–flower visitations

The total number of bee visitations per plant species using netting data from both years were counted and ranked. Furthermore, bee visitations were divided by the amount of open flowers per plant species found within quadrats on the same day to calculate bee visits per flower unit to provide a weighted importance of plant species considering their abundance (Russo et al., 2013). Bee visits per flower unit for flowers that were present, but did not occur within sampling quadrats remained unknown.

2.3 Statistical analysis

2.3.1 Comparison between field habitat types and season

All statistical analysis was performed using RStudio Version 1.0.44 (R Core Team, 2016). Generalized linear mixed-effect models (GLMMs) were applied to test for the impact of habitat type on bee and floral variables. The analysis was done separately for each year

given the differences in sampling intensity and for each sampling method. The lmer function

was used for response variables that conformed to a normal distribution and the glmer

function for response variables that could not be transformed to normal distribution (lme4

package, Bates et al., 2015). ANOVA tables using type III Wald tests (car package, John et

al., 2015) for each model with habitat type, season and their interaction as predictor variables

and site as a randomized block was computed. The interaction term was removed if it did not

significantly improve the model fit. Post hoc Tukey tests were used to determine significant

($p < 0.05$ and in a few cases $p < 0.10$) differences between habitat types, seasons and habitat

types within season for significant interaction terms, using the package multicomp (Bretz et

al., 2015). The sem.model.fits function (piecewiseSEM package, Lefcheck, 2016) was used

to investigate the percent of variance in the response variable explained by site as a random

effect by comparing marginal and conditional R^2 values (Lefcheck, 2016). For all models

with abundance and species richness as responses, a negative-binomial distribution with a

log-link function was used to account for over-dispersion in the data (Zuur et al., 2009).

Using the Akaike Information Criterion (AIC), gaussian and gamma distributions were

compared for modeling diversity using an identity or inverse link depending upon the

distribution. A small value (0.01) was added to zero values in order to run the model. Plots of

the residuals were used to check the assumptions of the models were met.

Bee and floral community composition among habitat types, sites and sampling

methods for each year were compared using a Nonmetric Multidimensional Scaling (NMDS)

analysis (vegan package, Oksanen et al., 2016). To test for statistical significance between

these communities, the adonis function with a Bray-Curtis distance distribution (vegan

package, Oksanen et al., 2016) was applied. Finally, to investigate the contribution of

geographic location on bee and floral community composition, similarities in species composition were correlated with geographical distances between sampling sites employing the Mantel test (vegan package, Oksanen et al., 2016) based on rank correlations and used a permutation test to calculate significance (Hannon and Sisk, 2009; Potts et al., 2003).

Due to differing sampling intervals between years, the data was grouped into three seasons (Early = April and May; Mid = June and July; Late = August) based on the main bee activity period (Tucker and Rehan, 2016), enabling an evaluation of how bee and floral variables changed temporally over distinct periods. Blueberry and potato crops are the most dominant crop types in Delta carrying flowers that attract foraging insects (Metro Vancouver, 2014; Zulian et al., 2013). Blueberry crops bloom in Early-season whereas potatoes bloom between Early- and Mid-season (West Coast Seeds, 2017). The temporal match between wild bees and resource supply by crop species most likely influences bee abundance (Brandt et al., 2017).

The coefficient of variation (CV) of visitation was used as a measure of the temporal variability in floral resources surveyed over the entire sampling year to determine the temporal stability of different habitat types (adapted from Orford et al., 2016). The CV was calculated by dividing the standard deviation among sampling days per plot, by the mean among sampling days per plot, using the raster package (Hijmans et al., 2016). The values are expressed in percentage where a higher value indicates a greater variation. A one-way ANOVA was applied to investigate if CV values differed between habitat types. Post hoc Tukey tests were used to determine significant ($p < 0.05$ and in a few select cases, $p < 0.10$) differences among habitat types and CV values using the package multicomp (Bretz et al., 2015).

2.3.2   Relationship between floral and bee variables

To investigate the relationship between floral resources and bee communities, a basic GLMM for each bee variable (bee abundance, species richness and diversity) containing week as a predictor variable and site as a randomized block using the glmer function (lme4 package, Bates et al., 2015) was set up. Each vegetation variable (floral abundance, species richness and diversity) was separately added to the basic models and compared in an ANOVA to assess whether the inclusion of the vegetation variable significantly improved the model fit. The AIC values of the models with different vegetation variables were compared to determine the best predictor variable for each bee variable. The sem.model.fits function (piecewiseSEM package, Lefcheck, 2016) was used to return goodness-of-fit statistics for the GLMM's to compare models among different response variables. Marginal and conditional R^2 values are a widely used measure of model fitness that can be interpreted as the percent of variance in the response variable explained by the model. The marginal R^2 values are associated with the fixed effects that are usually of interest, whereas conditional R^2 also include the random effects (Lefcheck, 2016). Only netted wild bees collected over two years were considered for this analysis due to the potential competition between floral resources and pan traps for bees (Morandin and Kremen, 2013). Honey bees were included in the analysis due to their high percentage within netting samples (52.38 %).

2.4   Results

2.4.1   Bee community variables

Altogether a total of 1317 individuals of 41 bee species in 16 genera and five families were collected. Out of the total amount, 594 or 43.5% was comprised of the managed honey

bee. The remaining 723 individuals of the 40 species were wild bees, however the Eastern

Bumble bee *Bombus impatiens* might have originated from greenhouses where they were

introduced for pollination from Eastern Canada (see Appendix A.3). The non-native *B.*

impatiens was the most common bumble bee species and accounted for 77 individuals or

10.62 % of all non-honey bee species sampled and was thereby the second most common

species after *Ceratina acantha* with 139 individuals or 19.17 %. Other non-native species to

BC included *Anthidium manicatum*, *Lasioglossum zonulum* and *Osmia caerulescens*. While

70 individuals of *Lasioglossum zonulum* were collected, only four individuals of *Anthidium*

manicatum and one individual of *Osmia caerulescens* were found. Three genera of Cuckoo

bees from three families were identified: *Nomada sp.*, *Sphecodes sp.* and *Coelioxys sp.* More

wild bees were collected with pan traps than nets in both years, 2015 (df= 1, $p < 0.05$) and

2016 (df= 1, $p < 0.001$).

Community analyses using Permutational Multivariate Analysis of Variance showed

that wild bee communities differed significantly between sampling methods in both 2015 (df

= 1, $p < 0.05$) and 2016 (df = 1, $p < 0.01$), but not among sites and habitat types. The Mantel

test showed no statistically significant correlation between the spatial proximity of sampling

sites and similarity in bee species composition.

2.4.2   Floral resources

Altogether, a total of 17 vegetation surveys were conducted in 2015 and 2016 at the

same time as each of the bee sampling rounds (either by net or pan trap). The data for two

sampling plots were removed as outliers due to unusual high flower counts recorded on the

first sampling day in 2015, which is likely, an observational error. In total 68 plant species

were recorded, only 12 were native to BC ("E-Flora BC: Electronic Atlas of the Flora of British Columbia," 2017) (Appendix A.2). Floral abundance was measured for a total of 42 plant species that were found within the quadrats. The most abundant species was Himalayan blackberry *Rubus armeniacus* (31.96 %), followed by Black hawthorn *Crataegus douglasii* (17.70 %) and Wild radish *Raphanus raphanistrum* (8.21%).

Community analyses using Permutational Multivariate Analysis of Variance showed that floral community composition did differ significantly between habitat types in 2015 (df = 2, p < 0.01) but not in 2016 and not among sampling sites in either year (Figure 2.2).
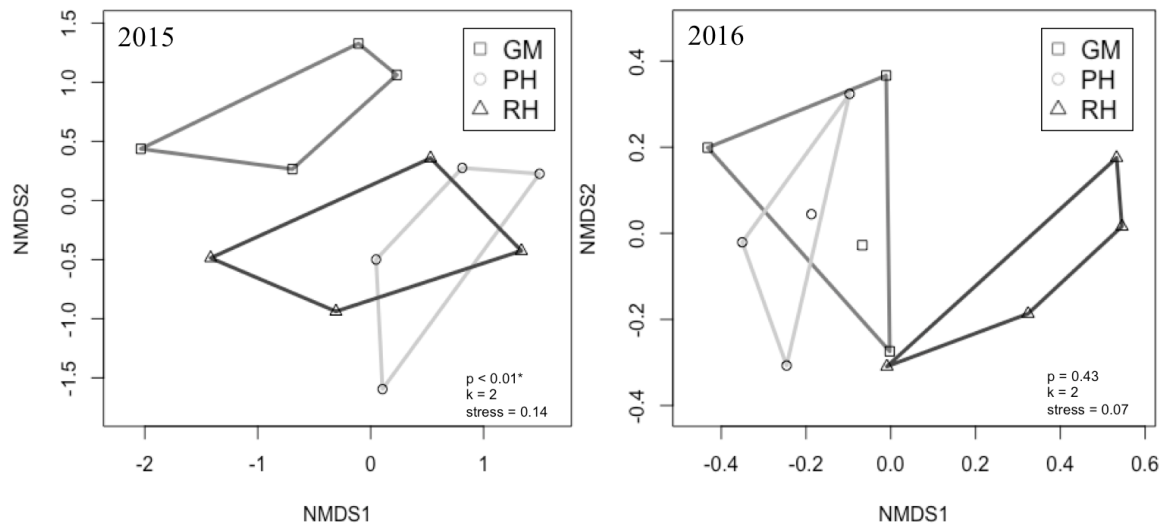
Figure 2.2: Three-dimensional NMDS ordination diagrams of plant species assemblages from 12 different sampling plots by habitat type in 2015 and 2016. Data from Early-season, Mid-season, and Late-season surveys have been combined in each year to describe the cumulative plant species assemblage occurring in a single sampling plot. Plot symbols closest together in the diagram had the most similar assemblages, in terms of both species composition and species relative abundance. A significant difference (p < 0.01) between plant species assemblages was found between habitat types in 2015, no significant difference was found in 2016.

The Mantel test showed a general pattern of decreasing similarity in floral composition with increasing distance between sampling sites but not at a statistically significant level.

2.4.3 Comparison between field habitat types and season

Habitat type was shown to have a significant impact on netted and pan-trapped wild bees and floral variables in both years. Season had a significant impact on netted and pan-trapped wild bees in 2015 and netted wild bees only in 2016, but was not found to impact floral variables in either year. Significant interactions between habitat type*season were found for pan-trapped wild bee species diversity and floral abundance (m^2 and plot-level) in 2015 and netted wild bee species diversity and netted honey bee abundance in 2016. The percent of variance in the response variable explained by site as a random effect, was < 0.04 % for all bee and floral variables in 2015. Site had a greater influence in 2016 where 37 % of variance in pan-trapped wild bee abundance and 36% of variance in floral abundance m^2 was explained by site, while the site effect on the remaining bee and floral variables was comparably smaller < 0.06 % (Table 2.3).

Table 2.3: Analysis of Deviance Table (Type III Wald chi-square tests) for netted (n=152) and pan-trap collected (n=570) wild bee abundance, species richness and diversity as well as floral abundance, species richness and diversity in planted hedgerows, remnant hedgerows and grass margin sampling plots (n=12) in Early-, Mid- and Late season in 2015 and 2016. Significant results are highlighted in bold for p<0.05 and * for p<0.10.

Year	Variable	Habitat type			Season			Habitat type * Season		
		chisq	df	p	chisq	df	p	chisq	df	p
2015	Netted Wild Bee Abundance	17.09	2	**<0.01**	3.47	2	0.17	4.19	4	0.38
	Netted Wild Bee Species Richness	5.80	2	0.06*	6.50	2	**<0.05**	2.72	4	0.60
	Netted Wild Bee Species Diversity	1.16	2	0.55	19.00	2	**<0.01**	5.49	4	0.24
	Pan-trapped Wild Bee Abundance	5.87	2	0.05*	10.83	1	**<0.01**	0.004	2	0.99
	Pan-trapped Wild Bee Species Richness	8.69	2	**0.01**	16.12	1	**<0.01**	0.00	2	0.95
	Pan-trapped Wild Bee Species Diversity	15.78	2	**<0.01**	5.48	1	**<0.05**	5.66	2	0.06*
	Floral Abundance (m^2)	4.05	2	0.13	0.09	2	0.95	17.61	4	**<0.01**
	Floral Abundance (plot-level)	4.60	2	0.09*	0.01	2	0.99	19.36	4	**<0.01**
	Floral Species Richness	11.14	2	**<0.01**	1.92	2	0.38	2.62	4	0.62
	Floral Species Diversity	5.35	2	0.06*	0.05	2	0.58	1.50	4	0.25
2016	Netted Wild Bee Abundance	1.61	2	0.44	7.81	2	**<0.05**	7.30	4	0.12
	Netted Wild Bee Species Richness	1.25	2	0.53	4.76	2	0.09*	4.94	4	0.29
	Netted Wild Bee Species Diversity	0.00	2	0.99	21.87	2	**<0.01**	16.16	4	**<0.01**
	Pan-trapped Wild Bee Abundance	2.21	2	0.33	3.51	2	0.17	1.09	4	0.89
	Pan-trapped Wild Bee Species Richness	5.12	2	0.07*	2.08	2	0.35	0.51	4	0.97
	Pan-trapped Wild Bee Species Diversity	7.07	2	**0.03**	0.81	2	0.66	1.43	4	0.83
	Floral Abundance (m^2)	0.58	2	0.74	1.38	2	0.50	2.11	4	0.71
	Floral Abundance (plot-level)	10.10	2	**<0.01**	0.66	2	0.71	1.01	4	0.90
	Floral Species Richness	6.69	2	**0.04**	3.81	2	0.14	3.08	4	0.54
	Floral Species Diversity	8.67	2	**0.01**	2.03	2	0.36	1.60	4	0.81
2015	Netted Honey Bee Abundance	0.22	2	0.89	9.22	2	**<0.01**	4.98	4	0.29
	Pan-trapped Honey Bee Abundance	2.37	2	0.30	2.84	1	0.09*	0.91	2	0.63
2016	Netted Honey Bee Abundance	5.64	2	0.06*	3.83	2	0.14	9.96	4	**0.04**
	Pan-trapped Honey Bee Abundance	3.81	2	0.15	13.15	2	**<0.01**	3.73	4	0.44

In 2015 wild bee abundance was 63.4 % higher in grass margins than in planted hedgerows and 76.8 % higher than in remnant hedgerows for net-collected specimens. Grass margins also had more than double the wild bee species of remnant hedgerows but were not significantly different than planted hedgerows. For pan-trapped specimens, grass margins had a 57.4 % higher bee abundance and a 65.5 % higher species richness than planted hedgerows but did not differ significantly from remnant hedgerows. The overall pan-trapped wild bee diversity was 66 % higher in grass margins than in planted hedgerows and 54.3 % higher than in remnant hedgerows. However, no difference between grass margins and

remnant hedgerows was found in Mid-season. Floral abundance per m^2 did not differ significantly between habitat types, but at the plot-level remnant hedgerows were found to have more than four times the floral abundance compared to grass margins, with no significant difference ($p > 0.10$) compared to planted hedgerows. However, this difference was mostly due to a high floral abundance in Early-season as there were no significant differences between habitat types in Mid-season. By Late-season remnant and planted hedgerows had significantly lower floral abundance per m^2 compared to grass margins ($z = -3.29$, $p < 0.05$ and $z = -3.72$, $p < 0.001$ respectively) and floral abundance at the plot-level was significantly lower for remnant hedgerows compared to both grass margins ($z = -3.5$, $p < 0.01$) and planted hedgerows ($z = -2.5$, $p < 0.05$). Grass margins in that year also had 41.3 % more floral species and a 69.9 % higher floral diversity than remnant hedgerows but did not differ significantly from planted hedgerows.

In 2016 no difference between habitat types was found for net-collected bees. However, net-collected bee diversity was significantly higher for planted hedgerows ($z = 2.45$, $p < 0.05$) and remnant hedgerows ($z = 2.62$, $p < 0.05$) in Late-season compared to grass margins. For pan-trapped specimens, grass margins had 43.3 % more wild bee species and a 43.6 % higher bee diversity compared to planted hedgerows but did not differ significantly from remnant hedgerows. Floral abundance per m^2 did not differ significantly between habitat types, but at the plot-level remnant hedgerows were found to have three times and planted hedgerows four times the floral abundance compared to grass margins. The highest floral species richness was also detected in planted hedgerows with 31.6 % more species than remnant hedgerows but no significant difference compared to grass margins. Floral diversity

in planted hedgerows was 47.5 % higher than in grass margins but did not differ significantly

from remnant hedgerows (Table 2.4).

Table 2.4: Wild bee and floral abundance, species-richness and diversity by habitat type in the ALR in Delta, over two years. Values, shown as mean ± standard error, significant differences are indicated with lower case letters.

Year	Variable	Grass margin		Planted Hedgerow		Remnant Hedgerow	
2015	Netted Wild Bee Abundance	4.10±1.21	a	1.50±0.31	b	0.95±0.28	b
	Netted Wild Bee Species Richness	1.70±0.38	a	1.20±0.24	ab	0.80±0.20	b
	Netted Wild Bee Species Diversity	0.31±0.38		0.28±0.24		0.15±0.20	
	Pan-trapped Wild Bee Abundance	6.25±1.65	a	2.66±1.29	b	3.25±1.38	ab
	Pan-trapped Wild Bee Species Richness	3.83±1.01	a	1.33±0.54	b	1.75±0.71	ab
	Pan-trapped Wild Bee Species Diversity	0.94±0.24	a	0.32±0.14	b	0.43±0.19	b
	Floral Abundance (m^2)	4.73±1.57		2.78±0.92		5.21±3.43	
	Floral Abundance (plot-level)	946.25±315.88	b	2123.10±703.32	ab	4085.35±2693.89	a
	Floral Species Richness	3.62±0.34	a	2.62±0.27	ab	2.16±0.31	b
	Floral Species Diversity	0.23±0.06	a	0.22±0.05	a	0.07±0.03	b
2016	Netted Wild Bee Abundance	0.63±0.23		1.05±0.57		1.26±0.34	
	Netted Wild Bee Species Richness	0.47±0.19		0.52±0.19		0.73±0.15	
	Netted Wild Bee Species Diversity	0.09±0.07		0.08±0.06		0.07±0.05	
	Pan-trapped Wild Bee Abundance	9.56±2.52		7.38±3.05		7.19±2.20	
	Pan-trapped Wild Bee Species Richness	4.62±0.86	a	2.62±0.70	b	2.68±0.47	ab
	Pan-trapped Wild Bee Species Diversity	1.12±0.20	a	0.62±0.16	b	0.72±0.13	ab
	Floral Abundance (m^2)	6.08±1.54		7.42±2.23		5.33±1.23	
	Floral Abundance (plot-level)	1216.00±308.58	b	5664.92± 706.57	a	4178.24±967.71	a
	Floral Species Richness	4.31±0.59	ab	5.29±0.51	a	3.62±0.22	b
	Floral Species Diversity	0.21±0.05	b	0.40±0.06	a	0.30±0.58	ab
2015	Netted Honey Bee Abundance	2.65±0.98		2.10±0.81		1.85±0.56	
	Pan-trapped Honey Bee Abundance	2.58±1.54		1.33±0.82		0.42±0.23	
2016	Netted Honey Bee Abundance	2.89±0.86	b	5.21±2.12	a	3.52±1.33	ab
	Pan-trapped Honey Bee Abundance	4.56±1.32		1.88±0.89		4.06±1.57	

In 2015 net-collected bee species richness and diversity increased from Early- to Late-season by 51.2 % and 92.9 % respectively. Pan-trapped bee abundance on the other hand declined by 75.4 %, species richness by 87.4 % and diversity by 97.6 % from Mid-season to Late-season. In 2016 net-collected bee abundance and species richness were increasing from Early- to Late-season by 76.9 % and 65.1 % respectively. Overall no significant differences between seasons were found for floral variables. Floral abundance decreased from Early- to Late-season, whereas floral species richness increased over the seasons in both years (Table 2.5).

Table 2.5: Wild bee and floral abundance, species-richness and diversity by season in the ALR in Delta, over two years. Values, shown as mean ± standard error, significant differences are indicated with lower case letters. No data (ND) was collected in Early-season for pan-trapped wild bees.

Year	Variable	Early Season (April & May)		Mid Season (June & July)		Late Season (August)	
2015	Netted Wild Bee Abundance	2.33±1.70		1.46±0.37		2.83±0.69	
	Netted Wild Bee Species Richness	**0.83±0.24**	**b**	**0.96±0.21**	**ab**	**1.70±0.33**	**a**
	Netted Wild Bee Species Diversity	**0.03±0.03**	**b**	**0.18±0.07**	**a**	**0.42±0.11**	**a**
	Pan-trapped Wild Bee Abundance	ND		**5.41±1.11**	**a**	**1.33±0.89**	**b**
	Pan-trapped Wild Bee Species Richness	ND		**3.25±0.62**	**a**	**0.41±0.22**	**b**
	Pan-trapped Wild Bee Species Diversity	ND		**0.82±0.12**	**a**	**0.07±0.04**	**b**
	Floral Abundance (m^2)	14.70±9.45		3.65±1.10		1.81±0.61	
	Floral Abundance (plot-level)	10507.65±7478.52		1921.12±448.78		474.07±134.45	
	Floral Species Richness	2.40±0.50		2.67±0.25		3.14±0.32	
	Floral Species Diversity	0.18±0.11		0.19±0.04		0.18±0.04	
2016	Netted Wild Bee Abundance	**0.48±0.20**	**b**	**0.88±0.20**	**ab**	**2.08±0.94**	**a**
	Netted Wild Bee Species Richness	**0.29±0.10**	**b**	**0.70±0.18**	**ab**	**0.83±0.27**	**a**
	Netted Wild Bee Species Diversity	0		0.13±0.06		0.13±0.08	
	Pan-trapped Wild Bee Abundance	13.83±4.53		6.00±1.38		6.33±2.30	
	Pan-trapped Wild Bee Species Richness	4.00±0.80		2.83±0.55		3.58±0.97	
	Pan-trapped Wild Bee Species Diversity	0.96±0.17		0.76±0.14		0.83±0.24	
	Floral Abundance (m^2)	8.17±2.50		5.30±1.13		5.66±1.41	
	Floral Abundance (plot-level)	4584.32±1755.84		3172.00±743.14		3435.47±957.77	
	Floral Species Richness	3.69±0.42		4.58±0.43		5.00±0.56	
	Floral Species Diversity	0.26±0.06		0.33±0.05		0.32±0.07	
2015	Netted Honey Bee Abundance	**4.91±1.32**	**a**	**1.71±0.52**	**b**	**1.33±0.68**	**b**
	Pan-trapped Honey Bee Abundance	NA		**1.83±0.82**	**a**	**0.67±0.67**	**b**
2016	Netted Honey Bee Abundance	6.47±2.05		3.00±0.85		1.08±0.45	
	Pan-trapped Honey Bee Abundance	**7.91±1.80**	**a**	**2.95±0.90**	**b**	**0.16±0.16**	**c**

The CV of floral abundance was significantly greater (p < 0.10) for remnant hedgerows in 2015 (df = 2, p < 0.10) than grass margins, but not planted hedgerows. CV values for species richness and diversity did not differ significantly. In 2016, again no significant differences for CV were found (Table 2.6).

Table 2.6: Results of Multiple Comparisons of Means: Tukey Contrasts tests of Coefficient of variation (CV) in % of floral variables by habitat type in the in the ALR in Delta, over two years. Values, shown as mean ± standard error, significant differences are indicated with lower case letters.

Year	Variable	Grass margin		Planted Hedgerow		Remnant Hedgerow	
2015	Floral Abundance (m^2 and plot-level)	**124.69±22.23**	**b**	**146.12±32.10**	**ab**	**223.76±29.55**	**a**
	Floral Species Richness	42.85± 3.00		44.05± 6.54		60.64±10.37	
	Floral Species Diversity	144.62±14.95		161.72±44.50		196.66±60.94	
2016	Floral Abundance (m^2 and plot-level)	187.30±47.64		177.40±36.79		126.26±18.80	
	Floral Species Richness	60.14±19.97		38.75± 5.75		28.99± 5.26	
	Floral Species Diversity	92.21±13.07		123.98±40.26		146.07±54.67	

2.4.4 Relationship between floral and bee variables

Several floral variables improved the basic model only containing week as a fixed and site as a random effect significantly. However, the best bee abundance model included floral abundance (AIC =597.5, p < 0.001), whereas the best bee species richness and bee species diversity model included floral species richness (AIC = 358.9, p < 0.01) and (AIC = 139.6, p < 0.001). The best model fit, determined by highest Marginal R^2 values, was found for the model with bee abundance as a response and floral abundance (m^2) as a fixed variable ($R^2 = 0.58$) (Table 2.7).

Table 2.7: Generalized linear models (Negative binomial or Gaussian error distribution) examining effects of floral abundance, floral species richness and diversity on bee abundance, bee species richness and diversity. Among each bee response variable models were compared in an ANOVA to determine the best model with the highest AIC value. Goodness-of-fit statistics for theses models were explored to identify the model with the highest Marginal R^2 value among different response variables.

Response Variable	GLMM	AIC	p	Cond. R^2	Marg R^2
Bee abundance	Bee abundance~ week + (1|site)	603.6			
	Bee abundance~ week + floral abundance (m^2) + (1|site)	597.5	**<0.01**	**0.58**	**0.58**
	Bee abundance~ week + floral abundance (plot) +(1|site)	597.8			
	Bee abundance~ week + floral species richness + (1|site)	602.3			
	Bee abundance~ week + floral diversity + (1|site)	601.6			
Bee species richness	Bee species richness~ week + (1|site)	363.0			
	Bee species richness ~week + floral abundance (m^2) + (1|site)	364.1			
	Bee species richness ~week + floral abundance (plot) +(1|site)	364.1			
	Bee species richness ~week + floral species richness + (1|site)	358.9	**0.01**	**0.17**	**0.17**
	Bee species richness ~week + floral diversity + (1|site)	362.1			
Bee species diversity	Bee species diversity~ week + (1|site)	141.8			
	Bee species diversity ~week + floral abundance (m^2) + (1|site)	364.1			
	Bee species diversity ~week + floral abundance (plot) +(1|site)	143.5			
	Bee species diversity ~week + floral species richness + (1|site)	139.6	**<0.01**	**0.18**	**0.16**
	Bee species diversity ~week + floral diversity + (1|site)	142.3			

2.4.5 Bee–flower visitations

During both field seasons 17 out of the total of 68 encountered plant species were visited by wild bees. Only five out of the 17 plant species are native to BC and recommended

for hedgerow restoration by the DF&WT. The remaining 12 species are all introduced

species, most of which are herbaceous except for *Rubus armeniacus*.

Visitation patterns evaluated using netting data indicate that the most visited plant

species was *Vicia cracca* with a total of 51 observed visits, followed by *Rubus armeniacus*

with 32 bee visitations and *Symphoricarpos albus* with 16 bee visitations. Most bee genera

were found on *Rubus armeniacus* with six different bee genera, while some plant species

were exclusively visited by only one genus (Figure 2.3).
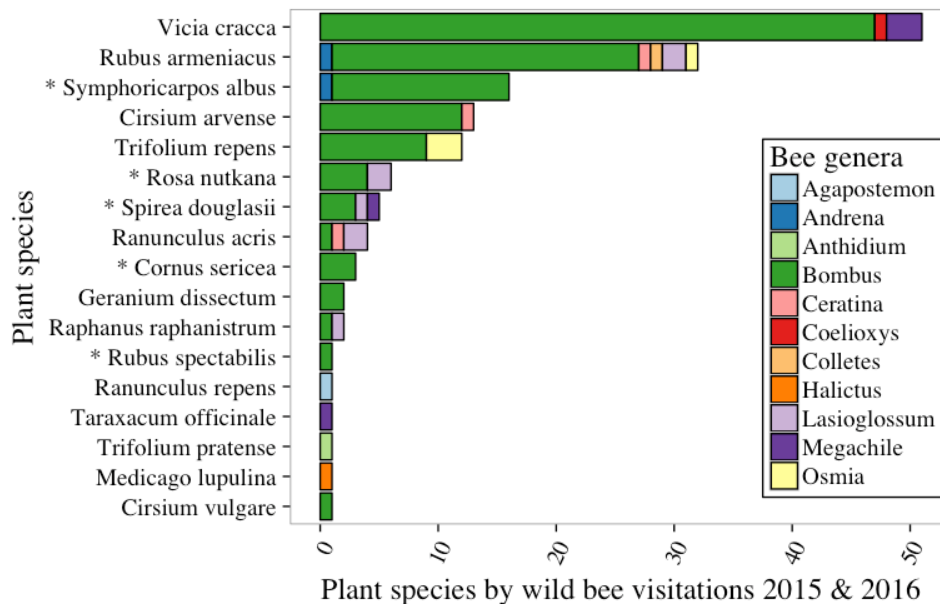
Figure 2.3: Number of wild bee visitations (n = 152) by plant species and bee genera from
surveys in 2015 and 2016. Native plant species are indicated with *.

Floral abundance data within quadrats for all 17 plant species visited were available

for the same day of the bee visitations. The most bee visitations per flower unit were counted

on *Rubus armeniacus* (2.67), followed by *Vicia cracca* (2.5) and *Trifolium pratense* (1.30)

(Table 2.8).

Table 2.8: Wild bee visitations per plant species (n= 152) recorded from ten net sampling days and weighted by the observed floral abundance within quadrats (5 m^2) on those sampling days to provide a weighted rank in comparison to the rank based solely on number of flower visitations. Native plant species are indicated with *.

Weighted Rank	Ranks based on flower visitations	Genus	Species	Bee visitations per flower unit
1	2	*Rubus*	*armeniacus*	2.67
2	1	*Vicia*	*cracca*	2.50
3	15	*Trifolium*	*pratense*	1.30
4	7	*Spirea*	*douglasii**	0.77
5	8	*Ranunculus*	*acris*	0.75
6	3	*Symphoricarpos*	*albus**	0.56
7	4	*Cirsium*	*arvense*	0.37
8	9	*Cornus*	*sericea/ stolonifera**	0.30
9	11	*Raphanus*	*raphanistrum*	0.10
10	6	*Rosa*	*nutkana**	0.05
11	13	*Ranunculus*	*repens*	0.03
12	5	*Trifolium*	*repens*	0.02
13	17	*Cirsium*	*vulgare*	0.10
14	12	*Rubus*	*spectabilis**	<0.10
15	13	*Taraxacum*	*officinale*	<0.10
16	16	*Medicago*	*lupulina*	<0.10
17	10	*Geranium*	*dissectum*	<0.10

The average floral abundance per m^2 of the 17 plant species visited by wild bees by habitat type revealed that habitat types with the highest floral abundance per square meter aligned with the highest net-collected wild bee abundance in both years (Figure 2.4).
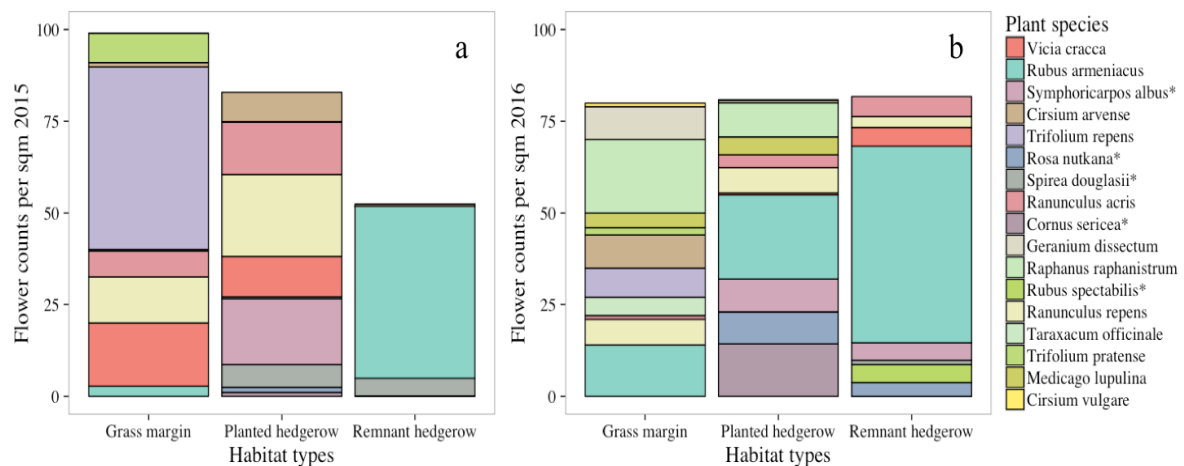
Figure 2.4: Plant species visited by wild bees by habitat types in 2015 (a) and 2016 (b).

2.4.6   Honey bees

Honey bees were analyzed separately for their habitat- and flower preferences and seasonal change in abundance. Honey bees did not show a clear preference for a habitat type, except in 2016, when significantly more netted honey bees were found in planted hedgerows overall (Table 2.4), however no difference between habitat types was found in Mid-season. For all samples, net-collected as well as pan-trapped, the seasonal pattern was similar so that honey bee abundance steadily decreased from Early- to Late-season (Table 2.5). No negative correlation between honey bee abundance and wild bee abundance, species richness or diversity was found.

During both field seasons 19 out of the total of 68 encountered plant species were visited by honey bees. Honey bees visited six plant species that were not observed to be visited by wild bees; *Rubus laciniatus, Rubus parviflorus, Malus fusca, Viburnum edule, Crataegus monogyna* and *Senecio vulgaris*. On the other hand, honey bees were not found on four species that wild bee visits were recorded; *Trifolium pratense, Taraxacum officinale, Rubus spectabilis* and *Geranium dissectum*. Seven out of the 19 plant species are native to BC and recommended for hedgerow restoration by the DF&WT. The remaining 12 species were all introduced species, most of which are herbaceous except for *Rubus armeniacus*, *Rubus laciniatus* and *Crataegus monogyna*. The most visited plant species was *Rubus armeniacus* with a total of 97 visitations, followed by *Crataegus monogyna* and *Rosa nutkana* with both 46 visitations (Figure 2.5).
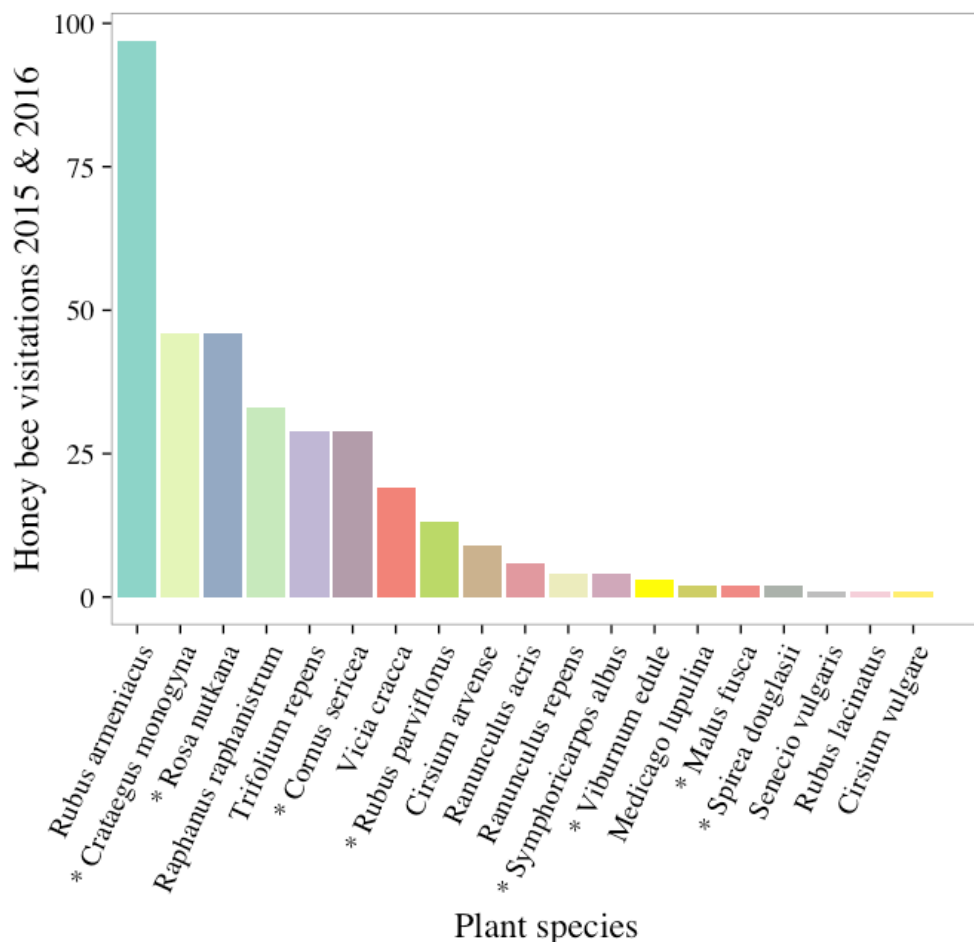
Figure 2.5: Plant species ranked by number of total honey bee visitations from surveys in 2015 and 2016. Native plant species are indicated with *.

The most honey bee visitations per flower unit were counted on *Rosa nutkana* (42.8), followed by *Rubus armeniacus* (31.2) and *Cornus sericera* (14.5) Floral abundance data within quadrats were not available for five out of the 19 plant species visited for the same day of the bee visitations, therefore honey bee-flower visitations per flower unit for *Malus fusca*, *Viburnum edule*, *Medicago lupulina*, *Crataegus monogyna* and *Senecio vulgaris* remained unknown (Table 2.9).

Table 2.9: Honey bee visits per plant species (n= 264) recorded for ten net sampling days and weighted by the observed floral abundance within quadrats (5 m^2) on those sampling days to get a weighted rank in comparison to the rank based solely on number of flower visitations. Native plant species are indicated with *.

Weighted Rank	Rank based on flower visitations	Genus	Species	bee visits per flower unit
1	3	*Rosa*	*nutkana**	42.75
2	1	*Rubus*	*armeniacus*	31.18
3	6	*Cornus*	*sericea**	14.50
4	9	*Cirsium*	*arvense*	5.79
5	4	*Raphanus*	*raphanistrum*	5.58
6	5	*Trifolium*	*repens*	3.83
7	10	*Ranunculus*	*acris*	3.01
8	7	*Vicia*	*cracca*	1.72
9	16	*Spirea*	*douglasii**	1.14
10	12	*Symphoricarpos*	*albus**	0.54
11	19	*Cirsium*	*vulgare*	0.50
12	18	*Rubus*	*laciniatus*	0.25
13	8	*Rubus*	*parviflorus**	0.22
14	11	*Ranunculus*	*repens*	0.08
unknown	2	*Crataegus*	*monogyna*	unknown
unknown	13	*Viburnum*	*edule**	unknown
unknown	14	*Medicago*	*lupulina*	unknown
unknown	15	*Malus*	*fusca**	unknown
unknown	17	*Senecio*	*vulgaris*	unknown

2.5   Discussion

2.5.1   Bee community variables

The species richness and species community composition observed in this study was, for the most part, within the range found in previous studies in natural habitats in Delta and the Lower Fraser Valley. Mackenzie and Winston (1984) recorded 48 wild bee species on five natural sites across the Lower Fraser Valley in 1981 and 1982. In 2010 Parkinson and Heron (2010) recorded 30 wild bee species on 64 natural sites along farm boundaries in the Lower Fraser Valley. Out of those 64 sites, 16 were located within the ALR in Delta where 21 wild bee species were found. In both cases the most common species was *Bombus mixtus*, followed by *Bombus vosnesenskii,* and *Bombus rufocinctus* (Mackenzie and Winston, 1984; Parkinson and Heron, 2010). In this study from 2015 and 2016, a total of 40 wild bee species

were collected. The most common species found was *Ceratina acantha*. Notably, the non-native *Bombus impatiens* with 77 individuals or 10.6 % was the second most common species, followed by *Bombus mixtus* and *Bombus flavifrons. Bombus impatiens*, which has been imported to BC for use as a commercial pollinator in greenhouses starting in 1999 (Parkinson and Heron, 2010), has significantly increased its abundance since it was first recorded outside of greenhouses in 2003 and 2004. The presence of *Bombus impatiens* outside of greenhouses in BC was cause for concern, due to the potential of future colonies being established by the queen and the risks associated with the transmissions of pathogens to wild congeners (Ratti and Colla, 2010). Disease spill over from managed to wild bees is considered one of the major threats to the declining members of the subgenus *Bombus*, such as the threatened and increasingly scarce *Bombus occidentalis* (Evans et al. 2008, NRC 2007, Ratti and Colla, 2010). *Bombus occidentalis* was a common pollinator in the early 1980's and made up to 25-30 % of bumble bees collected on berry farms in the lower mainland (Mackenzie and Winston, 1984). *Bombus occidentalis* has been listed under the Species at Risk Act in 2014 by the Committee on the Status of Endangered Wildlife (COSEWIC, 2017) and was not detected in this study.

Lasioglossum zonulum was the second most common non-native wild bee species found with 70 individuals. Even though records in North America go back many years, it is possibly an introduced rather than a native species originating from Europe and south-east China (Droege, 2012). Furthermore, four specimen of *Anthidium manicatum* were detected. *Anthidium manicatum* is native to Europe, Asia, North Africa, but is now considered the world's most widespread unmanaged bee species (Hicks, 2011). *Anthidium manicatum* males have been documented to show aggressive foraging behavior and thus might impact native

bee fauna by deterring them from foraging (Hicks, 2011). Only one individual of *Osmia caerulescens* was collected. *Osmia caerulescens* is native to Europe, North Africa and India. According to Droege (2012) it appears to be less common than it once was in southern Canada.

Three genera of Cuckoo bees from three families were identified. Cuckoo bees are cleptoparasites that lay their eggs in the nests of specific host bees, where their larvae consumes the host's bee larvae and food (Sheffield et al., 2013). According to Sheffield et al. (2013) cleptoparasitic bees particularly are a promising indicator taxa as they are the first guild to respond to disturbances. Cuckoo bees identified included *Nomada sp.* (parasitize *Andrena sp*. and *Agapostemon sp*.), *Sphecodes sp.* (parasitize other *Halictidae sp*.) and *Coelioxys sp*. (parasitize *Megachile* sp.) (Parkinson and Heron, 2010). The record for *Nomada sp.* was lost therefore it remains unknown in what habitat type, on which plant species or pan trap respectively it was collected.

2.5.2 Comparison between field habitat types

In 2015, grass margins were found to have the highest values in bee abundance, species richness and diversity for both net-collected and pan trapped bees, as well as for floral abundance, species richness and diversity (except floral abundance on the plot-level, due to differences in plot size for hedgerows). In 2016, the pattern was not as clear. Net-collected bee abundance and richness were highest in remnant hedgerows, whereas floral abundance and species richness were highest in planted hedgerows. Pan-trapped bees were again most abundant and species-rich in grass margins.

Different weather patterns during the two sampling years in 2015 and 2016 might have impacted differences in vegetation and bee variables in general. While in 2015, April and August had very high, above average precipitation, May, June and July were extraordinarily dry. In 2016, on the contrary, April, May and August received significantly less precipitation while June and July were wetter compared to 2015. The average seasonal temperature in 2015 was 1.8 (° C) higher with higher temperatures throughout all months except April and August compared to 2016. Weather events such as floods and droughts affect local bee communities in different ways (Goulson et al., 2015). Heavy precipitation is likely harmful to bee species that nest or hibernate underground and deters bees from flying (Goulson et al., 2015; LeBuhn et al., 2016). A lack of precipitation on the other hand can limit floral resources as flowering is triggered by the amount of rainfall and varies between plant species (Minckley et al., 2013). Bee species that are generalists are expected to deal better with drought effects as they collect pollen from many plant species and are thus not as dependent on a single floral host species as specialist bee species (Minckley et al., 2013).

Bees are also highly impacted by temperature in their development, survival, range and abundance (Bale et al., 2002). Papanikolaou et al. (2016) found bee diversity to be highly sensitive to temperature with an increase in temperature leading to a decrease in bee diversity, especially when a threshold of high temperature is surpassed. Changing flowering phenology attributed to climate change can decouple plant-bee interactions as bee species and their target flowers can respond differently to climatic clues (Scholes, 2016). While some species can locally adapt to local climate conditions, short-term temperature deviations around this longer-term mean might drive rapid declines (Papanikolaou et al., 2016). While in both years, monthly average temperatures were above the 1981-2000 climate normal, 2015

revealed higher variations around this longer-term mean. Differences in temperature and precipitation combined with one additional sampling round and simultaneously better skilled volunteers in 2016, likely contributed to the differences in bee and floral patterns observed between years.

A significant, positive relationship between abundant and diverse floral resources and net-collected bee abundance species richness and diversity was only found when including honey bees (Nicholls & Altieri, 2013). While the strongest relationship between floral abundance and net-collected bee abundance, including honey bees, did explain 58% of the variation in bee abundance, it did not explain the differences between habitat types alone. Floral abundance included the abundance of all 42 floral species found within the quadrats, whereas bees visited only 23 of those species. Thus, when excluding honey bees and analyzing floral abundance of floral species visited by wild bees only, the habitat types with the highest net-collected wild bee abundance aligned with highest floral abundance per square meter in both years.

Another indicator for higher net-collected bee abundance and richness might be the temporal variability in floral resources over the sampling season. Habitat types with the lowest CV values, hence with the most stable floral resources over time, could be contributing to the higher net-collected bee abundance seen in those habitats in both years. The importance of a continuous supply of floral resources for bee communities throughout the season has been stated by numerous studies (Oertli et al., 2005; Pfiffner and Müller, 2014; Scheper et al., 2015; Woodcock et al., 2016).

On the contrary, pan-trapped bees were most abundant in grass margins regardless of floral resources or the continuity thereof. Morandin and Kremen (2013) suggested that pan

traps may not be an accurate way to assess differences in pollinator communities between sites that differ in floral resources. It is likely that more bees were attracted to pan traps when few flowers were available (Morandin and Kremen, 2013), however no negative correlation between floral resources and pan-trapped bee abundance was found. Pan traps collected a significantly different and more species rich wild bee community than communities captured by net sampling from flowers confirming the sampling bias associated by the sampler's netting skills towards larger bee species (Roulston et al., 2007). Bees from the genus *Andrena*, *Agapostemon*, *Ceratina*, *Halictus*, *Lasioglossum*, *Megachile*, *Melissodes*, *Osmia* and *Sphecodes* were almost exclusively found in pan traps, while only the number of honey bees collected was higher and no genus was exclusively collected with nets. In general, a combination of pan traps and net collecting is recommended to extensively survey pollinator diversity (Moreira et al., 2016; Roulston et al., 2007). However, in a protocol designed to detect and monitor pollinator communities published by the Food and Agriculture Organization of the United Nations (FAO) in 2016, pan traps are recommended as the common, inexpensive and easy to standardize methodology for a global monitoring programme (LeBuhn et al., 2016).

Surprisingly, planted hedgerows did not contribute floral resources to local bee communities as expected. In 2015 significantly less net-collected bees were found in planted hedgerows compared to grass margins. In 2016 no difference was found between habitat types, however more net-collected bees were found in remnant hedgerows, portraying planted hedgerows as mediocre quality habitats for wild bees. The reason for the low performance of planted hedgerows might be due to the species composition of planted hedgerows. While planted hedgerows mostly consisted of native shrub and tree species, wild

bees showed a preference for herbaceous plant species that were naturally more common in grass margins, especially in 2015. Additionally, *Rubus armeniacus* was shown to be the most visited plant species per floral unit, which was the dominant plant species found in remnant hedgerows in both years. *Rubus armeniacus* also attracted six different bee genera consisting of 12 species, which was the highest number out of all plant species (Appendix A.3). These results differ from findings by Morandin and Kremen (2013) where wild bees collected from flowers at restored hedgerow sites were more abundant, species-rich, and diverse compared to weedy, unmanaged field edges.

Lastly, the temporal variability of floral resources (CV) in planted hedgerows was higher than grass margins in 2015 and higher than remnant hedgerows in 2016. One of the main challenges in habitat restoration in modern agro-ecosystems is to create alternative forage before, during and after the bloom of the main crop (Nicholls and Altieri, 2013). While monocultures of bee-pollinated crops, usually bloom in synchronous periods of only a few weeks, flight periods of many bees extend beyond the availability of these flowers (Nicholls and Altieri, 2013). A lack of season-long food sources can result in a decline of some pollinator species (Goulson, 2003). Especially for bumble bee queens that remain dormant throughout winter, food supply in early spring is crucial (Packer et al., 2007; Willmer, 2011). Early- and Mid- season (April to July) in this study coincides with the blooming period of blueberries and potatoes, which are the predominant crop types in Delta (Ministry of Agriculture, 2012b, 2017; Pesticide Risk Reduction Program et al., 2007). Hence, no data for "before" main crop bloom is available. However, fluctuations during the bloom and after bloom were recorded. Planted hedgerows may have indirectly contributed to

local netted bee diversity by providing Late-season forage in 2016, but no other results indicating positive interactions were found.

Despite the many demonstrated benefits of hedgerow restoration in agricultural landscapes, findings from this study suggest that hedgerow plantings in Delta could be optimized to better support local wild bee communities by adjusting the plant species composition to attain higher floral species richness and guarantee steadiness of flower provision throughout the season (see Ch 4.2 Recommendations).

2.6   Conclusions

Wild bees passively collected in pan traps were significantly more abundant, species-rich and diverse in grass margins than in planted and remnant hedgerows in both years. Abundance and species richness of wild bees collected with nets was significantly higher for grass margins in 2015 as well, but no difference between field margin types was found in 2016. Grass margins had the highest floral abundance per m^2 in 2015 and a significantly higher floral species richness compared to both hedgerow sites, whereas in 2016 planted hedgerows were found to have a higher abundance and species richness.

The sampling season in 2015 was characterized by long periods of low precipitation combined with high monthly average temperatures, which likely contributed to the decrease in floral species richness therefore limiting floral resources. The strongest, significant relationship was found between floral abundance and net-collected bee abundance, including honey bees, which explained 58 % of the variation in bee abundance. When selecting the floral abundance of floral species visited by wild bees exclusively, field margin types with the highest net-collected wild bee abundance aligned with highest floral abundance in both

years. Furthermore, field margin types offering the most stable floral resources throughout the season were also found to have the highest net-collected bee abundance in both years.

Wild bees showed a preference for herbaceous flowers. Eleven out of the 17 floral species observed to be visited by wild bees were herb species and only five of the 16 were recommended for hedgerow restoration by the DF&WT. These results indicate that floral abundance or floral species richness alone are not sufficient indicators for wild bee communities, but that plant species selection must be chosen specifically to target wild bees. Hedgerow plantings could be optimized to better support local wild bee communities by adjusting the plant species composition to attain higher floral species richness and guarantee steadiness of flower provision throughout the season. The inclusion of grassy strips as a buffer along planted hedgerows in the DF&WT Hedgerow Stewardship Program could be a valuable addition for wild bee conservation in Delta.

Chapter 3: The role of field margins for wild bee conservation at the landscape-scale

3.1 Introduction

The abundance or diversity of bees in a local area is likely related to the type, amount, or connectivity of land surrounding the surveyed area (Roulston and Goodell, 2011). The amount of semi-natural habitat within agricultural landscapes has been found to be positively associated and is often used as a significant predictor of healthy bee populations (Garibaldi et al., 2011; Goulson, 2003; Kim et al., 2006; Ricketts et al., 2008). However, habitat abundance may not necessarily be the best predictor; urban landscapes for example offer a limited amount of semi-natural habitat yet can be relatively rich in bee species (Roulston and Goodell, 2011).

In a study looking at the quality and isolation of habitat patches, Thomas et al. (2001) argue that habitat quality (available resources) of the patch is as important as patch size per se. Numerous studies have found that floral resource availability is the main driver in regulating bee populations (Potts et al., 2003; Roulston and Goodell, 2011; Torné, 2015), this study [Ch. 2.4.4]). Degradation, elimination, and fragmentation of large, abundant and plant-species-rich habitats may interrupt the interactions between wild bees and their resources (Gathmann and Tscharntke, 2002). Enhancing landscape connectivity can help to counteract some of the negative effects of habitat fragmentation by sustaining the genetic flow among sub-populations, facilitating species range shifts due to climate change and minimizing mortality risk (Saura and de la Fuente, 2016).

Wild bee species respond differently to spatial resource distribution (Torné, 2015). The flight radius of wild bees is determined by their body size (Greenleaf et al., 2007) and external factors such as landscape barriers (Zurbuchen et al., 2010). The maximum flight distance between nesting and food habitats for most wild bee species is between 300–1500 metres (Pfiffner and Müller, 2014). Findings from Gathmann and Tscharntke (2002) showed a maximum foraging range between 150–600 m for 16 solitary bee species. Furthermore Zurbuchen et al. (2010) state the predicted maximum foraging distances for small bee species between 100–200 m. Therefore local habitat structure appeared to be more important than large-scale landscape structure since nesting and foraging requirements must be available within a few hundred meters to maintain populations of these species (Zurbuchen et al., 2010).

On the other hand, larger bees such as bumble bees (*Bombus* spp.) have been found to fly distances over several thousand meters (Chapman et al., 2003) and are thus considered long-distance foragers (Steffan-Dewenter and Tscharntke, 1999). Therefore, larger bees may be less sensitive to fragmentation than smaller species that cannot fly long distances between habitat fragments (Roulston and Goodell, 2011). Steffan-Dewenter and Tscharntke (1999) found that with increasing isolation of flowering patches from semi-natural habitat, the mean body size of flower-visiting wild bees increased, emphasizing the positive correlation of body size and foraging distance...

Patches of semi-natural habitat distributed throughout productive agricultural landscapes may therefore increase bee diversity by improving access to resources by bee species that vary in body size. This increase in diversity can lead to improved pollination (Garibaldi et al., 2014; Kennedy et al., 2013; Ricketts et al., 2008). Including semi-natural

lands can also increase other ecosystem services such as natural pest regulation, soil and water quality protection by mitigating runoff, and soil erosion (IPBES, 2016; Wratten et al., 2012). Therefore, maintaining or enhancing landscape connectivity is a key part of current biodiversity conservation efforts (Saura and de la Fuente, 2016). It is however unclear how much semi-natural habitat, or connectivity between habitat patches is required to maintain wild bee populations or to ensure their associated ecosystem services.

Landscape connectivity analysis can help to identify the importance of semi-natural habitat or natural habitat for the maintenance or improvement of landscape connectivity. The importance of a habitat patch will be dependant on the quality of its habitat resources, its topological position or its connectedness within the landscape network (Saura and de la Fuente, 2016). Connectivity in this context is defined (and measured) as the amount of reachable habitat across a defined landscape, based on defined information inputs such as the amount of suitable habitat, the habitat attribute, the movement abilities and mortality risk of the focal species to disperse between the habitat patches (Saura and Torné, 2009).

The establishment of a spatially-explicit population model for an individual species is inherently prone to uncertainty in terms of accurately determining model inputs that represent the ecology of that species (Beier et al., 2017). The accuracy of these inputs is important as slight changes may result in significant differences in resulting connectivity indices based on population models (Rayfield and Fall, 2010). Due to this proneness to uncertainty in the quantification of functional connectivity, a more informative perspective can be gained by applying a hypothetical species approach (Huggard et al., 2007; Richards et al., 2002). In the case of hypothetical species, dispersal abilities are not based on one specific species, but are assigned to fall within the range of dispersal abilities of a group of species

that are present within that habitat type (Bevan, 2014; Huggard et al., 2007; Richards et al., 2002). Applying connectivity analysis to hypothetical species that are intended to represent and encapsulate groups of species with high and moderate sensitivity to fragmentation can be used to help the spatial targeting of conservation actions (Watts et al., 2010). The intention is to provide general indices of connectivity that are relevant to real species present in the area.

In this study, a landscape connectivity analysis using the Conefor 2.6 software package (Saura and Torné, 2012) was applied to a fragmented landscape in Delta, BC in order to demonstrate its use to target conservation or restoration actions for wild bees at the landscape-scale. A network of field margin habitat patches within the Agricultural Land Reserve (ALR) was evaluated for their importance and role in the overall landscape connectivity for two hypothetical species, representing groups of small and large bees with different dispersal abilities. Field survey data from 2015 and 2016 was incorporated into the model parameterization to define the habitat attributes for field margins. The attribute value, here the amount of open flowers, was chosen based on the season with the lowest habitat resources to best determine field margin habitat patches that are important in times of relative food scarcity. The specific objectives of this study were to: (1) Determine the season with the lowest floral abundance provided by all field margin habitat patches across the landscape; (2) Use Conefor to evaluate how the prioritisation of field margin habitat patches may vary between dispersal distances within an estimated dispersal range and between groups of small and large bees; and, (3) Identify the most important field margin habitat patches for small and large bees.

3.2 Material and methods

3.2.1 Study area

The municipality of Delta is located south of the city of Vancouver on the mainland of BC. The 9,403 ha of Delta's ALR is bordered by the Fraser River on the north, the City of Surrey on the east ,the U.S. border and Boundary Bay on the south, and the Strait of Georgia on the west (Ministry of Agriculture, 2012b). Cultivated land and field margins within the ALR landscape in Delta can be grouped into four main areas: Westham Island in the west, central-, northern- and the eastern- part of the ALR (Figure 3.1).
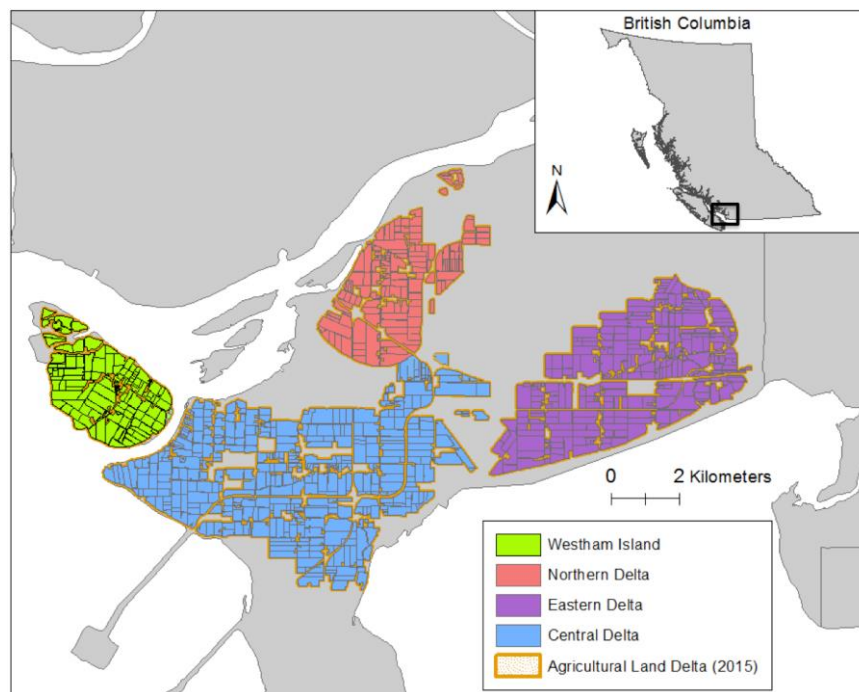
Figure 3.1: Spatial arrangement of the cultivated land within the delineation of the Agricultural Land Reserve in Delta, BC

The main field crop types produced on the 6,344 ha of cultivated land in Delta's ALR are vegetables (38%), forage/pasture (34%) and berries (19%) (Ministry of Agriculture, 2012b). Potatoes, beans, and mixed vegetables are the top field vegetable crops by area,

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whereas blueberries, cranberries and strawberries are the top berry crops (Ministry of Agriculture, 2012b). 1,155 ha (12%) consist of natural or semi-natural land cover, including vegetated land, wetlands and water bodies (Ministry of Agriculture, 2012b). The mean annual temperature registered for 1981-2010 was 10.6° C with 1262.4 mm of participation (Government of Canada, 2017).

3.2.2 Conefor Inputs Tool for ArcGIS

Conefor 2.6 (Saura and Torné, 2012) and the Conefor Inputs Tool for ArcGIS (Jenness, 2016) were used for the connectivity analysis. The software package calculates a range of graph-theoretic connectivity indices (Saura and de la Fuente, 2016). Graph theory defined by Minor and Urban (2017) is a network consisting of nodes potentially connected by links as a method to describe the distribution of habitat patches. Numeric values, based on the value of each node and the strength of each link, are assigned to nodes and links in order to quantify the emergent properties of the entire network (Minor and Urban, 2008).

The Probability of Connectivity (PC), Equivalent Connectivity for PC (EC(PC)) as well as derived metrics that quantify the importance of each individual patch (dPC) integrate both intrapatch (within patch) and interpatch (between-patch) connectivity in a single measure (Saura and de la Fuente, 2016). The connections between habitat patches are best characterized through a probabilistic model, in which there is a given probability of dispersal among habitat patches, typically modeled as a decreasing function of interpatch Euclidean distance (Urban and Keitt, 2001; Adriaensen et al., 2003 as cited in Saura and Pascual-Hortal, 2007). The probability indices were created by Saura and Torné (2012) in order to overcome some deficiencies they observed in other available connectivity indices (Saura and

Pascual-Hortal, 2007). The probability of connectivity indices have been shown to improve

performance compared to other existing indices and to be suited for landscape conservation

planning and change monitoring applications in particular (Saura and Pascual-Hortal, 2007).

3.2.3 Connectivity index calculation

3.2.3.1 Overall index values for the entire network

The **probability of connectivity (PC)** index calculation using the function as

described in Saura and Pascual-Hortal (2007):

$$
PC = \frac{\sum_{i=1}^{n} \sum_{j=1}^{n} a_i \cdot a_j \cdot p_{ij}^*}{A_L^2}
$$

$$
= \frac{PCnum}{A_L^2}
$$

where:

- n = is the total number of patches in the landscape

- a_i and a_j = the habitat attributes of patches i and j (e.g habitat quality)

- A_L = maximum landscape attribute (habitat and non-habitat patches), and

- p_{ij} *= the maximum product probability of all the paths between patches i and j

The connections between patches i and j are weighted by p_{ij} which is the probability

of direct dispersal (Saura and de la Fuente, 2016). The maximum product probability of a

path p_{ij} *

is the product of all the probabilities p_{ij} belonging to each step of moving through reachable habitat patches to get to a destination habitat patch (Bevan, 2014; Saura and de la Fuente, 2016). PC is defined as the probability that two individuals of a species randomly placed within habitat areas in a landscape are connected through a set of n habitat patches and connections p_{ij} among them (Saura and Pascual-Hortal, 2007). PC increases with improved connectivity and ranges from 0 to 1 (Saura and Pascual-Hortal, 2007).

The **equivalent connectivity for PC (EC(PC))** is a measure of the amount of reachable habitat resources in a landscape and is expressed in the same units as the attribute a_i (Saura and de la Fuente, 2016). EC(PC) is directly derived from PC and is calculated as the square root of the numerator of the PC index equation (Saura et al., 2011):

$$EC(PC) = \sqrt{PCnum}$$
$$= \sqrt{\sum_{i=1}^{n}\sum_{j=1}^{n} a_i \cdot a_j \cdot p_{ij}^*}$$

The maximum possible EC(PC) value for a given total amount of habitat resources occurs when all habitat area is confined in a single patch (no fragmentation) or when habitat patches are dispersed but with maximal interpatch connectivity (p_{ij} *= 1 between all patches) (Saura and de la Fuente, 2016). Thus EC(PC) values cannot exceed the sum of all a_i patches representing the total amount of habitat resources and will not fall below the a_i value of the patch with the most habitat resources (Saura and de la Fuente, 2016).

3.2.3.2　Habitat patch importance

Conefor calculates a change in PC value (dPC) for each patch that corresponds to the relative change in overall connectivity by removal of that patch expressed in percentage. These dPC values can be portioned into three different fractions to gain a more thorough understanding of the role of each individual habitat patch within a network (Saura and Rubio, 2010): *dPC = dPCintra + dPCflux +dPCconnector*

The intra fraction (dPCintra) is the contribution of the patch in terms of intrapatch connectivity. It corresponds to the amount of habitat resources provided by the specific patch that can be reached from within (Saura and Rubio, 2010). dPCintra is entirely independent of the patch's connections to other patches and returns the same value even if the patch is fully isolated (Saura and de la Fuente, 2016; Saura and Rubio, 2010).

The flux fraction (dPCflux) corresponds to the dispersal flux (weighted by the focal patch attribute) through the connections of the individual patch with all other patches in the network (Saura and Rubio, 2010). The focal patch is assumed to be the starting or the ending patch of that connection of flux. dPCflux incorporates both the attribute of the patch (e.g quality), where a higher attribute value creates more flux, and its position within the landscape network. This fraction only measures how well a patch is connected to the rest of the patches in the landscape and does not include the importance of a patch for maintaining connectivity among the other patches (Saura and de la Fuente, 2016; Saura and Rubio, 2010).

The connector fraction (dPCconnector) is the contribution of a patch as a connecting element or stepping stone between other habitat patches (Saura and Rubio, 2010). This fraction depends entirely on the topological position of a patch in the landscape network. dPCconnector is calculated independently of its attribute, but accounts for attributes of other

patches connected to the focal patch. Hence, dPCconnector for a focal patch will be higher

when it connects patches with higher attribute values (Saura and de la Fuente, 2016; Saura

and Rubio, 2010).

3.2.4   Data inputs

3.2.4.1   Map of suitable habitat and habitat attribute

The 2015 Delta crop cover map, which was developed by the DF&WT, Ducks

Unlimited Canada, and Canadian Wildlife Service, was used as the map base layer. Field

margins around agricultural fields on the Delta crop cover map were created by applying a

buffer around each field type using ArcGIS Version 10.2.2 (ESRI, 2014). The width of the

buffer was determined by the average width measured from ten randomly selected fields of

each crop type. Depending on the crop type the buffer width ranged between 3.9 to 4.7 m.

Field margins were then assigned to one of the three field margin habitat types; grass

margins, planted and remnant hedgerows to create a distinctive habitat class. Remnant

hedgerows were determined by visual interpretation and digitization of a 2012 orthophoto

(Figure 3.2 a).

Figure 3.2: Orthophoto (a) used for visual interpretation and digitization of field margins (b) assigned to three different field margin habitat types: purple = planted hedgerows, orange = remnant hedgerows and yellow= grass margins; analysis was applied across the landscape (c) with the buffered field margins that overlap merged into one distinct, contiguous habitat patch (d)

Planted hedgerows were added to the map from GPS field data (DF&WT, 2014), not including the two hedgerows that were planted in the fall of 2015. The field margins not assigned to either of the hedgerows types were assigned to grass margins (Figure 3.2 b). If field margins overlapped with neighbouring field margins of the same habitat type, they were merged into one, creating distinct, contiguous habitat patches and ignore potentially separating features like dirt roads or ditches (Figure 3.2 c, d).

Within the total area of 6,344 ha of cultivated land within Delta's ALR (Metro Vancouver, 2014) field margin habitat added up to 356.59 ha representing 30.83 % of the natural or semi-natural land cover and 5.6 % of the total land cover (Ministry of Agriculture, 2012b). The vast majority of the field margins were grass margins (total 321.2 ha, mean size 0.83 ha) followed by remnant hedgerows (total 33.3 ha, mean size 0.07 ha) and planted hedgerows (total 1.92 ha, mean size 0.08). All three field margin habitat types were considered as suitable wild bee habitat that offer floral resources and allow for a diversity of nesting habitats including bare soil or hollow plant stems (Goulson et al., 2015).

Floral abundance was the most clearly demonstrated factor influencing bee populations (see Ch. 2.4.4). Therefore, floral abundance measurements obtained from field survey data in 2015 and 2016 that was grouped into three seasons (Early = April and May; Mid = June and July; Late = August), was chosen as the best indicator for habitat quality. To calculate the floral abundance (average number of open flowers per m^2), considering the differences in dimensions between habitat types, the number of open flowers at the plot-level per habitat type (see Ch 2.2.6) was divided by the sampling plot area (200 m^2). The number of open flowers at the plot-level from both years was averaged and assigned to each field habitat type for the Late-season that had the lowest combined floral abundance from all three habitat types (see Ch. 3.3.1). No data was collected before the bloom of blueberries and potatoes, the most dominant crop types in Delta (Ministry of Agriculture, 2017). Notably, Early-season coincides with the blooming times of blueberry flowers and blooming of potato flowers occurs during Early- and Mid-season.

As the area of each habitat patch determines the amount of floral abundance, the final habitat attribute (*a* value) was calculated using the product of habitat area * habitat quality

for each habitat patch in the field margin shape file attribute table (Saura and de la Fuente, 2016).

3.2.4.2 Dispersal probabilities for focal species

There are a number of variables within the Conefor modelling approach, such as suitable habitat, habitat quality and dispersal distance, which all influence the model outputs. While suitable habitat and habitat quality are based on field data and relatively straightforward to parameterize, maximum dispersal distance is notoriously difficult to model or measure (Trakhtenbrotl et al., 2005). At a basic level, the spatial distribution of wild bees is expected to follow spatial distribution of flower and/or nesting sites (Torné, 2015). Bees are highly mobile species, however their ability to reach suitable habitat is determined by their body size (Gathmann and Tscharntke, 2002). To attain a realistic assessment of the actual flows and use of the landscape by different species, a functional connectivity approach that considers the ability of species to move through non-habitat areas is required (Tischendorf and Fahrig, 2000).

Conefor allows the integration of specific threshold distances for connectivity measurements. On top of dispersal distances, dispersal probabilities (p_{ij}) that represent the probability of direct dispersal between patches i and j can be defined. Connectivity applications on the landscape level generally assume symmetric dispersal probabilities where p_{ij} is equal to p_{ji} (Saura and Pascual-Hortal, 2007). Dispersal probabilities take into account that habitat patches with higher quality might be preferred by certain species even if they are more distant (Saura and Pascual-Hortal, 2007).

The majority of wild bees forage within a few hundred meters, whereas some are able to fly over thousands of meters (Greenleaf et al., 2007). For the purpose of this study, a range of dispersal abilities for groups of small and large bees present within the study area was derived from maximum forage distances of bee species of the same genus or with similar body sizes found in the literature. The maximum forage distance range was estimated between 100 m–300 m for small bees and between 800 m–1500 m for large bees based on results of numerous studies (Button and Elle, 2014; Chapman et al., 2003; Gathmann and Tscharntke, 2002; Steffan-Dewenter and Tscharntke, 1999; Zurbuchen et al., 2010). The dispersal probability of $p_{ij} = 0.05$ was chosen, based on the assumption that 5% of bees (small or large) were expected to successfully cover the maximum forage distance (Saura and Torné, 2012). Choosing several groups that represent species with high and moderate sensitivity to fragmentation can help to define targeted conservation guidelines benefiting one or more group of species (Watts et al., 2010).

3.2.4.3   Input files

The two necessary input text files for the connectivity analysis in Conefor were generated using the Conefor Inputs extension for ArcGIS (Jenness, 2016). The **node file** simply states the unique patch ID in the first columns and the second column contains its corresponding attribute value for that patch, here the area-weighted habitat quality. The **connection file** identifies the Euclidean (straight-line) distance (m) between the patches. The first two columns contain a pair of patch IDs, whereas the third column states the direct connection between the patch pairs. For each pair of patches, Conefor translates Euclidean distances into probabilities of direct dispersal (p_{ij}) by the internal decreasing exponential

function based on interpatch distance (Bevan, 2014; Saura and de la Fuente, 2016). Conefor determines the exponential decay rate in such a way that the function matches the user-specified probability and dispersal distance values (here $p_{ij} = 0.05$ with four different dispersal distances (d): 100 m, 300 m, 800 m and 1500 m). Thus, for this study, habitat patches separated by a distance smaller than d will get assigned $p_{ij} > 0.05$, while patches separated by distances larger than d will be assigned $p_{ij} < 0.05$ (Saura and de la Fuente, 2016).

3.3 Data analysis

3.3.1 Seasonal differences in floral abundance within field margins

To measure the seasonal differences in the amount of open flowers provided by field margins at the landscape-scale, the mean of the attribute values (product of habitat area * number of open flowers per m^2) for habitat patches of all types (n = 853) was compared between Early-, Mid- and Late-season. The data showed equal variances with non-normal distributions even when transformations were applied. Therefore the non-parametric Kruskal-Wallis one-way analysis of variance by ranks test was applied to determine the significance of differences among seasons in R (R Core Team, 2016). Post hoc tests according to Nemenyi for pairwise multiple comparisons of the ranked data were used to determine significant (p < 0.05) differences between seasons and floral abundance values (package PMCMR, Pohlert, 2016).

3.3.2   Comparison of connectivity indices between different dispersal distances

To quantify the change in the amount of reachable resources in a landscape, here the amount of open flowers provided by field margins, EC(PC) was compared within the estimated dispersal ranges and between groups of small and large bees. EC(PC) values were expressed in percentage of the maximum possible amount of habitat resources, hence the sum of all open flowers from all habitat patches, to obtain a comprehensive, comparable measurements (Saura and Torné, 2012). Using EC(PC) values compared to PC brings the advantage that they avoid very low metric values that might be obtained from PC, when the amount of habitat is very small compared to the extent of the analyzed landscape (Saura and Torné, 2012). In this study the amount of suitable habitat only represented 5.6 % of the analyzed landscape.

The change in PC value (dPC) for each habitat patch corresponds to the relative change in overall connectivity by removal of that habitat patch expressed in percentage. The sum of all dPC (%) values for each of the habitat patches in a landscape does not necessarily equal 100%. This sum will usually end up being greater than 100% (Saura and Pascual-Hortal, 2007). If a landscape network consists of several key connecting elements, the loss of each of these elements would disconnect the remaining habitat hence would result in a large decrease of reachable habitat (Saura and Pascual-Hortal, 2007). That loss, expressed in dPC values, would sum up to a percentage larger than 100 % when added together. The more critical connecting elements exist in a landscape, the more important is their presence for the stability of the whole network (Saura and Pascual-Hortal, 2007). With changing dispersal distances also the sums of dPC change. Therefore dPC values of habitat patches calculated with different dispersal distances cannot be compared directly. To evaluate how dPC values

of habitat patches may differ within the estimated dispersal ranges and between groups of small and large bees, ranks were assigned to habitat patches (n = 853) according to their dPC values (Saura and Rubio, 2010). Kendall's tau rank correlation coefficients (tau_b) were then calculated in R (R Core Team, 2016) for the ranks generated for the upper and lower ends of the estimated dispersal ranges as well as between groups for the lower bound of small and upper bound for large bees. Thus the differences in ranks due to changing flight distances within the focal group of species as well as in between groups of small and large bees could be evaluated (Saura and Rubio, 2010), indicating the relative uncertainty in modeled dPC associated with the range of flight distances.

3.3.3 Most critical areas for habitat connectivity and habitat availability

The assigned ranks based on the dPC value of habitat patches for the upper and lower dispersal distances were added together to create a new rank where the lowest sum was assigned the highest combined rank to best represent each group of species (Appendix 0) (Bevan, 2014; Huggard et al., 2007; Richards et al., 2002). Maps were created to visualize most critical areas for habitat connectivity and habitat availability based on the ten highest combined ranks for each group of small and large bees. Overlapping habitat patches from both groups were identified and visualized in a new map highlighting the areas of habitat critical for both bee groups.

3.4   Results

3.4.1   Seasonal differences in floral abundance within field margins

Significant differences in floral abundance on the landscape level were found between Early-, Mid- and Late-season ($p < 0.001$). The combined floral abundance of all three habitat types was the highest in Early-season, followed by Mid-season and Late-season having the lowest floral abundance. The same pattern was apparent when looking at planted and remnant hedgerows individually, but not for grass margins. Grass margins was the only habitat type that had a slightly, but not significantly, higher floral abundance in Late-season compared to Mid-season (Figure 3.3).
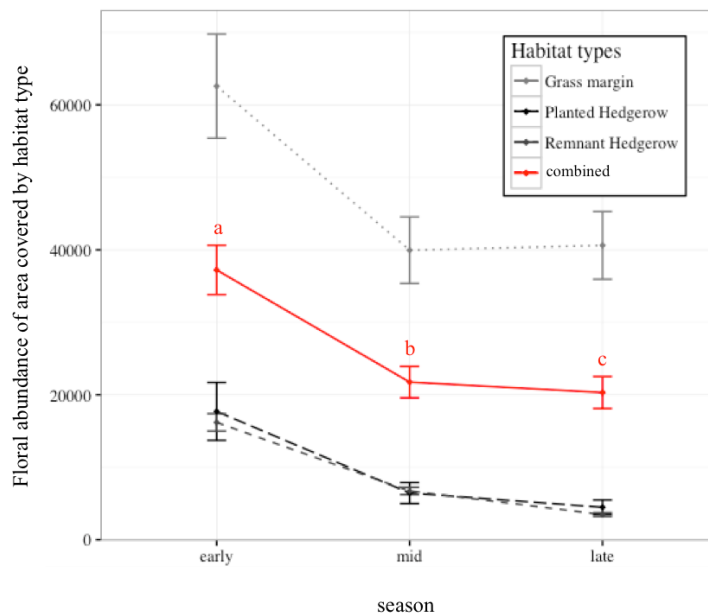
Figure 3.3: The mean abundance of open flowers by habitat type and for all three habitat types combined (red) at the landscape level. Significant differences among seasons for the combined habitat means ($p < 0.001$) are indicated with letters.

3.4.2 Comparison of connectivity indices between different dispersal distances

The analysis of connectivity indices showed that there were important differences between predictions for small bees, with dispersal distances between 100– 300 m, and large bees, with dispersal distances 800– 1500 m. The amount of reachable resources, here the amount of open flowers, measured by EC(PC) varied by 32.2 % between the upper and lower dispersal distance for small bees and by 16.2 % for large bees while the difference between the lower bound for small bees (100 m) and upper bound (1500 m) was 57.7 %. The percentage of reachable flowers out of the maximum amount of all open flowers from all habitat patches, varied by 14.3 % between the upper and lower dispersal distance for small bees and by 11.5 % for large bees while it differed by 41.3 % between the lower bound for small bees (100 m) and upper bound (1500 m).

The sum of dPC differed by 13.11 % between landscapes for the upper and lower dispersal distance of small bees and 7.1 % for the upper and lower dispersal distance of large bees (Table 3.1), while the difference between the lower bound for small bees (100 m) and upper bound (1500 m) was only 5.7 %. The contribution of the three fractions to the overall dPC value was mostly influenced by the dPCflux fraction, followed by the dPCconnector fraction but a rather low contribution of the dPCintra fraction for all dispersal distances (Table 3.1). The contribution of dPCintra decreased with increasing dispersal distances, whereas the contribution of dPCflux increased with increasing dispersal distances. The contribution of dPCconnector was higher for d = 300 compared to d = 100 m and d= 800 m compared to d = 1500 m (Table 3.1).

Table 3.1: Conefor connectivity indices calculated for four different dispersal distances.

Dispersal distances		PCnum	EC(PC)	EC(PC) in % of total amount	Sum dPC (%)	Sum dPC intra (%)	Sum dPC flux (%)	Sum dPC connector (%)
small	100 m	2.75e13	5244594	30.3	257.5	14.1	171.7	71.7
	300 m	5.98e13	7733230	44.6	270.6	6.5	187	77.1
large	800 m	1.10e14	1.04e7	60.1	270.3	3.6	192.9	73.9
	1500 m	1.54e14	1.24e7	71.6	263.2	2.5	195	65.7

Significant correlations ($p < 0.001$) were found for all calculations within and between both bee groups. Kendall's tau correlation coefficients between habitat patch ranks for upper and lower dispersal distances ranged from ($tau_b = 0.88$) for small bees (Figure 3.4 a) to ($tau_b = 0.95$) for large bees (Figure 3.4 b).
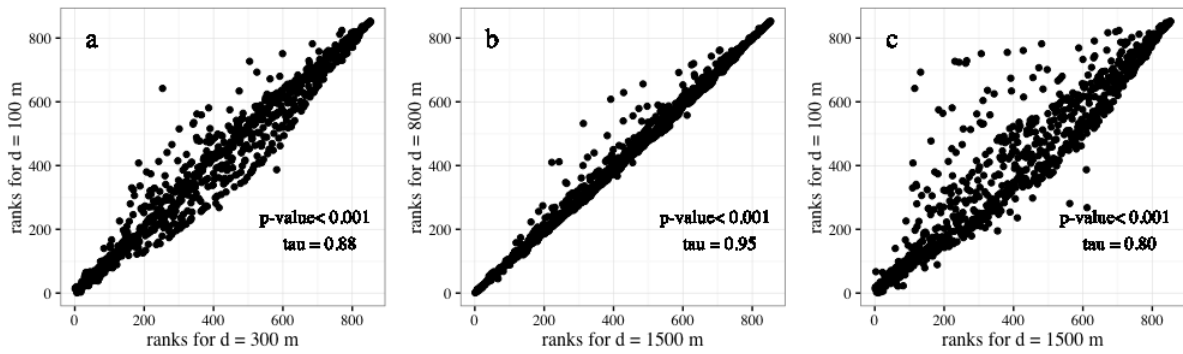
Figure 3.4: Kendall's tau correlation coefficient for ranks based of dPC values of all habitat patches ($n = 853$) calculated between the upper and lower dispersal distances of small (a) and large (b) bees as well as between ranks of the lower dispersal distance for small bees and the upper dispersal distance for large bees (c).

These correlations were both stronger than those between the ranks of the lower dispersal distance for small bees and the upper dispersal distance for large bees, which resulted in a correlation coefficient of ($tau_b = 0.80$) (Figure 3.4 c). These results confirm that higher correlations can be achieved by applying estimated dispersal distance ranges that represent small and large bees instead of using one dispersal distance for both bee groups.

3.4.3   Most critical areas for habitat connectivity and habitat availability

The top ten identified habitat patches based on the combined rank for small bees varied on average by 3.7 ranks from the upper and 4.5 ranks from the lower dispersal distance (Table 3.2).

Table 3.2: Calculation of the top ten combined ranks for small bees, drawn from assigned ranks based on the dPC value of habitat patches for the upper and lower dispersal distance. Ranks were added together to rank_overall and the new rank_combined was created where the lowest sum of rank_overall was assigned the highest rank_combined.

Node ID	dPC_300m	rank_300m	dPC_100m	rank_100m	rank_overall	rank_combined	Variance in ranks
803	6.585847	6	10.12005	1	7	1	0-5
805	6.472798	7	9.842786	2	9	2	0-5
542	6.420595	8	7.699407	4	12	3	1-5
632	7.151033	4	5.977891	10	14	4	0-6
506	7.36079	3	5.678066	12	15	5	2-7
742	15.97799	1	5.421988	14	15	6	5-9
646	5.891653	11	7.06775	5	16	7	1-4
570	6.20777	10	6.259788	7	17	8	2-5
834	8.030245	2	4.470124	17	19	9	7-10
515	5.780754	12	6.257333	8	20	10	2-6

The top ten identified habitat patches based on the combined rank for large bees varied on average by 0.6 ranks from the upper and 0.7 ranks from the lower dispersal distance (Table 3.3).

Table 3.3: Calculation of top ten combined ranks for large bees, drawn from assigned ranks based on the dPC value of habitat patches for the upper and lower dispersal distance. Ranks were added together to rank_overall and the new rank_combined was created where the lowest sum of rank_overall was assigned the highest rank_combined.

Node ID	dPC_1500m	rank_1500m	dPC_800m	rank_800m	rank_overall	rank_combined	Variance in ranks
742	16.84206	1	20.22835	1	2	1	0
649	13.66775	2	9.759775	2	4	2	0
608	11.34	3	8.037126	4	7	3	0-1
834	8.389831	4	9.350224	3	7	4	0-1
658	6.918185	6	6.387582	5	11	5	0-1
680	7.379048	5	5.783262	7	12	6	1
506	5.208233	9	6.070534	6	15	7	1-2
555	5.445688	8	5.72565	8	16	8	0
805	5.664101	7	5.377087	12	19	9	2-3
642	5.084962	10	5.530387	10	20	10	0

Four habitat patches were found within the top ten ranks or both small and large bees. Maps were created to visualize most critical habitat patches for habitat connectivity and habitat availability to best represent groups of small (Figure 3.5 a) and large bees (Figure 3.5 b) and to highlight overlapping, critical habitat patches for both bee groups (Figure 3.6).
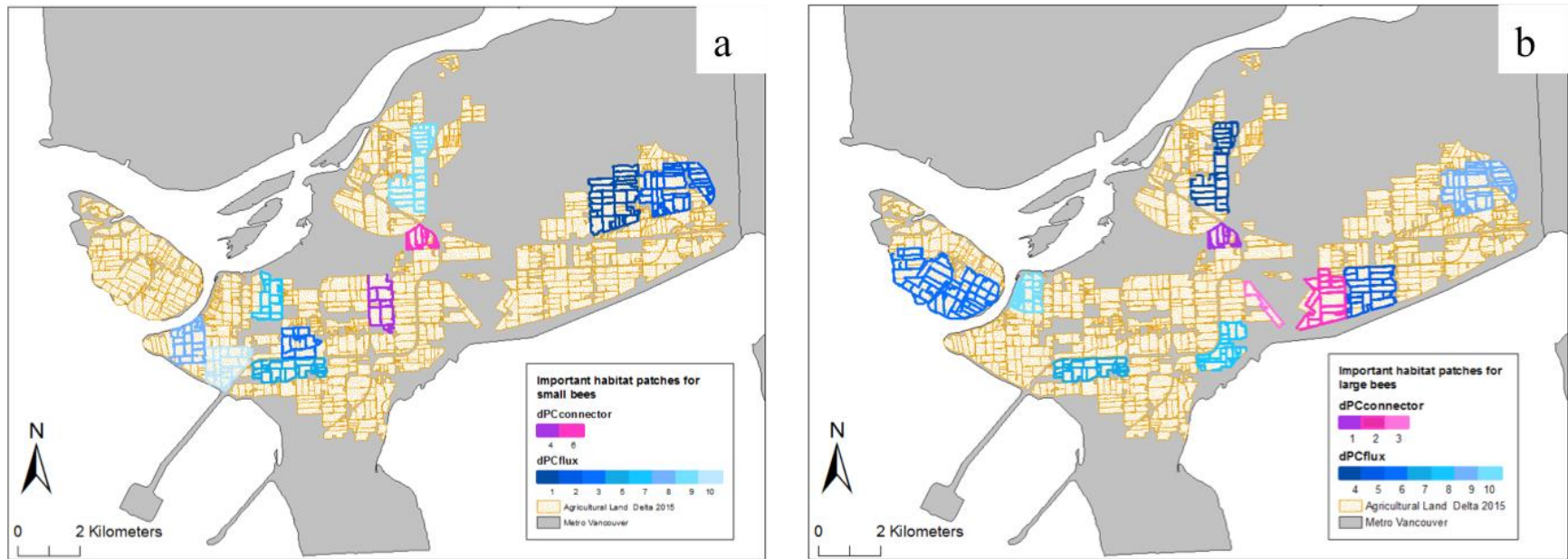
Figure 3.5: Ten most important field margin habitat patches in the Agricultural Land in Delta and their role to the maintenance of overall landscape connectivity and availability for groups of (a) small and (b) large bees drawn from combined ranks based on dPC values (Table 3.2 and Table 3.3). The roles of habitat patches are indicated based on the main contribution of the dPC fractions (either dPC Connector or dPCflux).
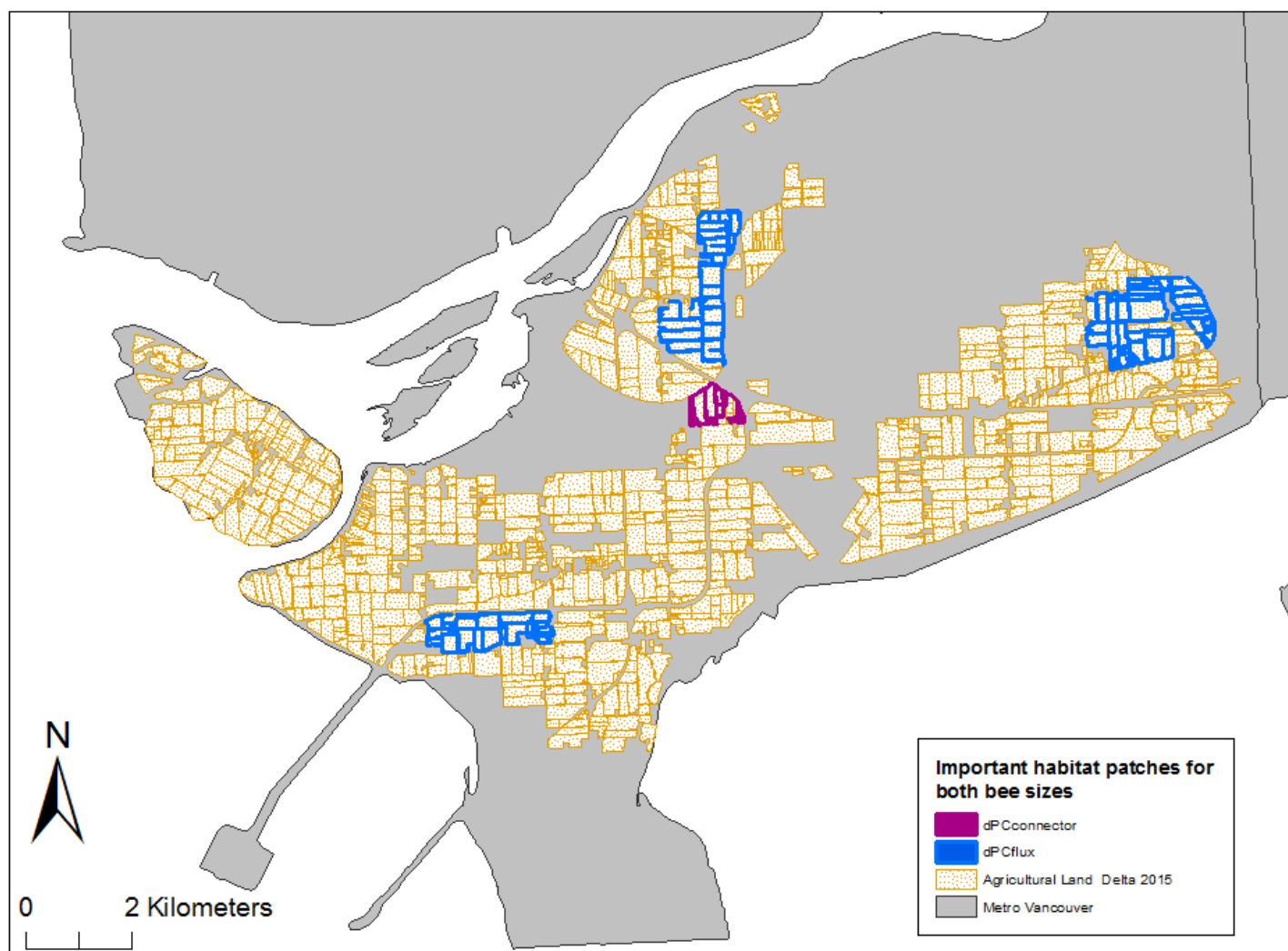
Figure 3.6: Overlapping field margin habitat patches of common importance for both groups of small and large bees drawn on combined ranks based on dPC values. The roles of habitat patches are indicated based on the main contribution of the dPC fractions.

3.5 Discussion

3.5.1 Seasonal differences in floral abundance within field margins

When considering the proportion of the different field margin habitat types and their contribution to the overall floral abundance at the landscape-level, significant differences were found between all three seasons. The highest floral abundance in all three field margin types combined was found in Early-season (April and May). Early-season overlaps with the blooming period of blueberries and potatoes, the predominant crop types in Delta (Ministry of Agriculture, 2012b, 2017; Pesticide Risk Reduction Program et al., 2007). The provision of floral resources by mass-flowering crops in the early season might reduce the importance of alternative food resources in field margins (Riedinger et al., 2015). The reversed pattern is apparent for the Late-season, when mass-flowering crops have dwindled and the importance of alternative food resources in field margins increases (Grass et al., 2016). However, Mackenzie and Winston (1984) found that both abundance and diversity of pollinators were lower on the crop plants (blueberry and raspberry) than on the surrounding native vegetation in the Lower Fraser Valley. Monocultures provide resources of little nutritional diversity and/or quality for bees (Gill et al., 2016). Alternative forage is thus invaluable to sustain them nutritionally during crop bloom (Fisher, 2016; Hannon and Sisk, 2009) and are critical before and after the crop bloom (Menz et al., 2011; Nicholls and Altieri, 2013). Several species of the genus *Bombus*, *Ceratina*, *Halictus*, *Lasioglossum*, *Megachile* and *Melissodes* have been observed after main crop bloom in Late-season. Female bees, with the exception of *Megachile spp.* and *Melissodes spp.*, overwinter as adults and need to be able to generate food reserves before hibernation (Wilson and Carril, 2016). The decline of food resources in field margins in Late-season specifically might have implications on bee species that overwinter as adults (Moisset and Buchmann, 2010; Willmer,

2011; Wratten et al., 2012). Ideally, habitat restoration measurements should be designed using plant species that do not bloom concurrently with major crop species but rather offer crucial resources in times of food scarcity and to reduce the concern of competition between crop flowers and simultaneously blooming wildflowers (Holzschuh et al., 2008; Morandin and Kremen, 2013; Vaughan et al., 2007). Since no data for floral resources was gathered before the main crop bloom, here the connectivity analysis was focused on Late-season to best determine habitat patches that are important for providing food resources to bees after main crop bloom when resources are most constrained and therefore potentially the most limiting for population persistence.

3.5.2 Comparison of connectivity indices between different dispersal distances

The Kendall's tau rank correlation coefficients (tau_b) between ranks based on dPC values of two dispersal distances at the upper and lower end of an estimated maximum foraging range were relatively high within both groups of small and large bees and stronger than the correlation between the ranks of the lower dispersal distance for small bees and the upper dispersal distance for large bees. Especially for large bees, the upper and lower dispersal distances differed by 700 m but the correlation was high (tau_b= 0.95) compared to small bees where the upper and lower dispersal distance only changed by 200 m, yet the correlation was considerably lower (tau_b= 0.88). These results suggest that the identification of important habitat patches based on ranks is fairly robust to uncertainties in the determination of dispersal distances along an estimated dispersal range for large bees, however differences between dispersal distances for small bees led to a disproportional higher change in ranks.

The contribution of the three fractions to the overall dPC value was mostly influenced by the dPCflux fraction, which indicates that most of the final importance of each habitat patch for the overall habitat availability is determined by the attribute of the habitat patch (e.g floral abundance*area), where a higher attribute value and its connectedness within the landscape network creates more flux (Saura and de la Fuente, 2016). The contribution of dPCflux increased with increasing dispersal distances due to improved inter-patch connectivity achieved with higher dispersal distances. The contribution of the dPCconnector fraction, which is solely based on the topological position of habitat patches within the landscape network and the potentially negative effects by removal thereof, also plays a role. Interestingly, the contribution of dPCconnector was the highest for d = 300 m, indicating that the habitat network contained more key-connecting elements that would disconnect the remaining landscape if removed, compared to other dispersal distances (Saura and Pascual-Hortal, 2007). The contribution of dPCconnector was also higher for d= 800 m compared to d = 1500 m. Therefore, bees with a dispersal distance between 300 and 800 m are likely more sensitive to fragmentation in the present network than bees with a dispersal distance of 100 m or 1500 m. These findings suggest a low inter-patch connectivity for bees with d = 100 m whereas for bees with d= 1500 m inter-patch connectivity seems to be high, thus the removal of existing habitat patches would have a lower impact on the remaining network. The contribution of intrinsic habitat characteristics, independent from the configuration and spatial structure of the habitat in the landscape, however, seems to be comparatively much lower in this landscape. The contribution of dPCintra decreased with increasing dispersal distances, indicating that for bees with shorter dispersal distances floral resources within the habitat patch do have a higher importance.

3.5.3   Most critical areas for habitat connectivity and habitat availability

All identified important habitat patches for large and small bees belonged to the grass margin habitat type. Grass margins were the most dominant field margin type in the landscape representing distinct, contiguous habitat patches. Therefore, a higher intra-patch connectivity was achieved, which positively influenced the overall importance of this field margin type.

The analyses of the top ten ranked combined habitat patches for large bees showed that the variance in ranks for both ends of the estimated dispersal distance range was marginal. Within the group of large bees, modelling landscape connectivity for a dispersal distance that falls within an estimated range can be used pragmatically to guide and support management actions (Watts et al., 2010). Large bees, such as bumble bees have a special importance as pollinators for common insect-pollinated crops in Delta. In contrast to honey bees, bumble bees have the ability to buzz pollinate flowers which makes them more effective pollinators of crops that require that kind of pollination, such as blueberries and cranberries (Michener, 2000; Moisset and Buchmann, 2010). A study by Button and Elle (2014) in the Lower mainland of BC showed that increased abundance of wild bumble bees led to higher blueberry yields. Therefore, if the focus of conservation efforts is on species that are potentially valuable pollinators of adjacent cropland, habitat patches shown most critical for large bees might be prioritized for protection or restoration efforts.

The top ten identified combined habitat patches for small bees however revealed variations in ranks that potentially mislead conservation decisions. Slight changes in dispersal abilities for small bees resulted in disproportional higher differences in connectivity indices compared to large bees (Rayfield and Fall, 2010). These findings conform with studies that have stated the impact of local habitat structure for small bees and the necessity of dense networks of

habitat patches for the maintenance of these species (Gathmann and Tscharntke, 2002;

Zurbuchen et al., 2010). In the case of small bees, applying a dispersal distance that lies within a

range of dispersal abilities, may give a coarse informative perspective on conservation planning.

However, the accuracy of dispersal distance inputs becomes more imminent when prioritizing

areas for conservation efforts targeted for small bees.

The habitat patches identified, using Conefor, as most important habitat were, for the

most part distinct for small bees and large. The ten most important field margin habitat patches

for small bees were mostly driven by the dPCflux fraction, except for two habitat patches that

had a higher dPCconnector fraction. The locations of these two connecting elements suggest an

importance of maintaining the connection between the central- and the northern- part of the

ALR. The absence of connecting elements and important habitat patches for dispersal flux on

Westham Island, suggest a weak connection with the rest of the network. The absence of a

connecting element between the central- and the eastern-part, indicates a low connection

between these parts as well, however, the presence of an important habitat patch for flux

dispersal indicates that the intra-patch connection in the eastern- part is essential to the overall

availability of flowers. Westham Island is separated from the rest of the habitat network through

the Fraser river with a width >300 m and the eastern part is separated by Burns bog, an

undeveloped wetland area and the Boundary Bay Airport which creates a gap within field

margins >300 m. Burn's bog is located outside of the ALR delineation and was thus not included

in the landscape connectivity analysis. However this area likely acts as a connecting element

between the central-, northern- and eastern-part of the ALR and might thus reduce the

importance of identified connecting elements within the ALR borders. Field data revealed that

there were no differences in wild bee communities between sampling sites located in different parts of the ALR (Ch 2.4.1).

Seven out of the top ten ranked habitat patches for large bees were driven by their dPCflux fraction, whereas the top three ranked habitat patches were driven by the dPCconnector fraction. The locations of these three connecting elements suggest an importance of maintaining the connection between the central- northern- and eastern-part of the ALR. The absence of connecting elements in the western boarder of the central part and the presence of an important dPCflux habitat patch on Westham Island suggest a well-connected habitat network for large bees.

Identified habitat patches of importance differed highly between groups of small and large bees with only four habitat patches were found to overlap. Three out of the four overlapping habitat patches were driven by their dPCflux fraction, whereas one habitat patch was driven by the dPCconnector fraction. The location of this connecting element highlights the importance to maintain the connection between the floral resources in the central- and the northern-part of the ALR for both small and large bees. Additionally, three habitat patches that are important sources of dispersal flux were located in central- northern- and the eastern-part that provide an abundance of habitat resources while being well connected within the network of habitat patches. These identified habitat patches that are most critical for habitat connectivity and habitat availability need to be prioritized for conservation efforts to prevent further fragmentation crucial food resources. While no direct bee-flower visitations in the four identified important habitat patches were obtained, field data for Late-season showed that three different bumble bee species *Bombus impatiens*, *Bombus vosnesenskii* and *Bombus californicus*, one leafcutter bee *Megachile perihirta* and one leaf-cutter cuckoo bee *Coelioxys rufitarsis* were visiting flowers

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such as *Taraxacum officinale*, *Vicia cracca*, *Trifolium repens*, *Cirsium arvense* and *Raphanus raphanistrum* found in grass margins. These data suggest that these bees would likely benefit from targeted conservation efforts.

3.6 Conclusions

Differences between floral abundance, which was used as the habitat attribute for the Conefor landscape connectivity analysis as an indicator for quality, were found between seasons. Late-season was shown to have significantly less floral abundance compared to Early- and Mid-season. The decline in floral abundance in field margins coincides with the dwindling presence of mass-flowering crops, highlighting the importance of protecting alternative food resources in field margins for bees that are active in Late-season and particularly for bees that overwinter as adults (e.g bumblebee queens) and thus need to be able to generate food reserves (Moisset and Buchmann, 2010; Willmer, 2011; Wratten et al., 2012).

The amount of reachable habitat resources in a landscape as measured by EC(PC) differed more between the upper and lower dispersal distance for small bees compared to large bees. A smaller increase in dispersal distance resulted in a higher amount of reachable resources for small bees, suggesting that a close neighbourhood of foraging habitat within few hundred meters is crucial to maintain habitat connectivity of small bees.

The determination of habitat importance drawn from ranks based on dPC values were highly correlated between the upper and lower dispersal distances for both small and large bees. These correlations were both stronger than those between the ranks of the lower dispersal distance for small bees and the upper dispersal distance for large bees. Ranks for the upper and lower dispersal distances were combined to visualize most critical areas for habitat connectivity

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and habitat availability that best represent groups of small and large bees. However, differences between dispersal distances for small bees lead to a disproportional higher variance in ranks compared to large bees, thus needs to be handled with caution when prioritizing areas for conservation efforts targeted for small bees.

The spatial distribution of the ten highest ranked combined habitat patches critical for habitat connectivity and availability for both small and large bees show that there are some overlapping habitat patches that are critical for both bee groups. The most critical areas for habitat connectivity and availability were mostly contributing in terms of the dispersal flux through the connections of the individual habitat patch with all the other habitat patches in the network and in terms of the topological position of the habitat patch in the network for both groups of bees. When assessing the level of connectivity between habitat patches, the landscape should be compared to the scale of short- and long-distance dispersal, to target conservation management decisions. However, areas of importance were found to overlap for both groups of bees, so focusing conservation efforts in those areas is most likely to benefit both groups of species.

Chapter 4: Conclusion

4.1   Research conclusions

The success of conservation activities likely depends on both the implementation of strategies at the farm- and landscape-scale. While small-scale habitat restorations along farm fields offer the potential to enhance wild bee communities and their associated pollination services, they can only be sustained when considering the habitat connectivity at the landscape-scale.

The results of this research suggest that planted hedgerows consisting of a diverse mix of native trees and shrubs do not necessarily lead to more abundant, species-rich and diverse wild bee populations compared to remnant hedgerows and grass margins. In fact, overall wild bees collected from flowers and pan traps were significantly more abundant, species rich and diverse in grass margins than in both hedgerow types. Wild bees showed a preference for non-native, herbaceous flower species compositions in grass margins that significantly differed from the two hedgerow types. While floral abundance was shown to be the best predictor variable for netted bee abundance, it did not explain the differences between habitat types alone. However, when selecting the floral abundance of species visited by wild bees exclusively, field margin types with the highest net-collected wild bee abundance aligned with highest floral abundance in both years.

Field margins can offer additional floral resources to complement mass-flowering crops that are often blooming for a short period of time, while flight periods of many bees extend beyond the availability of these flowers. The dominant crop types in Delta bloom in Early- (blueberries) and Mid-season (potatoes) which offer temporary food resources for a select number of bees (Vaughan et al., 2007).

While these mass-flowering crops are dwindling in Late-season, this also coincided with the lowest floral abundance found in field margins compared to Early- and Mid-season. The findings of this research suggest considering available floral resources at the landscape-scale in order to detect temporal differences and highlight the importance of protecting alternative food resources in field margins in times of scarcity. Landscape connectivity analysis using the software package Conefor furthermore enables the spatially explicit identification of field margins that provide crucial food resources in times of food scarcity or act as connecting elements. By assessing the habitat connectivity for bees with short- and long-distance dispersal abilities, conservation measurements can be targeted to benefit one or several groups of species and thus be used to guide conservation decisions with broad impact.

Here I demonstrated how field data collected at the farm-scale can be combined with landscape connectivity analysis incorporating qualitative, temporal and spatial aspects to optimize conservation management. This study contributes to the body of research that emphasizes the importance of plant species selection that attract local bee communities and offer floral resources when mass-flowering crops are not available. The application of the Conefor landscape analysis to a real-world case study contributes to the understanding of how connectivity indices can inform conservation planning in agricultural landscapes.

4.2   Recommendations for wild bee conservation at the farm-scale

Given the farm-scale results of Ch. 2, hedgerow restoration within the DF&WT Hedgerow Stewardship Program could be optimized to better support the thriving of wild bees by adjusting the plant species composition to attain higher floral species abundance over time. This would provide flowers throughout the diversity of bee activity times. Temporal variability

in emergence and foraging habits have been observed in Delta's field margins among several genera of bees: *Bombus*, *Ceratina*, *Halictus*, *Lasioglossum* and *Megachile* were observed Early- to Late-season, *Andrena*, *Hoplitis* and *Osmia* were observed Early- to Mid-season, and *Melissodes* were observed Mid- to Late-season, demonstrating the importance of ensuring blooms through a long time period. The inclusion of grassy strips as a buffer along planted hedgerows in the DF&WT Hedgerow Stewardship Program could be a valuable addition for wild bee conservation in Delta.

4.2.1   Enhance floral species richness

Wild bees visited only five out of eight native shrub and tree species that were recommended for the Hedgerow Stewardship Program. Only six out of 17 plant species that were visited were shrubs including the non-native species *Rubus armeniacus*. On shrubs, the most bee visitations per flower unit were counted on *Rosa nutkana* (42.8), followed by *Rubus armeniacus* (31.2) and *Cornus sericea* (14.5) (Table 2.8). Native shrubs that have been visited by wild bees should be promoted, while additional, beneficial native species could complement the current plant species recommendations for hedgerow plantings (Appendix C.1). The majority of bee visits were recorded on non-native herbaceous plant species found in grass margin habitats or the understory of hedgerows (Table 2.8). While it is problematic to promote these non-native plant species, they have been shown to attract a variety of wild bees, the majority consisting of bumble bees (Figure 2.3). If grassy strips can be included as a buffer around planted hedgerows, simply leaving these areas alone and protecting them from pesticides could create additional foraging opportunities for wild bees (Vaughan et al., 2007). In addition, these areas might be enhanced by removal of invasive plants and/or the addition of key native flowering plants (Vaughan et al.,

2007) (Appendix C.2). Native flowering plants are well-adopted to growing conditions, offer

good sources of nectar and pollen and are usually not weedy (Vaughan et al., 2007). If mowing is

used, where possible grassy strips should not be mowed during critical wildlife nesting season

and not until past peak bloom (Vaughan et al., 2007).

4.2.2   Use synergies with the DF&WT Grassland Set-aside Stewardship Program

The DF&WT currently runs five other Stewardship Programs besides the Hedgerow

Stewardship Program, such as the Grassland Set-aside Stewardship Program which encourages

farmers to plant fields with grasses and clover to restore the soil (DF&WT, 2015c). My results

suggest a modest increase in the DF&WT grassland set-aside mix (DF&WT, 2015c) with

legumes and forbs could provide nectar and pollen for supporting wild bees (Orford et al., 2016).

Moreover, results from Orford et al. (2016) suggest that enhanced pollinator functional diversity,

richness and abundance lead to increased pollination services and consequently higher yields in

surrounding crop fields (Orford et al., 2016). Fallowed fields sown with a ground cover also

provide various other benefits, such as improving erosion control and soil permeability, nitrogen

fixing, buffer against invasive weeds, and shelter other beneficial insects (Vaughan et al., 2007).

When selecting plant species for wild bees, nectar-rich broadleaf cover crops should be

prioritized (Vaughan et al., 2007). Some plant families such as Boraginaceae, Brassicaceae,

Apiaceae and Fabaceae have been used in beneficial insect conservation practices in the last

decade (Barbir, 2016; Gill et al., 2016). For example, *Trifolium spp*., *Vicia spp*., and *Phacelia*

spp. are common cover crops that are highly attractive to a variety of wild bee species (Vaughan

et al., 2007). A survey in 2010 by Parkinson and Heron (2010) found a relatively high bumble

bee abundance and diversity in DF&WT grassland set-asides, with the two most uncommon

species exclusively observed foraging on *Trifolium pratense* at grassland set-asides in Delta. While grasses are not typically attractive to bees, grass cover crops such as rye and oats provide potential nesting sites for bumble bee colonies, benefit soil health and are easily integrated into diverse, multi-species seed mixes (Vaughan et al., 2007)(Appendix C.3).

4.3   Recommendations for wild bee conservation at the landscape-scale

The results of my landscape analysis in Ch. 3, show how conservation management decisions can be informed by considering differences for species with short- and long-distance dispersal abilities. Identified habitat patches of importance differed highly between groups of small and large bees (Figure 3.6). However, some habitat patches were found to overlap for both groups of bees. My recommendation would be to focus conservation interventions to protect or enhance habitat in those areas in order to benefit both groups of species. These identified habitat patches that are most critical for resource availability and connectivity need to be protected from further fragmentation. Habitat patches identified as connecting elements could be prioritized for habitat restoration to enhance dispersal of wild bees between otherwise isolated fragments to promote diverse pollinator communities at the landscape-scale.

4.4   Limitations and future research

There are several limitations to the evaluation of hedgerow establishment on farm field margins in Ch. 2. First, it is important to state that there are many other factors influencing bee populations that have not been considered in this study, such as nesting resources, pesticides, diseases and climate change (IPBES, 2016). Secondly, the sampling method applied for this study was modified from previously established sampling protocols (Droege, 2012; Sheffield,

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2009), however during my field work, other guidelines were published urging to standardize the methodology for better comparison between studies on a global scale (LeBuhn et al., 2016). Future studies of this nature should consider adopting this standardized methodology. While field surveys were conducted from April to August, vegetation and bee species present in field margins in early spring were not included. Thus floral abundance before blooming times of most dominant crops was not assessed and some wild bees, such as species of the genus *Andrena* and *Osmia* might be underrepresented. Lastly, the skills between volunteers conducting field sampling in 2015 varied highly compared to 2016. In 2015, the volunteers who participated in the study altered greatly between sampling days, whereas in 2016, a steady team of volunteers was conducting the field work over the entire season. Working with a consistent, well-trained team of volunteers would have likely substantially reduced the variability in the dataset.

There are several limitations to the landscape connectivity analysis applied in Ch. 3. First, only field margins were considered as suitable habitat for wild bees even though there are other land covers, such as woods, grassland and fallow fields that are habitats most likely used by wild bees as well. Those land covers were not included in the spatial analysis given the field data was only collected for field margin habitats. Furthermore, the analysis was limited to the landscape within the delineation of the ALR. The landscape context outside the border is not included, but might influence the identification of important habitat patches. For example Burns Bog, an undeveloped wetland area between the central-, northern- and eastern-part of the ALR likely connect the parts of the ALR outside the borders thus might reduce the importance of identified connecting elements. Lastly, there is a bias in the identification of important habitat patches towards grass margins. Grass margins are the most dominant field margin type in the landscape thus when buffers around fields were applied to create field margins, overlaps occurred mostly

between neighbouring grass margins, which were merged into one distinct, contiguous habitat patch. This leads to a higher intra-patch connectivity, which positively influences the overall importance of the patch. However, the maximal width between merged grass margins was 9.4 m, thus even if the patches are not solidly connected, the risk associated in crossing such a distance is marginal. For future studies, it would be interesting to incorporate other land covers into the landscape analysis and include natural areas adjacent to the ALR border to get a more realistic measurement of connectivity. Direct monitoring of wild bee species to test these connectivity indices as indicators of species maintenance and their associated pollination services across the landscape should be continued in the future.

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Appendices

Appendix A : Wild bee and plant species lists

A.1 Total wild bee species per habitat type

Table A.1: Distribution of wild bee species among grass margins, planted and remnant hedgerows from 2015 and 2016. Species are arranged in alphabetical order. Bees not identified to species are specified as sp.1 unidentified male species are specified as sp.a and are excluded from the total number of bee species. Introduced species in red.

Genus	Species/morphospecies	Grass margin	Planted Hedgerow	Remnant Hedgerow	Total number
Agapostemon	*texanus*	10	1	2	13
Andrena	*caerulea*		1		1
Andrena	*prunorum*		1		1
Andrena	*rufosignata*			3	3
Andrena	*saccata*		1	2	3
Andrena	*sp.1*			1	1
Andrena	*sp.a*			2	2
Anthidium	*manicatum*	3	1		4
Bombus	*californicus*	24	7		31
Bombus	*flavifrons*	36	4	9	49
Bombus	*impatiens*	42	22	13	77
Bombus	*melanopygus*	1		2	3
Bombus	*mixtus*	27	16	30	73
Bombus	*NA*	3		1	4
Bombus	*rufocinctus*	12	3	5	20
Bombus	*sp.a*	1	1	3	5
Bombus	*vagans*	1			1
Bombus	*vosnesenskii*	11	4	4	19
Ceratina	*acantha*	20	58	61	139
Coelioxys	*rufitarsis*	2			2
Colletes	*kincaidii*	1		2	3
Halictus	*confusus*	1	1	1	3
Halictus	*rubicundus*	7	1	1	9
Halictus	*tripartitus*	13	24	9	46
Hoplitis	*grinnelli*		1		1
Hylaeus	*modestus*	9	6	3	18
Lasioglossum	*Dialictus*	17	11	15	43
Lasioglossum	*Evylaeus*	1	1	2	4
Lasioglossum	*zonulum*	38	20	12	70
Megachile	*melanopahea*	1			1
Megachile	*montivaga*	4			4
Megachile	*perihirta*	15	5	5	25
Melissodes	*microstrictus*	4		1	5
Melissodes	*rivalis*	1			1
Melissodes	*sp.1*	1	1		2
Melissodes	*sp.a*			2	2
Nomada	*sp.1*				1
Osmia	*bucephala*	7			7
Osmia	*caerulescens*	1			1
Osmia	*dolerosa*	7	4	1	12
Osmia	*posilla*	2		2	4
Osmia	*simillima*	1		1	2
Osmia	*sp.1*	3			3
Osmia	*sp.a*	3		1	4
Sphecodes	*sp.1*	1	2		3
Sphecodes	*sp.a*		1		1
Total number of bee species		**33**	**24**	**24**	**40**
Total number of bees		**331**	**198**	**196**	**723**

A.2   Total plant species per habitat type

Table A.2: Distribution of floral plant species among grass margins, planted and remnant hedgerows from 2015 and 2016. Species are arranged in alphabetical order. Values indicated with 0, were present in sampling plots but did not occur within the sampling quadrats. Plants that were not identified to species are specified as sp. Native species recommended by the DF&WT are marked with *.

Genus	Species	Grass margin	Planted Hedgerow	Remnant Hedgerow	Total floral abundance per m^2
Brassica	*campestris*	0	0		0
Capsella	*bursa-pastoris*	0	2.9	0.2	3.1
Cardamine	*sp.*		0	0.8	0.8
Cerastium	*arvense*	1			1
Cirsium	*arvense*	1.2	8	1.5	10.7
Cirsium	*vulgare*	0	0.2		0.2
Convolvulus	*arvensis*			0	0
Cornus	*sericea/ stolonifera**		15.4	0	15.4
Crataegus	*douglasii**			112.5	112.5
Crataegus	*monogyna*		82.1	1.1	83.2
Crepis	*capillaris*	14	0	0	14
Cucurbita .	*sp.*		0		0
Epilobium	*ciliatum*	0	49.4	4.1	53.5
Galinsoga	*ciliata*	0			0
Geranium	*dissectum*	0	0.6	2	2.6
Geranium	*sp.*	1.2		0	1.2
Hypericum	*perforatum*	0		0	0
Hypochaeris	*radicata*	0	5.6		5.6
Lactuca	*serriola*	0	0		0
Lamium	*purpureum*		0.7		0.7
Leucanthemum	*vulgare*	0			0
Lotus	*corniculatus*	0.6			0.6
Malus	*fusca**		24.1	1.5	25.6
Malva	*sp.(purple)*	0			0
Malva.	*sp (white)*	2.6	2	0	4.6
Matricaria	*discoidea*	18.6		1.6	20.2
Medicago	*lupulina*	0	4.8		4.8
Melilotus	*alba*	0			0
Microseris	*borealis*		0.7		0.7
Myosotis	*laxa*	0		0	0
Pastinaca	*sativa*	0.4			0.4
Persicaria	*maculosa*		0		0
Philadelphus	*lewisii**		0		0
Plantago	*lanceolata*	0	11.1	2.6	13.7
Plantago	*major*	0			0
Polygonum	*aviculare*	0			0
Polygonum	*persicaria*	0			0
Ranunculus	*acris*	7	17.8	5.5	30.3
Ranunculus	*repens*	12.6	29.4	3	45
Raphanus	*raphanistrum*	0	9.4	59	68.4
Rosa	*eglanteria*			34.4	34.4
Rosa	*nutkana**		10	3.9	13.9
Rubus	*armeniacus*	2.8	23.4	100.5	126.7
Rubus	*laciniatus*	0	0.4	1.1	1.5
Rubus	*parviflorus**		0.9		0.9
Rubus	*spectabilis**		0.1	4.8	4.9
Rubus	*ursinus*		0.7	1.3	2
Senecio	*viscosus*	0	0		0
Senecio	*vulgaris*	0	0	0	0
Sinapis	*arvensis*		0.6		0.6

| Genus	Species	Grass margin	Planted Hedgerow	Remnant Hedgerow	Total floral abundance per m^2
Sisymbrium	*officinale*			0	0
Solanum	*americanum*		0.5		0.5
Solanum	*dulcamara*	0	0		0
Sonchus	*arvensis*	0			0
Sonchus	*asper*	0	1.2	0	1.2
Sonchus	*oleraceus*	0			0
Sorbus	*sitchensis**			0	0
Spirea	*douglasii**		6.1	6	12.1
Stellaria	*media*	0	0.2		0.2
Symphoricarpos	*albus**		27	4.8	31.8
Taraxacum	*officinale*	0.4	0.1	0	0.5
Trifolium	*pratense*	8		0	8
Trifolium	*repens*	49.8	0	0	49.8
Tsuga	*heterophylla**		0		0
Viburnum	*edule**		0	0	0
Vicia	*americana*	0	0		0
Vicia	*cracca*	17.2	11.5	5.1	33.8
Vicia	*sativa*	0			0
Total number of Plant species		**47**	**55**	**35**	**68**
Total number of open flowers		**137.4**	**346.9**	**357.3**	**841.6**

A.3 Bee-flower visitation list

Table A.3: Number of wild bee species by plant species or pan-trap collected in 2015 and 2016.

Plant species/ Wild bee species	*Cornus sericea*	*Rosa nutkana*	*Rubus spectabilis*	*Spirea douglasii*	*Symphoricarpos albus*	*Rubus armeniacus*	*Vicia cracca*	*Ranunculus repens*	*Ranunculus acris*	*Taraxacum officinale*	*Trifolium repens*	*Cirsium arvense*	*Trifolium pratense*	*Medicago lupulina*	*Raphanus raphanistrum*	*Geranium dissectum*	*Cirsium vulgare*	Blue Pan trap	White Pan trap	Yellow Pan trap	Total
Agapostemon texanus								1										6	4	2	13
Andrena caerula																				1	1
Andrena prunorum																				1	1
Andrena rufosignata																		1	1	1	3
Andrena saccata																		1		2	3
Andrena sp.1																			1		1
Andrena sp.a					1	1															2
Anthidium manicatum														1				1	1	1	4
Bombus californicus							1		1			1						16	10	2	31
Bombus flavifrons						2	28				2	1						8	7		48
Bombus impatiens	3	3		3	8	11	10				6	8			1		1	8	6	8	77
Bombus melanopygus						1												2			3
Bombus mixtus		1	1		5	8	4					1				2		27	14	10	73
Bombus NA																		2	1	1	4
Bombus rufocinctus						1						1						8	3	7	20
Bombus sp.a					1	1												4			7
Bombus vagans																		1			1
Bombus vosnesenskii					1	2	4				1							7	2	2	19
Ceratina acantha						1	1		1			1						46	51	39	139
Coelioxys rufitarsis						1														1	2
Colletes kincaidii						1													1	1	3
Halictus confusus																		1		2	3
Halictus rubicundus														1				1	2	5	9

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Plant species/ Wild bee species	Cornus sericea	Rosa nutkana	Rubus spectabilis	Spirea douglasii	Symphoricarpos albus	Rubus armeniacus	Vicia cracca	Ranunculus repens	Ranunculus acris	Taraxacum officinale	Trifolium repens	Cirsium arvense	Trifolium pratense	Medicago lupulina	Raphanus raphanistrum	Geranium dissectum	Cirsium vulgare	Blue Pan trap	White Pan trap	Yellow Pan trap	Total
Halictus tripartitus																		17	11	18	46
Hoplitis grinnelli																		1			1
Hylaeus modestus																		4	1	13	18
Lasioglossum zonulum		2		1		2			2									22	16	25	70
Lasioglossum Dialictus															1			26	10	6	43
Lasioglossum Evylaeus																		1		3	4
Megachile melanophea																		1			1
Megachile montivaga																		2		2	4
Megachile perihirta				1			3			1								12	2	5	25
Melissodes microstrictus																		1	1	3	5
Melissodes rivalis																		1			1
Melissodes sp.1																		2			2
Melissodes sp.a																		2			2
Nomada sp.																					1
Osmia bucephala																		3	4		7
Osmia caerulescens																		1			1
Osmia dolerosa						1						1						3	3	4	12
Osmia posilla												1							2	1	4
Osmia simillima																		1		1	2
Osmia sp.1																		2		1	3
Osmia sp.a												1							2	1	4
Sphecodes sp.1																		1	2		3
Sphecodes sp.a																				1	1
Total number of bee species	**1**	**3**	**1**	**3**	**5**	**12**	**7**	**1**	**3**	**1**	**6**	**6**	**1**	**1**	**2**	**1**	**1**	**35**	**27**	**32**	
Total number of bees	**3**	**6**	**1**	**5**	**16**	**32**	**51**	**1**	**4**	**1**	**12**	**13**	**1**	**1**	**2**	**2**	**1**	**242**	**159**	**169**	**723**

Appendix B : Maps dPC ranking
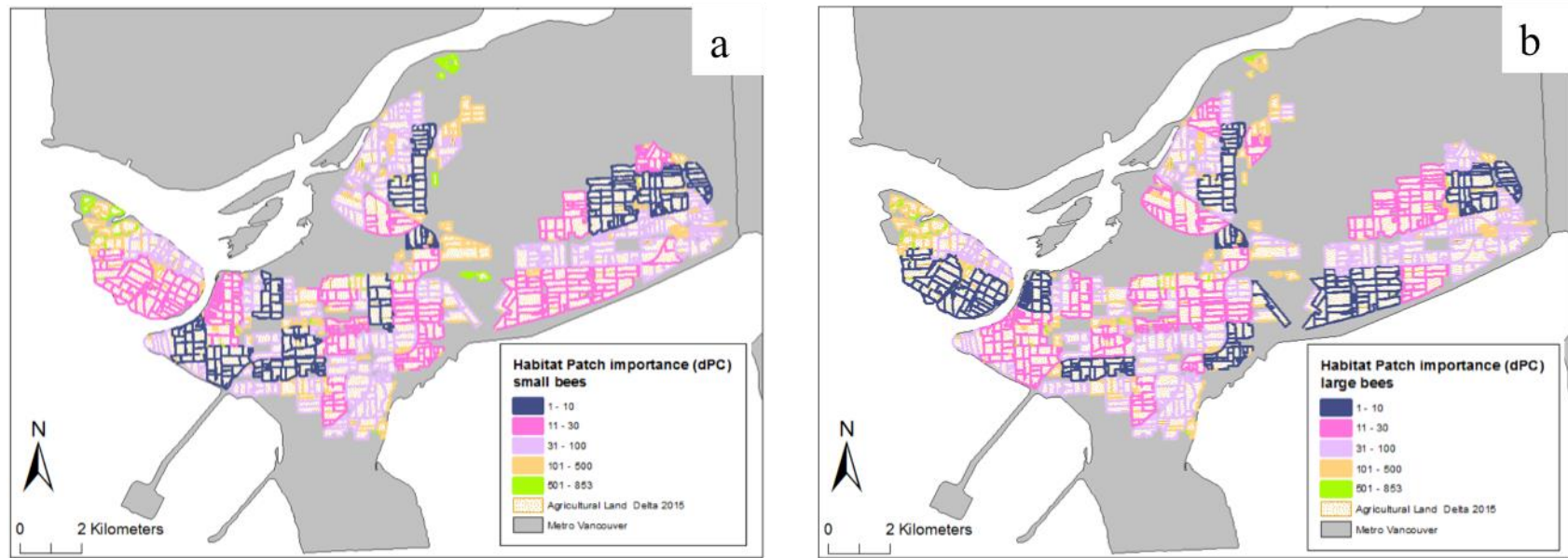
Figure B.1: All (n=853) field margin habitat patches in the Agricultural Land in Delta drawn from combined ranks based on dPC values for groups of a) small and (b) large bees.

Appendix C : Recommended plant species lists

Blooming times and status of listed plant species information in the lists provided below, has been derived from: Agriculture and Agri-Food Canada, Atlas of the Flora of British Columbia, Invasive Species Council of British Columbia and the Inter-Ministry Invasive Species Working Group.

C.1   List of recommended shrub species

Table C.1: List of recommended shrub species to promote or add to the DF&WT Hedgerow Stewardship Program based on a review of the literature specific to the region (DF&WT, 2015b; Mackenzie and Winston, 1984; Neumann et al., 2016; Vaughan et al., 2007). *visited by wild bees in this study

Catergory	Genus	Species	Status	Before berry bloom	Early- and Mid-season	Late-season
DF&WT species	*Spirea*	*douglasii**	native		x	
	Symphoricarpos	*albus**	native		x	
	Cornus	*sericea/ stolonifera**	native		x	
	Rosa	*nutkana**	native	x	x	
	Rubus	*spectabilis**	native	x		
	Rubus	*parviflorus*	native	x		
	Salix	*spp.*	native	x		
	Philadelphus	*lewisii*	native	x	x	
	Ribes	*sanguineum*	native	x	x	
	Rubus	*parviflorus*	native	x	x	
	Rosa	*gymnocarpa*	native		x	
	Mahonia	*aquifolium*	native		x	
	Corylus	*cornuta*	native	x		
	Holodiscus	*discolor*	native		x	x
	Physocarpus	*capitatus*	native		x	
Additional Native shrubs	*Lonicera*	*involucrata*	native		x	
	Gaultheria	*shallon*	native	x	x	
	Rubus	*ursinus*	native	x		
	Viburnum	*trilobum*	native		x	
	Ceanothus	*velutinus*	native	x	x	
	Oemleria	*cerasiformus*	native	x		
	Prunus	*virginiana*	native	x	x	

C.2 List of recommended herbaceous species

Table C.2: List of recommended herbaceous species to add, tolerate or remove in grass stripes buffering DF&WT hedgerows. Recommendations based on (Neumann, 2016; Vaughan et al., 2007). *visited by wild bees in this study

Catergory	Genus	Species	Status	Before berry bloom	Early- and Mid-season	Late-season
flowering herbs	*Vicia*	*cracca**	exotic		x	
	Trifolium	*pratense**	exotic		x	
	Ranunculus	*acris**	exotic		x	x
	Cirsium	*arvense**	noxious		x	x
	Raphanus	*raphanistrum**	exotic		x	x
	Ranunculus	*repens**	of concern	x	x	
	Trifolium	*repens**	exotic		x	
	Cirsium	*vulgare**	of concern		x	x
	Taraxacum	*officinale**	exotic	x	x	x
	Medicago	*lupulina**	exotic	x	x	x
	Geranium	*dissectum**	exotic	x	x	x
	Anaphalis	*margaritacea*	native		x	
	Chamanerion	*angustifolium*	unlisted		x	x
	Fragaria	*chiloensis*	native	x		
	Geum	*aleppicum*	native		x	x
	Lupinus	*arcticus*	native	x		
	Solidago	*canadensis*	native			x

C.3 List of recommended cover crop species

Table C.3: List of recommended cover crop species for pollinators with benefits for soil health, selected from (Vaughan et al., 2007; West Coast Seeds, 2014).*visited by wild bees in this study

Catergory	Genus	Species	Status	Before berry bloom	Early- and Mid-season	Late-season
Cover crops	*Vicia*	*cracca**	exotic		x	
	Trifolium	*pratense**	exotic		x	
	Trifolium	*repens**	exotic		x	
	Helianthus	*annuus L.*	exotic		x	x
	Phacelia	*tanacetifolia*	exotic		x	
	Vicia	*faba*	exotic		x	
	Vicia	*villosa*	exotic	x		
	Crimson	*clover*	exotic	x		
Grasses	*Avena.*	*sativa*	exotic		x	
	Secale	*cereale*	exotic	x		
	Fagopyrum	*esculentum*	exotic		x	
	Hordeum	*vulgare*	exotic		x	