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*A comprehensive assessment of the anticipated climate change impacts to
pollinator populations in agricultural areas and a summary of field data collected
during the 2021 growing season.*

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for the Climate Change Adaptation Program

March 2022

Cover photo

Photo cover page: Vancouver bumble bee (*Bombus Vancouverensis Nearcticus*)

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Project Funding

Funding for this project has been provided in part by the Northern Development Initiative Trust and in part by the governments of Canada and British Columbia under the Canadian Agricultural Partnership, a federal-provincial-territorial initiative. Additional project support was provided by the University of Northern British Columbia. The program is delivered by the Investment Agriculture Foundation of BC.



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Table of Contents

I. Literature Review: anticipating how climate change will affect pollinators in the Bulkley-Nechako and Fraser-Fort George Regional Districts.....	5
Introduction.....	6
Methods	7
Results	7
Review	10
<i>Are pollinators generally in decline?</i>	10
<i>Is the geographic distribution of pollinators changing?</i>	11
<i>Which pollinators are most at risk of decline?</i>	14
<i>Key findings relevant to the BNFFG</i>	16
Theme 1: Temperature.....	17
<i>Is temperature linked to pollinators at a macro scale?</i>	17
<i>Can temperature influence pollinator behaviour?</i>	18
<i>Are pollinators adapting to increased temperatures?</i>	21
<i>Is pollination success affected by temperature?</i>	22
<i>Can greater variability in temperature affect pollinators?</i>	23
<i>Will higher winter temperatures affect pollinator activity?</i>	23
<i>Does overwintering temperature affect pollinator development and emergence?</i>	24
<i>How does temperature interact with other environmental pressures?</i>	25
<i>Key findings relevant to the BNFFG</i>	26
Theme 2: Phenology.....	27
<i>Is the emergence phenology of pollinators changing?</i>	27
<i>Are phenological mismatches occurring?</i>	28
<i>What causes phenological mismatches?</i>	29
<i>What are the consequences of phenological mismatch?</i>	32
<i>Can pollinators and plants avoid phenological mismatches?</i>	33
<i>Key findings relevant to the BNFFG</i>	35
Theme 3: Precipitation.....	35
<i>How does precipitation affect pollination?</i>	36
<i>Key findings relevant to the BNFFG</i>	37
Theme 4: Extreme events.....	38
<i>How do heat waves affect pollinators and pollination?</i>	38
<i>What other extreme weather events may affect pollinators?</i>	39
<i>Key findings relevant to the BNFFG</i>	39

Theme 5: Other co-occurring changes.....	39
<i>How will increased CO₂ affect pollination?</i>	40
<i>Will climate change affect parasites?</i>	40
<i>Key findings relevant to the BNFFG</i>	41
Synthesis: potential targets for adaptation or mitigation.....	41
<i>Target 1: Maintaining non-crop floral resources for pollinators</i>	42
<i>Target 2: Monitoring intensive use of domesticated pollinators</i>	42
<i>Target 3: Maintaining landscape heterogeneity and habitat connectivity</i>	43
Conclusions.....	44
References.....	46
II. Best Practices Review.....	58
Guides for pollinator habitat enhancement.....	58
Interactive tools for pollinator habitat enhancement projects.....	60
Technical guides and BMP documents for land management and agriculture.....	61
BMP for pesticide use.....	63
Multimedia resources (webinars).....	64
Support for pollinator conservation and/or habitat enhancement initiatives.....	65
III. Sampling Protocol.....	67
Preamble.....	67
Protocol aims.....	67
Site selection.....	67
Sampling density & frequency.....	68
Method 1: Pan traps.....	68
Method 2: Vane traps.....	69
Sorting specimens.....	70
References.....	71
IV. Baseline inventory.....	73
Sample collection & processing.....	73
Morphotype identifications.....	73
Taxonomic summary.....	74
Field edge effects.....	77
Haying.....	78
Relative performance: vane and pan traps.....	79
Notes for future sampling in the BNFFG.....	80
References.....	80

I. Literature Review: anticipating how climate change will affect pollinators in the Bulkley-Nechako and Fraser-Fort George Regional Districts

Abstract

Climate change in the Bulkley-Nechako and Fraser-Fort George Regional Districts (BNFFG) is expected to result in altered temperature and precipitation regimes, and growing seasons that start earlier and last longer. A literature review was carried out to evaluate how pollinator insects may be expected to respond to these changes. A literature search yielded 174 articles that satisfied the criteria for inclusion in the review (peer-reviewed, primary research). The majority of studies in the review (112) are predominantly focused on bees (Order Hymenoptera), but flies, beetles, and butterflies and moths (Orders Diptera, Coleoptera, and Lepidoptera) are also represented to a lesser extent. Generally, declines in pollinator insects are being documented in many locations, but they seem to be more likely to occur in areas with more anthropogenic disturbance. Climate change may affect pollinators directly (e.g., heat causes mortality in species with low thermal tolerance) or indirectly, by altering plant growth and physiology (which may alter nectar rewards, pollen quality, timing of flowering, etc.). Pollinator responses to climate change may vary widely, even among closely related taxa, but it is clear that shifts in temperature, precipitation, seasonal phenology, and other factors (extreme events, CO₂ and parasites) affect the abundance, diversity, behaviour, and physiology of pollinator insects. There is general consensus that high pollinator biodiversity buffers systems against pollination declines induced by many of the stressors that are exacerbated by climate change. Thus, managing landscapes to maintain high pollinator biodiversity is likely the best way to protect against future pollination deficits associated with climate change. The literature suggests three general avenues are likely to be critical in planning to ensure pollination services remain healthy in the BNFFG: maintaining non-crop floral resources for pollinators; monitoring how domestic pollinators are affecting wild populations due to competition and spread of disease; and ensuring landscape heterogeneity and habitat connectivity to provide a variety of nesting, predator avoidance, and foraging sites.

Introduction

Many agricultural operators are dependent on the services provided by insect pollinators to produce crops and/or seeds required for future production¹. Adapting agricultural practices to cope with climate change requires consideration of how pollinator abundance, diversity, and interaction with plants may be affected when seasonal patterns of temperature, precipitation, and other factors differ from their historical norms. It is likely that insect pollinators will be affected by climate change because insects have limited thermoregulatory capabilities, and thus their metabolism is largely tied to environmental temperatures. Each species has evolved to perform best within a fixed range of temperatures, termed their thermal tolerance², and so changes in ambient temperatures may alter physiological function³ or cause distribution shifts as populations move in search of better conditions⁴. Additionally, because pollinators are dependent on host plants, climate-induced alterations in the physiology, phenology, and distribution of host plants may indirectly affect the pollinators that depend on them.

The objective of this literature review is to use existing research to evaluate how pollinators in the Bulkley-Nechako and Fraser-Fort George (BNFFG) regional districts are likely to respond to predicted changes in climate. Climate change in the BNFFG is expected to result in higher average daily temperatures (particularly in the summer), a longer growing season (due to more frost-free days and cumulative growing degree-days), more precipitation (in all seasons other than summer, but particularly in the spring and fall), and generally more extreme events (i.e., more days > 30°C, and heavy rain events) by 2050⁵. BNFFG is somewhat unique in that warmer temperatures are not predicted to be accompanied by substantially more arid conditions and prolonged drought, unlike many other areas in the world – though projections of summertime precipitation have significant uncertainty associated with them (change in precipitation by 2050 under Intergovernmental Panel on Climate Change [IPCC] Representative Concentration Pathway [RCP] 8.5, 95% confidence interval = - 16% to 13%⁶).

Insect-pollinated crops in the BNFFG include a variety of forage and cover crops (e.g., alfalfa, clover, some legumes), vegetables, and some fruits⁷. Pollination of these crops may be carried out by both native insects and domesticated honey bees, *Apis mellifera* Linnaeus 1758 (Hymenoptera: Apidae), and any effects of climate change on the abundance, diversity, and/or composition of pollinator communities have the potential to alter the productivity of these crops. However, effects of climate change on the quantity and/or timing of flowering, pollen, and nectar production, and the nutritional quality of crops and other (native) plants also have the potential to affect pollinators⁸, and thus the provision of pollination services.

¹ Klein et al. 2007

² Burdine and McCluney 2019

³ Scaven and Rafferty 2013

⁴ Sirois-Deslille and Kerr 2018

⁵ BC Agriculture & Food Climate Action Initiative 2019

⁶ BC Agriculture & Food Climate Action Initiative 2019

⁷ BC Agriculture & Food Climate Action Initiative 2019

⁸ Hedhly et al. 2009; Waser and Price 2016

Methods

An initial search of the Web of Science database for “climate change” and pollinat* and (insect OR bee) in the topic field yielded 416 peer-reviewed articles. This was further filtered by eliminating articles that were not primary research (i.e., they were review, opinion, or methodological, and lacked any data analysis), and research areas not immediately salient to this review (e.g., geology, molecular biology, etc.; see supplemental information spreadsheet for full records). The final set included 174 peer-reviewed articles. Studies were classified by type (experimental, observational, or predictive model), and taxa of interest prior to in-depth review. Their findings were summarized and organized into thematic areas. The review was structured to answer key questions regarding the current and predicted future response of pollinators to climate change in general, and to specific aspects of climate expected to change in the BNFFG (e.g., temperature, phenology, precipitation, etc.).

Results

Of the 174 articles selected for review, roughly half were observational (88), with the remainder being experiments (48) or predictive models (38). In terms of the taxonomic groups covered, most articles dealt with more than one species (128 multi-species studies; 46 studies were species-specific). Numerous studies evaluated pollinators from more than one taxonomic order simultaneously (52 studies; group-specific information is shown in Table 1), but most studies focused on one or more species in the order Hymenoptera (112). The most frequently studied genus within Hymenoptera was *Bombus* Latreille 1802 (Apidae; 34 studies), followed by *Apis* Linnaeus 1758 (Apidae; 13 studies; see Table 2 for detailed tallies of families and genera that were the subjects of articles selected). Studies that dealt solely with non-Hymenoptera taxonomic orders were rare: four each for Diptera and Lepidoptera, and two for Coleoptera.

Table 1. Studies returned by the literature search, grouped by taxonomic breadth.

Study breadth	Order	Number of studies
Single order		122
	Coleoptera	2
	Diptera	4
	Hymenoptera	112
	Lepidoptera	4
>1 order		52
	Diptera and Hymenoptera	2
	Diptera and Lepidoptera	2
	Hymenoptera and Lepidoptera	6
	Coleoptera, Diptera and Hymenoptera	2
	Diptera, Hymenoptera and Lepidoptera	2
	> 3 orders of pollinating insects	26
	Primarily focused on plants	12

Table 2. Detailed tally of the taxonomic families and species investigated among studies that focused on a single order. Note that the totals for each order do not sum to equal the total number of studies for each order shown in Table 1, because some studies focused on several species from different genera and thus are counted more than once on the table below, while others did not distinguish among families and thus are not counted here.

Order	Family	Genus	Number of studies
Coleoptera	Curculionidae	Elaeidobius	1
Diptera	Muscidae	multiple	2
	Syrphidae	multiple	1
		Merodon	
Hymenoptera	Agaonidae	multiple	1
		Pleistodontes	1
	Andrenidae	Andrena	2
		Apidae	Apis
	Bombus		34
	Braunsapis		1
	Ceratina		2
	Euglossina		2
	Exoneurella		1
	Habropoda		1
	Melipona		6
	Nomada		1
	Plebeia		2
	Trigona		2
	Xylocopa		4
	Colletidae		Colletes
	Halictidae	Halictus	1
		Lasioglossum	1
	Megachilidae	Megachile	2
		Osmia	11
Vespidae	Vespula	1	
Lepidoptera	Nymphalidae	Speyeria	1
	Pieridae	Pieris	1

Review

Are pollinators generally in decline?

Studies conducted at a variety of spatial scales, using data that span decades to centuries, largely indicate that pollinator abundance and diversity in human-influenced habitats is decreasing, while evidence of pollinator declines from studies conducted in less-disturbed habitats is mixed. Oliver et al (2015) found that over the past 40 years, 27% of 720 species that provide pollination services in Great Britain have declined in abundance. Mathiasson and Rehan (2019) evaluated museum specimens collected over a 125-year period (1891 – 2016) in New Hampshire, USA, and determined that the relative abundance of 14 of 119 non-*Bombus* wild bee species had declined, while eight had increased. Though the overall decline in relative abundance in this study was small, the effect on community composition may be substantial (Mathiasson and Rehan 2019). In their assessment of 205 wild bee species in Belgium, Duchenne et al. (2020a) found that over the past 70 years species occupancy had declined by 33% relative to historical samples. In an assessment of museum specimens of 438 bee species in the northeastern US, collected over 140 years, Bartomeus et al. (2013a) found that 56% of all species evaluated had declined slightly, and community composition differed significantly as a result, but only 3 of 187 species that were evaluated individually had declined substantially, all of them *Bombus* spp.

Bumble bees (Hymenoptera: Apidae: *Bombus*) are the genus of pollinators most frequently shown to be in serious decline. In the Pyrenees, Ormosa et al. (2017) found 28 species of bumble bees in contemporary surveys (2006 – 2015), nine fewer than in the past (survey data from 1905 – 2005). In Japan, Suzuki-Ohno et al. (2020) found that populations of five of six bumble bee species (*Bombus diversus* Smith 1869, *B. hypocrita* Pérez 1905, *B. ignitus* Smith 1869, *B. honshuensis* Tkalcu 1968, and *B. beaticola* Tkalcu 1968) they evaluated have declined over the past 26 years (populations of one species, *B. ardens* Smith 1879, have increased). In Colorado, Pyke et al. (2016) showed a decline in the abundance of queens of eight bumble bee species when comparing surveys conducted in 1974 and 2007 – while worker bees did not decline in overall abundance, they were observed at higher elevations than in the past (see below). Graves et al. (2020) documented a 93% decline in probability of local occurrence of *B. occidentalis* Greene 1858 in the continental US between 1998 and 2018. In Vermont, Richardson et al. (2019) found that four of 17 *Bombus* species observed in the past (1915 – 1999) were absent from more recent collections (2000 – 2014), populations of four more had declined significantly, but populations of four others, all from the subgenus *Pyrobombus* (*B. bimaculatus* Cresson 1864, *B. impatiens* Cresson 1863, *B. ternarius* Say 1837, and *B. vagans* Smith 1854), had increased substantially in relative abundance.

For areas of intact, less-disturbed habitat, evidence of wholesale pollinator decline is somewhat mixed. Loboda et al. (2018) documented an 80% decline in the total abundance of muscid flies (Diptera: Muscidae), important pollinators in Arctic ecosystems (Tiusanen et al. 2016), over an 18-year period (1996 – 2014) in Greenland, in addition to widespread reductions in species richness. In an evaluation of butterfly and moth abundance records

collected between 1992 and 2018 across North America, Crossley et al. (2020) did not identify any widespread declines: instead, population declines were generally restricted to areas where conditions were hot and dry. In their study of wild bee, butterfly, and moth assemblages from a protected area (national park) in northern Sweden, Franzen and Ockinger (2012) showed that between 1944 and 2008 (a 64-year interval) the abundance and diversity of butterflies and moths increased, while wild bee species remained stable. While there were shifts in community composition and changes in the elevational ranges of some species (see below), the authors noted the changes they documented were small, particularly in comparison to what had been reported from neighbouring agricultural areas in northeastern Europe (Franzen and Ockinger 2012). Herrera (2019) quantified pollinators visiting 65 plant species on sites in undisturbed Mediterranean montane habitats, collecting observations in at least two different years over a period of two decades (1996 – 2016). Despite substantial interannual variability, pollinator functional abundance (visitation probability to patches and/or plants) increased over the study period, primarily due to increased visitation by small solitary bees (Hymenoptera: several genera within the families Megachilidae, Andrenidae, and Colletidae) and small beetles (Coleoptera: genera from within the families Buprestidae, Dasytidae, Scarabaeidae, and Meloidae; Herrera 2019). Dupont et al. 2009 quantified spatio-temporal variation in species composition and structure of pollination networks from six sites (most within parks or conservation areas) on a large latitudinal gradient (from 28 to 74 ° N), demonstrating high interannual consistency in the number of species and interactions, and network structural parameters (connectance, nestedness, modularity and centralization) within all sites. While there was variation in the composition of species and identities of interactions (high turnover in the identities of floral visitors in particular), climate variables did not significantly influence network parameters over the period assessed, and the authors interpreted the stability in the network as indicative of some resilience to change or extinction of species, as they may be replaced by species with similar ecological roles (both pollinators and plants; Dupont et al. 2009).

Is the geographic distribution of pollinators changing?

The geographic distribution of a pollinator typically reflects the suitability of the environment for a given species (climate and host plant availability). Numerous studies have used species distribution models to document changes in the geographic range (area and/or location) of suitable habitat occupied by pollinators and their host plants, or to predict the future distribution of pollinator species under climate change. The geographic distribution of many species is predicted to shift, and in some cases be reduced in area, as climate change alters which areas are suitable for occupation. Many species are expected to shift their range toward higher latitudes or elevations in search of cooler temperatures.

In the UK, Polce et al. (2014) assessed the potential for climate change to affect orchard plants, 22 species of bee (Hymenoptera: Andrenidae [five spp.], Apidae [seven spp.], Halictidae [seven spp.], and Megachilidae [three spp.]) and eight species of hoverfly (Diptera: Syrphidae) pollinators, between the present day and 2050. They predicted that most

species were likely to shift northward, and that 20 species (14 of 22 bee species, and six of eight hoverfly species) would experience an expansion in range area, while the remaining ten would contract; by 2050 the areas considered most suitable for orchards are areas predicted to have low pollinator availability (Polce et al. 2014). Bezerra et al (2019) evaluated in the Neotropics how passion fruit (*Passiflora edulis* Sims; Malpighiales: Passifloraceae), and its two main pollinator bee species (*Xylocopa frontalis* Olivier 1789 and *X. grisescens* Lepeletier 1841; both Hymenoptera: Apidae) may be affected under two IPCC environmental scenarios. They predicted substantial declines in habitable area for both the pollinator species and for the host plant at two time points under both IPCC scenarios (declines of 15.4 to 57.7% for pollinators, and 42.9 to 64.8% for passionfruit). Most notably, they found that the spatial overlap of areas suitable for the plant and its pollinators is likely to decline 31.9 – 54.9% (Bezerra et al. 2019). The distribution and richness of wild buzz-pollinators needed to pollinate field tomato crops is also expected to decline in areas with significant tomato crops by 2050, in both the US and Brazil (15 species in the US: Carrasco et al. 2020; 5 species in Brazil: Elias et al. 2017). In a study of ten wild bee species in Brazil, many of which provide crop pollination services, Giannini et al. (2012) predicted that, under two different IPCC scenarios (RCP 4.5 and 8.5), all but one species would experience a decrease in total area of suitable habitat by 2050. The two species expected to have the greatest decrease in area both presently occupy areas with cooler, more humid climate (Atlantic Moist Forest ecosystem type). In a subsequent study, Giannini et al. (2020) predicted how climate change would affect 216 bee species in the Eastern Amazon (Brazil), and showed likely declines in the occurrence area of 95% of species by 2070, and a loss of climatically suitable areas within their current distribution for 85 – 96% of species (depending on the IPCC scenario used). Of particular importance is the prediction that bee species with smaller geographic distributions and important crop pollinator species are likely to experience higher losses of occurrence area than others (Giannini et al. 2020). Also in Brazil, Marchioro et al. (2020) predicted that species-specific responses to climate change of the plant *Mimosa scabrella* Benth (Fabales: Fabaceae) and its pollinators, *Melipona quadrifasciata anthidioides* Lepeletier 1836 (Hymenoptera: Apidae) and *M. q. quadrifasciata* Lepeletier 1836 will lead to a reduction in the spatial association of these species of 71% in the coming years under moderate IPCC projections (RCP 4.5). For *M. quadrifasciata*, a predicted shift southwards and inland during the next 70 years may also cause a pollination deficit in important coffee-growing regions in Brazil (Giannini et al. 2015a). In a broader evaluation of coffee-growing regions across Latin America, Imbach et al. (2017) predict a decline in bee richness over 34 – 51% of the area presently used to grow coffee by 2050, which may make cultivation more difficult, and perhaps impossible.

In southeastern Europe, Milicic et al. (2018) forecasted the distribution of 44 species of hoverflies (Diptera: Syrphidae), and found that in the near future (2041 – 2060), 25 species were predicted to have reductions in their ranges, while 19 were likely to have increased areas of occupancy. The predicted change in range size for most species was < 20%, which the authors suggest is an indication of resilience of hoverflies in general to climate change; however, because there is likely to be a loss of species at lower elevations, where agriculture typically takes place, pollination services provided by hoverflies in some areas may be

compromised by climate change (Milicic et al. 2018). Giannini et al. (2020) and Polce et al. (2014) both indicate that habitat generalists are predicted to dominate species assemblages in the future, and in all of the studies reviewed above, crop pollination services are likely to decline in the absence of mitigation or adaptation measures, due to changes in the geographic distribution of pollinators.

Patterns of change in geographic distribution are likely to be species-specific; even species which are phylogenetically close may have differences in how their range is altered by climate change. Francoso et al. (2019) found that the area suitable for bumble bees *B. morio* Swederus 1787 (panmictic) and *B. pauloensis* Friese 1912 (several phylogeographic lineages) in Brazil is likely to decrease by > 45% by 2070. For *B. pauloensis*, the lineages occupying the edges of the species' distribution will be most affected, while those in the center of the range will not experience significant reductions. Similarly, Giannini et al. (2017) predicted that a locally important pollinator, *Melipona subnitida* Ducke 1911 (Hymenoptera: Apidae) in Brazil will shift towards the edges of its current distribution, which may cause reduced gene flow and limit dispersal, threatening population persistence. In a related finding, Jaffe et al. (2019) showed that gene flow among *M. subnitida* populations was linked to increased forest cover and thermal stability - which are both reduced by deforestation. Giannini et al. (2013) predicted that, due to changes in the distribution of 33 potential host plant species, *Xylocopa* bee species will likely shift northward in Brazil by 2050, even under moderate climate change scenarios. Martins et al. (2015) predicted that areas suitable for threatened South American bumble bee, *Bombus bellicosus* Smith 1879, will migrate southwards. While Nemesio et al. (2016) found that deforestation was a larger threat than climate change for *Euglossa marianae* Nemésio 2011 (Hymenoptera: Apidae), a species endemic to the Brazilian Atlantic Forest, they also showed that the areas with the best climatic conditions for this species will likely expand southwards in the future, which is problematic given its limited dispersal capabilities. Faleiro et al. (2018) found that orchid bee (Hymenoptera: Apidae; 37 spp. in the genus *Euglossa* were evaluated) populations in the Brazilian Atlantic Forest would likely decline in the future, but that some climatically appropriate areas will be retained. In contrast with others, they found that changes in richness and community composition would largely occur at the western and southern borders of the area evaluated (Faleiro et al. 2018).

One change in distribution that has been noted in numerous studies is a shift in elevation for many species, as higher elevation sites are generally cooler than those lower down, and may be subject to less anthropogenic disturbance. The clearest evidence of widespread elevational shift is found in studies on bumble bees, but similar shifts in butterflies and moths have also been documented. Nooten and Rehan (2020) evaluated 125 years of museum specimens from four *Bombus* species in New Hampshire, USA and found that all species showed a shift toward higher elevation, regardless of their current conservation status (declining, stable, or increasing). In an assessment of six bumble bee species in Japan over 26 years, Suzuki-Ohno et al. (2020) found that all have shifted to occupy higher elevation habitats than historically. Ornos et al. (2017) also found in their study of *Bombus* in the Pyrenees that at high and low elevations, species showed a tendency towards a reduced altitudinal range and a preference for subalpine habitats that were part of protected areas. In Colorado, Pyke et al (2016)

documented upward shifts in elevation between 1974 and 2007 for six of eight bumble bee species they studied. Mathiasson and Rehan (2019) found that of the 14 non-*Bombus* wild bee species in New Hampshire known to be in decline, ten had also shifted northward and/or were currently found at higher altitudes than they were historically. In a national park in northern Sweden, Franzen and Ockinger 2012 found that among the 46 bee and butterfly species present in surveys conducted in 1944 and 2008, 17 were found at the same elevation in both surveys, 17 were found at lower elevations in 2008, and 12 were found at higher elevations in 2008 than in 1944. The resulting changes in community composition were greatest at higher altitudes, due to the presence of new high alpine species, but this study also showed that southern species (new to this region) had colonized habitats at lower alpine altitudes (600 – 800 m a.s.l.; Franzen and Ockinger 2012).

Responses to climate change are likely to be species-specific, and not all pollinators are predicted to decline or have large geographic shifts. Studying 12 wild bee species in South Africa, Kuhlmann et al. 2012 found that two species were likely to experience range expansion, while range contractions of 32% – 99% were predicted for six species; four species are expected to have large shifts in their range. Stable or increased populations in the future were predicted for two Australian species, both adapted to arid conditions. A small carpenter bee, *Ceratina australensis* Perkins 1912 (Hymenoptera: Apidae) is expected to have an expanded range in the future (Dew et al. 2019), and the same trend is expected for an arid-zone bee, *Exoneurella tridentata* Houston 1976 (Hymenoptera: Apidae; Silva et al. 2018). Silva et al. (2017) predicted that the distribution of introduced allodapine bee, *Braunsapis puangensis* Cockerell 1929 (Hymenoptera: Apidae), in Fiji is likely to expand in the South Pacific with climate warming, and because of its resistance to honey bee pathogens, it may replace *Apis mellifera* in the region as an important crop pollinator if *A. mellifera* populations decline. In Brazil, increased ambient temperatures expected under IPCC projections are predicted to decrease imported *Apis* populations, but not native *Trigona spinipes* Fabricius 1793 (Hymenoptera: Apidae; Gianni et al. 2015b). Hoverflies have also been identified as one group of pollinators that may be particularly resilient to climate change, and many species within this family may experience stable or increasing areas of distribution in the future (Milicic et al. 2018; Polce et al. 2014).

Which pollinators are most at risk of decline?

There is little agreement regarding which species are most likely to decline due to climate change, but bumble bees are of significant concern in many ecosystems. Phylogeny does not seem to consistently predict whether a species is at risk, but there is some evidence that species which are less common and/or distributed over smaller geographic areas may be more likely to decline than others, which is often related to the effects of land-use change at a local scale. Studies on bees generally identify large body size and lower critical thermal maxima as being associated with declines (Bartomeus et al. 2013a; Nooten and Rehan 2020), but it is unclear whether these findings extend to other taxonomic orders.

In their assessment of 119 non-*Bombus* wild bee species over 125 years, Mathiasson and Rehan (2019) found that guild affiliation did not predict the current status of a species (increasing, decreasing, or stable). Ghisbain et al. (2020) did not find that phylogeny predicted the response of 68 west Palearctic bumble bee (*Bombus*) species to climatic or land-use change. However, in a global assessment of 112 species of *Bombus*, Arbetman et al. (2017) found that species in decline are more likely to be from the subgenus *Thoracobombus* (~ 64% of species assessed) and less likely to be in the subgenus *Pyrobombus* (~ 6%). An interesting example of differing responses to climate change among closely-related species is provided by Milic et al. (2019), who modeled the response of three cryptic species within the hoverfly *Merodon atratus* species complex (*M. atratus* Oldenberg 1919, *M. balkanicus* Šašić 2016, *M. virgatus* Vujić & Radenković 2017) to climate change, and found they are likely to experience different outcomes, despite their morphological similarities. Generally, the range of this complex is expected to decrease and move into the Alps, but *M. balkanicus* is projected to go extinct due to climate change, while the other two closely-related species are expected to persist (Milic et al. 2019). However, populations with greater genetic diversity may have a better chance of persistence in the future – using microsatellites, Koch et al. (2018) identified patterns of greater genetic diversity in lineages of *Bombus huntii* Greene 1860 that have inhabited climatically unstable areas since the last glacial maximum.

Using simulations to identify what demographic factors had the greatest effects on pollinator diversity and abundance, Ramos-Jiliberto et al. (2020) identified increased larval mortality and increased competition for space among larvae as the most important contributors to pollinator diversity declines, but that decreases in pollinator biomass were linked to increased adult mortality. Small geographical ranges and a lack of internal parasites (and thus reduced immune capacity) were associated with greater vulnerability to extinction in bumble bees (Arbetman et al. 2017). In their study of West Palearctic bumble bees, Ghisbain et al. (2020) also identified limited current geographic distribution as a risk factor for decline. Species distribution models generated by Ghisbain et al. (2020) showed that between 1970 and 2000 the occurrence of localized bumble bee species was best predicted by specific land-cover characteristics, while climatic characteristics best predicted the occurrence of widespread species, indicating that land use change, not climate, was a greater threat for bumble bees with small geographic ranges. Mathiasson and Rehan (2020) also noted the importance of specific host plants when they characterized plant-pollinator networks in New England over 125 years, during which the number of exotic bee and plant species in the region had both increased substantially. They also found that wild bees which are presently in decline were often associated with threatened and endangered plant species, and that a number of specialist pollinators were now extinct, likely due to local extirpation of host plants (Mathiasson and Rehan 2020). In Greenland, Loboda et al. (2018) found that less common species of muscid flies (Diptera: Muscidae) were much more affected by climate change over an 18-year period than common ones. In contrast, Milicic et al. (2018) found that hoverfly species with existing small ranges were not predicted to lose proportionally more area than species that currently have a large spatial distribution.

In a study of four *Bombus* species, using 125 years of museum specimens, Nooten and Rehan (2020) found that species currently in decline tended to have larger body sizes than species

not in decline. In their assessment of 438 bee species in the northeastern US over a 140-year period using museum specimens, Bartomeus et al. (2013a) identified that declining abundance was associated with narrow dietary niche, small phenological breadth, and large body size. In comparing nine native bee species in Costa Rica during a three-year period that included El Niño and La Niña years, Frankie et al. (2005) also found that species with larger body sizes showed greater interannual variation, which they attributed to a greater sensitivity to climate-driven variation in floral resources. Scheper et al. (2014) determined that declines in wild bee species in the Netherlands were linked to larger body size and reduction in the availability of preferred host plants, which indicate that wild bees may be declining in some areas due to food limitation; larger bees are particularly at risk because they require more pollen. In this study, other traits (diet breadth, length of flight period, sensitivity to climate change) did not explain declines as well as body size and host plant availability did (Scheper et al. 2014).

Hamblin et al. (2017) found that, in sampling 15 wild bee species near Raleigh, NC, the species that had the lowest critical thermal maxima (solitary species and bumble bees) declined the most in response to warming. In the northeastern US, Bartomeus et al (2013a) showed that, over a 140-year period, bee species with increasing relative abundance tended to have range boundaries closer to the equator, indicating they were tropical-adapted or – tolerant, and thus perhaps pre-adapted to cope with climate change.

Key findings relevant to Bulkley-Nechako and Fraser-Fort George

1. There is good evidence to suggest pollinator populations are affected by climate change: many species are in decline, while others are shifting to live at higher elevations or latitudes (which are generally cooler than sites at lower elevation and/or closer to the equator). Climate change-related declines in pollinators seem to be more likely to occur in areas with more anthropogenic disturbance. Development and land use change in areas adjoining agricultural zones may have greater consequences for pollination services than would be expected in the absence of climate change.
2. Pollinator population responses to climate change may vary widely, even between closely related taxa. Larger-bodied pollinators (specifically bees from the genus *Bombus*), and species with limited thermal tolerance seem more likely to be affected by climate change than others. There is mixed evidence that species with smaller geographic ranges, and those with more specialist dietary preferences (i.e., only a few suitable host plant species) may also be at greater risk of climate change-induced declines.
3. General (large-scale) trends in pollinator occurrence or abundance may not be representative of changes occurring in a given locality. Shifts in the geographic distribution (area and/or location) of pollinator species should be expected, but are generally difficult to predict in the absence of a detailed understanding of the ecology and physiology of a given pollinator species (e.g., preferred host plants, phenology, thermal tolerance, etc.) and highly localized data regarding land use and plant communities. However, if these data can be collected, new species distribution modeling tools can

produce high quality, accurate predictions that are likely to be useful in anticipating changes in population distributions and identifying areas at risk of pollinator deficits in the future.

Theme 1: Temperature

Temperatures in the BNFFG region are expected to increase, particularly in summer; under the most extreme IPCC scenario (RCP 8.5) annual mean temperature is predicted to be 3.1 – 3.2°C higher across the region by 2050 (BC Agriculture & Food Climate Action Initiative 2019). Temperature may affect insect pollinators directly and indirectly. Direct effects occur due to the fact that insects are ectotherms, and so their activity levels, metabolism, and development (and therefore need for nectar and pollen) are expected to increase with ambient temperature. However, because high temperatures can damage proteins and cell structure, and melt wax (Martins 2015), excessively hot conditions may potentially lead to mortality of individuals and (for social species) failure of colonies. Indirectly, temperature can affect insect pollinators via effects on host plants. If temperature can alter growth, nectar and/or pollen production of plants, the types and number of pollinators attracted to plants may change. Warmer temperatures, in the absence of water shortage, may promote vegetative growth (Osborne et al. 1997), but effects on floral traits (which are what pollinators are attracted to) may vary.

Is temperature linked to pollinators at a macro scale?

Temperature is among the primary factors influencing the distributions of pollinator species, and shifts in temperature are expected to continue to shape community composition and biodiversity in the future. In a study of the bee faunas in the southwest Pacific archipelago (Fiji, Vanuatu, Samoa), Groom et al. (2014) found that during the last era of significant climate change, following the last glacial maximum (~ 10 000 years ago), all three islands saw massive population declines prior to the glacial maximum and rapid expansion thereafter (when the climate was warming). However, during this period there was little human influence in terms of land use or urbanization, which are likely to interact with climate to influence how pollinators respond in the present day. In a study of the current climate regime shift, Aguirre-Gutierrez et al. (2017) found that, in a long-term dataset collected on 398 Dutch pollinator species (bees, butterflies and hoverflies) in three periods (1951 –1970, 1971 –1990 and 1998 –2014) temperature was generally more important than precipitation in influencing the distribution of pollinators, and that the influence of temperature on the distribution of bees and hoverflies was greater in more recent time periods than in the past. In Germany, from 2010 – 2012, bee species richness across multiple sites declined as temperature rose (after correcting for phenology; Papanikolaou et al. 2017a). Loboda et al. (2018) showed that, over an 18-year period (1996 – 2014), summer temperatures in the Arctic were an important predictor of the abundance of Dipterans from the Muscid family, and that over this period, there was an 80% decline in overall abundance of muscids associated with

increasing temperatures. The recent (ca. 1920) north westward expansion of bumble bee *Bombus haematurus* Kriechbaumer 1870 from Serbia into Central Europe was associated with warming winter temperatures in recently colonized areas, particularly during the queen overwintering and emergence phases: warmer winters were identified as a significant promoter of colonization of new areas for this species (Biella et al. 2020). Chen et al. (2018) predict northward expansion of figs and their pollinators, after finding that temperature constrains reproduction at the northern edge of their range in China (25 ° 50' N). Teixeira et al. (2018) predicted the expansion of one subspecies of *Melipona quadrifasciata*, *M. q. anthidioides*, but not another, *M. q. quadrifasciata*, in Brazil over the next 80 years, due to differences in thermal tolerance.

Comparisons along elevation gradients are also used to assess how temperature may affect pollinators and plants, as lower elevations typically experience warmer conditions relative to sites higher up. Adedoja et al. (2018), Classen et al. (2015), Lara-Romero et al. (2019) and Minachilis et al. (2020) found that pollinator community composition differed with elevation, in South Africa, Tanzania, the Canary Islands, and Greece, respectively. The studies from Africa and the Canary Islands found that the diversity and abundance of bees was lowest in the highest elevation zone (coldest), and in two studies this was linked more strongly to temperature than to floral resources or level of habitat disturbance (Adedoja et al. 2018, Classen et al. 2015). Comparing how four major groups of pollinators (bees, wasps, beetles, and flies) varied with elevation, Adedoja et al. (2018) found that bees and beetles, but not wasps and flies, were affected by reduced temperatures, differences in abiotic weather conditions (e.g. strong winds), and lower flowering plant diversity at higher elevations. As a consequence, interaction networks involving bees and beetles differed with elevation, but networks that involved flies and wasps did not (Adedoja et al. 2018). Along an elevational gradient (2350 – 3520 m a.s.l.) in the Canary Islands Lara-Romero et al. (2019) also documented an increase in average body size of pollinators as elevation increased (largely due to an increase in bumble bees); the pollinator-plant network also had progressively less specialization and lower modularity as elevation increased (i.e., more species of pollinators interacted with a given species of plant, and a given species of pollinator was more likely to visit multiple plant species). In Greece (farther from the equator than Africa or the Canary Islands), Minachilis et al. (2020) found that bumble bee richness was greatest at the highest elevation surveyed, while plant richness was greatest at lower to intermediate altitudes. However, the bumblebee-plant visitation networks were larger, more diverse, and more generalized at intermediate altitudes.

Can temperature influence pollinator behaviour?

Numerous studies have demonstrated a positive relationship between ambient temperature and pollinator visitation rates, in a variety of insect species (bees, beetles, and flies: Adedoja et al. 2018, bees: Classen et al. 2015, butterflies: Torres-Diaz et al. 2007; bumble bees: Sanderson et al. 2015). However, warmer ambient temperatures may also cause pollinators to spend more time thermoregulating, and less time pollinating – particularly for pollinators that

live in colonies, because it is more difficult to regulate the temperature of a hive than the body of a single individual.

Increases in floral visitation rates with rising temperatures seem more likely in temperate zones, as Antiqueira et al. (2020) found that maximum daily temperatures did not affect pollinator visitation to roseleaf bramble (*Rubus rosifolius* Smith; Rosales: Rosaceae) in Brazil (tropical). Physical traits may also be linked to how pollinators respond to ambient temperature. Cui and Corlett (2016) found in observing floral visitors in Yunnan, China (21 ° 41' N) that small, open-nesting mining bee species *Apis florea* Fabricius 1787 (Hymenoptera: Apidae) and *A. andreniformis* Smith 1857 increased in abundance with temperature, and in the case of *A. andreniformis*, humidity as well. Larger, related species, *A. cerana* Fabricius 1793 and *A. dorsata* Fabricius 1793, declined in abundance with temperature.

Thermodynamic modeling shows that as temperature increases, honey bees perform less work and generate more heat, which is not efficiently dissipated from the hive. As a result, 1°C of warming can create 90% more entropy stress (energy released as heat rather than used to do work) for honey bees, relative to that of cooling (Yildiz and Ozilgen 2019). The large carpenter bee, *Xylocopa frontalis*, alters its activities in response to body heat generation and ambient temperature, thermoregulating to reduce body heat using air convection, nectar dehydration (evaporation), and reducing foraging activity during the hottest times of day (de Farias-Silva and Freitas 2020). In a study that manipulated the temperature of *Bombus terrestris* Linnaeus 1758 colonies, workers in a high-temperature treatment spent more time wing-fanning – which may be energetically costly – and performed less wax canopy construction (Holland and Bourke 2015).

Increases in activity caused by temperature may not always be due to direct effects on metabolism, but may also be driven by changes in plant traits. In an experiment, Parsche et al. (2011) demonstrated differences in the response of bees and flies to an increase in vegetative growth expected under climate change. Flies visited flowers of *Sinapis arvensis* L. (Brassicales: Brassicaceae) placed at greater heights above the ground more often, while bees did not. However, seed set was generally high across all treatments, which indicates that this plant-pollinator system had sufficient redundancy to ensure pollination despite environmental variability (Parsche et al. 2011). Russo et al. (2020) showed a weak positive effect of ambient temperature on visitation rates to nodding thistle (*Carduus nutans* L.; Asterales: Asteraceae), which they attributed to temperature-induced increases in pollen lipid content, increasing its attractiveness to pollinators. Hoover et al. (2012) used an experiment to show that increasing temperature accelerated flowering and increased the number of flowers and the quantity of nectar produced by pumpkin (*Curcubita maxima* Duchesne; Cucurbitales: Cucurbitaceae) plants, but this did not alter the volume of nectar consumed by bumble bees (*Bombus terrestris*) or the rate at which they visited flowers. Descamps et al (2020) found that for two bee-pollinated Boraginaceae species, annual *Echium plantagineum* L., and biennial *Echium vulgare* L., exposed to increased temperatures, flowering was accelerated, but floral traits that attract pollinators were generally negatively affected. Specifically, flower size and nectar rewards were decreased, particularly in the annual species (Descamps et al. 2020). Takkis et

al. (2015) showed that nectar production peaked at different temperatures for two widespread Mediterranean plants, *Ballota acetabulosa* (L.) Benth and *Teucrium divaricatum* Sieber ex Heldr (both Lamiales: Lamiaceae). Under some climate warming projections, daily temperatures where those two species live will exceed estimated critical high temperature values, and thus the authors predict reduced nectar secretion and therefore fewer resources available to bees in the future.

Does temperature affect pollinator survival and reproduction?

The consequences of increased temperatures for pollinators vary among species and depend on the amount of warming that occurs. In many studies, increased temperatures are associated with negative outcomes only when they are accompanied by low humidity or drought. Increased adult mortality with increased temperature has been documented in some species, particularly those with larger body sizes (e.g., Megachilidae), but that is often associated with temperature effects on resources rather than a direct effect on pollinator physiology. In an observational study, Switanek et al. (2017) found that honey bee colonies in Austria had greater mortality when conditions were warmer and drier, but that higher ambient temperatures alone (in the absence of changes in precipitation) did not increase mortality. The authors attributed increased mortality to an indirect effect of changes in floral resources caused by warm, dry conditions, not a direct effect on bees (Switanek et al. 2017). Sutton et al. (2018) found that an Australian temperate-zone fig wasp, *Pleistodontes imperialis* Saunders 1882 (Hymenoptera: Agaonidae) survival to emergence declined above 39°C, and that adult longevity declined slightly above 30°C, and much more substantially above 35°C. Fitness consequences were observed at temperatures 5°C above the summer daily mean maximum. Similarly to Switanek et al. (2017), longevity was more strongly affected by temperature when humidity was low (Sutton et al. 2018). Notably, these problematic effects of increased heat and aridity may not be restricted to Hymenoptera: in a study of 456 butterfly species across North America, Crossley et al. (2020) found that population declines occurring between 1993 and 2018 were clustered in areas where conditions became hotter and drier, while populations on sites that remained cooler and wetter did not have decreases in butterfly abundance.

For some species, increased temperatures seem to be problematic even when humidity does not vary. CaraDonna et al. (2018) altered the temperatures for *Osmia ribifloris* Cockerell 1900 (Hymenoptera: Megachilidae) during larval development through adult emergence by painting nest boxes, and found that in warmed boxes (1.8°C to 2.6°C higher than controls), mortality was much greater, varying from 30% to 75% between years. Body mass and fat content of bees were also reduced. Radmacher and Strohm (2011) documented increased mortality of *Osmia bicornis* Linnaeus 1758 adults, but not larvae, kept at warmer conditions. Becker et al. (2018) experimentally manipulated the temperature of colonies of *Melipona interrupta* Latreille 1811 (a eusocial stingless bee in Brazil), and observed that temperatures above or below typical ambient (28 to 30°C) were associated with increased mortality of immature bees, and lower male:female ratios. At higher temperatures, queens were produced more frequently, but overall caste ratios were unaffected.

Other species may be resilient to, or even experience some benefit from, rising temperatures. For pollen specialist bee, *Osmia iridis* Cockerell & Titus 1902, monitored in subalpine habitats over three years, Forrest and Chisholm (2017) found that exposure to increased temperatures (number of hours at > 16°C) was correlated with greater floral resource availability and also promoted nest provisioning by the bees. But those seeming benefits did not increase lifetime reproductive output due to the higher costs in warmer temperatures of increased rates of brood parasitism (by wasp *Sapyga pumila* Cresson 1880). Higher rearing temperature (25°C vs. 20°C) of *Bombus terrestris* colonies was associated with greater colony size and number of queens produced, but did not alter the timing of peak colony size or when males were produced. No clear effects on longevity were observed (Holland and Bourke 2015). In a manipulation of *Bombus terrestris* hives conducted by Zaragoza-Trello et al. (2020), increased temperatures were also associated with consistently larger queens and workers and no difference in longevity.

Are pollinators adapting to increased temperatures?

The evolution of smaller body size (better for dissipating heat) in response to climate change has been investigated in several taxa, and results appear mixed – other factors, such as land-use change, may also contribute to long-term changes in morphology. Costa et al. (2017) found that temperature, rather than resources, was the most important determinant of the average body sizes of several groups of pollinators (bees, moths, and frugivorous birds) living along an elevational gradient in Tanzania. Nooten and Rehan (2020) showed that, in the northeastern US, all four *Bombus* species they evaluated have evolved smaller body sizes over the past 125 years, with larger-bodied species showing a greater decline in size than smaller-bodied species. The authors identified these body size changes as indicative of adaptation to a change in environmental conditions (Nooten and Rehan 2020). Peters et al. (2016) collected 35 bee species in Germany and found that smaller body size, shorter hair, and longer wings were associated with higher thermal activity thresholds. Those same traits were negatively correlated with elevation in samples collected in the field, at both the individual and species level (Peters et al. 2016). However, Gerard et al. (2021) determined that over 115 years (1901 – 2016) the body size of five bumble bee species in Europe had changed in response to altered landscape composition, not climate change. Oliveira et al. (2016) also found significant declines in the size of 18 large-bodied bee species in Holland, but only among females, over 147 years. They also identified land use, rather than changes in temperature during the flight period of bees, as the primary cause of change in body size of female bees (Oliveira et al. 2016). In their study of body size evolution of herbivorous beetles (29 species, not all are pollinators) in Israel, Baar et al. (2018) found that over the past 100 years there appeared to be no consistent relationship of body size with environmental variables in beetles. However the two of the three of click beetle (Coleoptera: Elateridae) species assessed, which do perform some pollination, did show a reduction in body size with temperature.

Other evolved adaptations that have been documented in pollinators to cope with warmer climates include changes in thermal tolerance, altered developmental rates, and shifts in

morphology of mouthparts related to changing floral resources. Gonzalez et al. (2020) showed that the critical thermal maximum of carpenter bees (*Xylocopa violacea* Linnaeus 1758) in Greece decreased with elevation, indicating that lower-elevation populations were likely to have better tolerance for hot conditions. Badejo et al. (2018) found that among wasps (*Vespula vulgaris* Linnaeus 1758) collected in Finland, those found at higher latitudes were darker and larger, characteristics associated with a thermoregulatory advantage in cold climates. A common garden experiment conducted on *Osmia lignaria* Say 1837 from three different regions (Utah, Washington, and California) by Pitts-Singer et al. (2014) showed a strong effect of region of origin on survival and developmental physiology: larvae from California developed more slowly but were more metabolically active and had greater survival than others when reared in Californian conditions. Miller-Struttmann et al. (2015) determined that tongue length in two alpine bumble bee species decreased over 40 years, despite no increase in small-flowered plants or co-evolution of shallower flowers in plants historically preferred by bumble bees. The authors argued that these bumble bee species have likely become more generalist in their foraging due to fewer floral resources, which they linked to warmer summers (Miller-Struttmann et al. 2015). Schurch et al. (2016) predicted that, under climate change, sociality in *Halictus rubicundus* Christ 1791 (Hymenoptera: Halictidae; a facultatively social species which forms colonies in warmer locales) is likely to arise in northerly locations, where it has not occurred in the past, and this is likely to result in more workers available to provide pollination services in these areas – if sufficient floral resources are present.

Is pollination success affected by temperature?

Despite the typically positive effects of temperature on visitation rates in temperate zones, species-specific responses may not always benefit pollination success. Plants with diverse pollinator assemblages are likely to experience fewer consequences of changes in temperature than are specialists. In western Greenland, Urbanowicz et al. (2018) observed that in warmer temperature zones, higher pollinator visitation was linked to lower pollen limitation and higher fruit set for plants reliant on insect-mediated outcrossing for maximum reproductive success. Along an elevation gradient in the Coast Range Mountains of BC, Straka and Starzomski (2015) showed that the seed set of bee-pollinated plant species was best predicted by degree-days >15°C, but that seed set in general was higher for plants that had more potential pollinators. Rader et al. (2013) predicted that under the most extreme IPCC scenarios in the Northeastern US, watermelon pollination services provided by managed pollinator *A. mellifera* will decline by 14.5%, while services provided by wild bees are likely to increase, resulting in a net increase in pollination services of 4.5% by 2099.

When Richman et al. (2020) transplanted alpine meadow turfs to lower elevations to introduce them to warmer climates (simulating climate change), pollinator visitation rates were greatest for transplants that experienced moderate warming, at mid-elevations; lower elevations were much warmer and had fewer pollination visits to transplanted plots, but visitation rates at surrounding native vegetation were greatest. In this study, two of three alpine plant species transplanted to lower elevations had lower reproductive success at the

warmest sites, and the authors attributed this to the lower pollination visits these plants received from key pollinator species (bumble bees in particular); the third species appeared to be less specialist in terms of pollinator requirements, and thus did not have the same decline in reproductive success as other transplanted species (Richman et al. 2020). There is some additional evidence that plants that rely on bumble bees may have reduced success when temperatures rise, as Greenop et al. (2020) used field cage experiments to show that faba bean (*Vicia faba* L.; Fabales: Fabaceae) plants pollinated by buff-tailed bumble bees (*Bombus terrestris*) kept at higher temperatures (31 vs. 25°C) had lower seed set and reduced total bean weight.

Can greater variability in temperature affect pollinators?

Only two studies found in the literature search directly assessed how variability in temperature may influence pollinators: one used observational data to assess how the stability of wild bee populations may be affected, while the other tested how variability in temperature may alter development and survival of pollinators. In a study of wild bees conducted from 2010 – 2013 in Germany, Papanikolaou et al. (2017b) found that when intra-annual variability in temperature and precipitation was high, stability in wild bee abundance declined, but this was buffered in more heterogeneous landscapes, which had more stable abundance in wild bees even when temperature was highly variable. Radmacher and Strohm (2011) exposed *Osmia bicornis* larvae to experimental treatments of constant and fluctuating temperature, and found that variable temperatures generally caused fewer adverse effects than consistently warmer temperatures. While prepupal weight was lower under warmer stable temperature, this effect was weakened when temperature was variable (even if the modal temperature in the fluctuating temperature treatment was warmer than the control treatment); adult mortality was also lower in the variable temperature treatments in comparison with consistently warm treatment groups (Radmacher and Strohm 2011). Temperature variability caused faster development, but stable warm temperatures also decreased the duration of all developmental phases except the prepupal stage, which was longer (Radmacher and Strohm 2011).

Will higher winter temperatures affect pollinator activity?

Pollinator species have different strategies for surviving winter: a few migrate to overwinter at lower latitudes, while many others enter diapause, a state of suspended development. One of the cues that regulates diapause is temperature, and thus climate change has the potential to alter when pollinators enter and terminate diapause. As winter temperatures rise, some pollinators may become active during a larger part or all of the year, and will require floral resources during times when they may not be available. In some areas, conditions also may become mild enough for new pollinator species to become established in locations that previously were too cold: these species may begin to compete for floral resources with native species.

Dos Santos et al. (2015) predicted that by 2080, 36% of the southern Brazilian endemic *Plebia droryana* Holmberg 1903 (Hymenoptera: Apidae) populations will be active during the winter, but whether there is sufficient pollen and nectar to sustain these populations in winter has yet to be evaluated. Typically, this species enters diapause between 10 and 8°C, but in its current range, minimum winter temperature is expected to rise (from a median of 10.1°C at present to 13.4°C by 2080; Dos Santos et al. 2015). In a laboratory manipulation of winter temperature, *P. droryana* terminated diapause (initiating brood cell building) with even brief exposure (<3 days) to temperatures from 16°C to 22°C (Dos Santos et al. 2016). Winter activity of *Bombus terrestris* has been reported in the UK (i.e., lack of typical diapause as observed in historically cold winters), and Owen et al. (2013) showed that workers were able to tolerate acute cold temperatures typical of a mild winter, without entering diapause. They also found some evidence of adaptation to cold (rapid cold hardening) in workers, and that bees that consumed more pollen had lower supercooling point temperatures (i.e., sufficient resources conferred greater tolerance to cold snaps). In a related study, Owen et al. (2016) showed that *Bombus terrestris dalmatinus* Dalla Torre 1882, an imported pollinator to the UK, has similar cold tolerance to the UK native *Bombus terrestris audax* Harris 1776, and a strong rapid cold hardening capacity. This is evidence that *B. t. dalmatinus* may be able to weather the milder UK winters expected with climate change, perhaps establish in southern areas of the UK, and may potentially be able to displace native *B.t. audax*. In southern Florida, Downing et al. (2016) documented significantly higher cold snap-induced mortality of introduced (tropical) *Centris nitida* Smith 1874 (Hymenoptera: Apidae) bees in comparison to native (subtropical) *C. errans* Fox 1899; the authors suggest that climate change, and the loss of winter cold snaps, may permit the introduced bee species to establish in the wild and compete directly with native species in the near future.

Does overwintering temperature affect pollinator development and emergence?

Common garden experiments with orchard bees (*Osmia lignaria*) have shown limited developmental plasticity in response to manipulations of overwintering temperatures. Increased temperature generally caused earlier emergence, but developmental flexibility depended on the region of origin of a given individual (Pitts-Singer et al. 2014). Pupae from cooler regions (Oregon, Washington) were able to advance their emergence more in response to warmer temperatures than were pupae from California (Pitts-Singer et al. 2014). Schenk et al. (2018a) also showed that *Osmia bicornis* and *O. cornuta* Latreille 1805 had on average earlier emergence and lower body weight at emergence when overwinter temperatures were higher. However, they also found that, on an individual basis, lower body mass was associated with later emergence, which they interpreted as an indication that both body condition and temperature may act as triggers for emergence (Schenk et al. 2018b). CaraDonna et al. (2018) altered the rearing temperature of *Osmia ribifloris* during larval development through adult emergence and found that increased temperatures delayed emergence dates and increased variability in phenology more generally.

These differences in response to overwintering temperatures may be, in part, related to life history characteristics. One comparison of nine bee species showed that species that overwinter as adults or late pupal stage, and emerge earlier in the year (spring bees - *Osmia bicornis*, *O. cornuta*, and *Chelostoma floristomne* Linnaeus 1758; Hymenoptera: Megachilidae), tend to emerge earlier and at lower body weights in response to higher overwintering temperatures (Frund et al. 2013). Species that overwinter as pupae, and emerge later in the year (summer bees), either emerge at similar (*Megachile ligniseca* Kirby 1802, *M. centuncularis* Linnaeus 1758, *M. versicolor* Smith 1844), or greater body weight (*Hylaeus communis* Nylander 1852, Hymenoptera: Colletidae; and *Heriades truncorum* Linnaeus 1758, Hymenoptera: Megachilidae) and have similar (*M. centuncularis*, *H. truncorum*) or delayed (all others) emergence dates when overwintering temperatures are higher, though there was an exception to this general pattern (negative effect of temperature on body mass in summer bee *Coelioxys mandibularis* Nylander 1848, Hymenoptera: Megachilidae; Frund et al 2013).

How does temperature interact with other environmental pressures?

Because pollinators are subjected to multiple environmental stressors simultaneously, the effects of increased temperature may vary depending on other factors. For example, increased temperatures may have greater effects on pollinators when water availability is low. Switanek et al. (2017) found that higher ambient temperatures alone (if there was sufficient precipitation) did not increase mortality in *Apis mellifera* hives – increases in mortality were observed only when temperatures were high and precipitation was low. However, Descamps et al. (2018) manipulated air temperature and water for *Borago* spp. plants, and found that bumble bees visited flowers in the lowest temperature, highest watering treatment twice as often as all other treatment groups. They attributed this to the greater nectar concentration, larger and more numerous flowers, and more viable pollen produced by plants in the lower-temperature, well-watered treatment (Descamps et al. 2018).

Another combination of stressors investigated in several studies is climate change and land-use change. Climate change is hypothesized to select for individuals with smaller size (due to increased development speed), while fragmented landscapes are expected to select for larger size (to travel among more disparate patches). Over 118 years (1902 – 2019), Kelemen and Rehan (2020) found that the body size of carpenter bees (*Ceratina calcarata* Robertson 1900, Hymenoptera: Apidae) decreased with increasing summer temperatures, but there was an additional effect of intensive agricultural expansion (1974 – 2019) that affected the sexes differently – males responded with decreased size but female size increased. If the increased size of females is unsustainable with ongoing climate warming, this species may be extirpated from some areas (Kelemen and Rehan 2020). In their study of five bumble bee species in Europe, Gerard et al. (2021) found that over the past century, landscape fragmentation caused changes in body size – the largest species showed increases in body size with fragmentation, while the smallest species decreased body size with fragmentation – and while temperature increased during the time period studied, it did not influence body size of the species that were evaluated.

Though Papanikolaou et al. (2017b) found that landscape heterogeneity could buffer the effects of variability and increasing temperatures on wild bee abundance and diversity in Germany, other evidence suggests that land use change may be particularly problematic for species with narrow thermal niches. Kuhsel and Bluthgen (2015) showed that pollinator communities that occupy more intensively-used grasslands in Germany have greater thermal resilience, because they are typically inhabited by species with broader thermal niches and complementary thermal optima. Species with narrower thermal tolerance are less likely to persist in highly disturbed, less heterogeneous landscapes (Kuhsel and Bluthgen 2015).

Key findings relevant to Bulkley-Nechako and Fraser-Fort George

1. Temperature is among the primary factors influencing the distributions of pollinator species historically; the research in this area focuses primarily on bees and flies, but there is some evidence that butterflies, moths, and beetles are at least somewhat affected by temperature in similar ways. In some systems, increased temperature has been associated with decreased pollinator species richness. Climate change is likely to make temperature an even more important structuring agent influencing pollinator community composition and biodiversity in the future.
2. Increasing temperature is linked with the movement of species ranges towards the poles and to higher elevations. This means new species may colonize areas where they have not been found in the past, particularly species that are well-adapted for heat (species historically found south of the BNFFG). Native species with larger body size, low heat tolerance, and/or narrow thermal niches are likely to move farther north or to higher elevations, if sufficient floral resources are available in these areas. In the BNFFG, warming winters in particular may be an important promoter of the colonization of new species, and the extirpation or extinction of native species.
3. At a local scale, during the growing season, higher ambient temperatures may promote higher floral visitation rates, but this may not translate to more pollination occurring due to pollinators' need to spend time thermoregulating or foraging for water, which may reduce the time available for pollination-related activities. Temperature is a physiological stressor, and hotter conditions are associated with mortality of pollinators with larger body sizes (i.e., bees from the genera *Bombus* and *Megachilidae*). Heat-induced mortality of bees in general (not just those with large bodies) becomes more likely when a rise in temperature co-occurs with low humidity or a shortage in floral resources; there is some evidence that if sufficient resources are available to pollinators, increased temperatures are not problematic. However, all of the studies that evaluated the physiological effects of heat on pollinators were conducted on bees, so whether these findings also apply to flies, butterflies and/or beetles that provide pollination services remains unclear.
4. Pollinators may also be indirectly affected by temperature, due to effects on plant physiology. In warmer conditions, plants may grow taller, produce nectar and pollen with different chemical characteristics, and/or alter the number and shape of flowers in a way that alters the attractiveness of plants to pollinators, or the nutritional value of floral

resources consumed by pollinators. Whether the attractiveness and value of floral rewards increases or decreases in response to higher ambient temperatures seems to vary depending on the types of plants and pollinators assessed. In systems with resilience in the pollinator community (i.e., high diversity), temperature-induced changes in floral traits and attractiveness may not be a significant concern, but in areas with low pollinator diversity, this could affect pollination success.

Theme 2: Phenology

For insect-pollinated plants to reproduce, pollinators must be present while plants are in bloom. Co-evolution in the timing of major life history events (pollinator emergence and plant flowering) has evolved in many plant-pollinator systems, to ensure that floral resources are available to pollinators when they emerge, and to ensure that plants get pollinated. In many systems temperature, humidity, and/or precipitation are the signals used to trigger emergence and flowering, but the timing of these events is expected to change due to shifts in climate.

In 2002, Fitter and Fitter documented that 16% of 385 British plant species flowered significantly earlier in the 1990s than previously, and on average, these species had advanced by 15 days in the decade, while only 3% of species had later flowering dates in the 90s than previously. Temperature in the month before flowering was the best predictor of flowering, and spring-flowering species were most likely to have altered their flowering date in response to climate signals. Annual and insect-pollinated species were identified as more likely to have advanced their flowering date in comparison to perennials and wind-pollinated plants (Fitter and Fitter 2002). While changes in timing are not problematic *per se*, asynchrony in emergence and flowering can cause a phenological mismatch between plants and pollinators. When this occurs, pollinators may emerge to find there are insufficient resources available, and plants may experience reproductive failure due to a lack of pollination.

In the BNFFG regions, predictions for the future include more frost-free days (average = + 52), increased total growing degree-days (average = + 520), and a longer growing season length in general (+ 38 days) by the 2050s (BC Agriculture & Food Climate Action Initiative 2019). Essentially, the growing season is expected to start earlier and last longer, but plant species may not all respond to climate change by flowering earlier and/or remaining in bloom longer. While warmer temperatures are expected year-round, of particular importance is the potential for an earlier start to the growing season. If plants and pollinators respond differently to this shift in the growing season, there is the potential for phenological mismatch to occur, and a variety of potential consequences for both plants and pollinators alike.

Is the emergence phenology of pollinators changing?

Phenological shifts in insect emergence have been documented, though their magnitude varies; some species may use different strategies (e.g., shift in elevation) to cope with

changes in plant phenology rather than alter their emergence timing. Box et al. (2019), in their comprehensive assessment of climate change in the Arctic over a 47-year period (1971-2017), identified two key biotic changes resulting from intensification of the hydrological cycle: condensed flowering and pollination periods, and phenological mismatch between plant flowering and pollinators. These changes are more evident in the Arctic now, because climate change is proceeding more quickly there, but we may expect that sub-polar and temperate regions will also experience these changes, though perhaps in these regions it will advance more slowly. Duchenne et al. (2020b) found that over the past 60 years in Europe the flight dates of >2000 pollinator species have become on average six days earlier, and flight lengths two days shorter, with northeastern Europe being slightly less affected than elsewhere on the continent. In the northeastern US, the emergence date of 10 wild bee species has advanced by a mean of 10.4 ± 1.3 d over 130 years, most of this since 1970, in tandem with global increases in temperature (Bartomeus et al. 2011). Between 1974 and 2007 flowering phenology of plants on sites near the Rocky Mountain Biological Laboratory has shifted to occur earlier than in the past, while bumble bee workers have not altered their emergence dates, instead shifting to higher elevations while preserving similar emergence timing (Pyke et al. 2016). In the eastern US, Lee et al. (2018) predicted that the introduced pollinator, the Japanese hornfaced bee *Osmia cornifrons* Radoszkowski 1887 (Hymenoptera: Megachilidae), will have significantly advanced emergence by 2100, under both moderate and extreme IPCC climate scenarios (RCP 4.5 and 8.5). Interestingly, the number of days needed for development was predicted to be shorter in the central and northern regions (-0.27 and -0.65 days/decade, respectively) of the eastern US, but longer in the southern region ($+0.57$ days/decade).

Are phenological mismatches occurring?

Evidence of phenological mismatches varies among systems and species; however, areas with greater biodiversity in pollinators and plants may be somewhat buffered from mismatches, in comparison to those with less redundancy in the plant-pollinator network. In some cases, mismatches have been due to climate-induced earlier insect phenology rather than earlier plant phenology, but in other cases flowering has advanced more than insect emergence has.

In the Mediterranean Basin, Gordo and Sanz (2005) evaluated long-term temporal trends of 45 plants, four insects and six migratory insectivorous birds, and found that most spring and summer events for all species had advanced since the mid-1970s, but that insect phenology showed a steeper advance than plant phenology, which they interpreted to indicate a likely decoupling in some plant-insect interactions like those of pollinators and flowers. In evaluating how alpine plants (two species) and animals (five species, three of which are potential pollinators) in Australia have altered their timing over a 30-year period (1977 – 2007), Green (2010) showed that the timing of snowmelt largely dictated when plants flowered, while insects showed much more diverse responses, leading to phenological mismatches. For instance, annual arrival dates of migratory bogong moths (*Agrotis infusa* Boisduval 1832; Lepidoptera: Noctuidae) have been delayed, while March flies (*Scaptia* spp. Walker 1850; Diptera: Tabanidae) and Macleay's swallowtails (*Graphium macleayanum*

Leach 1814; Lepidoptera: Papilionidae) showed no significant change in emergence linked to climate. In a study of 15 wasp and bee species using artificial nests in the Rocky Mountains, Forrest and Thomson (2011) found evidence of phenological mismatch in flowering plants, but noted that this is unlikely to threaten the species they studied because most of them are generalists, and only one (*Osmia lignaria*) was at risk of emerging before any plants had flowered. Pyke et al. (2016) documented decreased synchrony between plants and bumble bees in 2007 in comparison to 1974 in surveys conducted in Colorado, due to earlier flowering of plants in 2007. In subalpine habitat in Colorado, from 1990 to 1995, Thomson (2010) found that snowmelt is the primary trigger for flowering in *Erythronium grandiflorum* Pursh (Liliales: Liliaceae), and that reproductive success has been declining due to pollen limitation, which was linked to increasing asynchrony in the timing of flowering in this plant relative to when bumble bees emerged. Hutchings et al. (2018) modeled flowering of the spider orchid (*Ophrys sphegodes* Mill; Asparagales: Orchidaceae) and emergence of its main pollinator (*Andrena nigroaenea* Kirby 1802; Hymenoptera: Andrenidae) in central England from 1659 – 2014, and showed that while all phenological events had advanced during this time, in tandem with progressively warmer spring weather, all events did not advance at the same pace. To be pollinated successfully, male *A. nigroaenea* must emerge before *O. sphegodes* flowers, and both of these events must occur before female *A. nigroaenea* emerge; at present the emergence dates of males and females are closer together than in the past, and male emergence dates occur earlier relative to flowering than in the past (Hutchings et al. 2018). When springs were exceptionally warm, female emergence could precede flowering and occasionally male emergence as well: with warming trends continuing, the number of years in which *O. sphegodes* experiences complete reproductive failure (due to a lack of pollination) is likely to increase (Hutchings et al. 2018).

Duchenne et al. (2020a) found that interannual temperature variation changes over the past 70 years have reduced on average the mean flight period of 205 wild bee species in Belgium; they linked this to a reduced pollination season length (15 days shorter). In a related study, they (Duchenne et al. 2020b) speculated that the changes in insect emergence and length of flight season may also alter pollination due to changes in the redundancy and complementarity of pollinator assemblages. Adedoja et al. (2020) sampled insect pollinators in four major groups (bees, wasps, beetles and flies) over two spring seasons (2017 and 2018) along an elevation gradient in South Africa, and found that at the lowest elevation, bees peaked later than flowering plants, while at middle elevations, they peaked earlier than flowering plants, suggesting a mismatch is occurring that may increase over time (Adedoja et al. 2020).

What causes phenological mismatches?

While some theoretical modeling has suggested that phenological mismatches may be an adaptive response to change in seasonal patterns (Lindh et al. 2018, Kehrberger and Holzschuh 2019a), more evidence indicates mismatches are the product of a difference in sensitivity between plants and pollinators to seasonal triggers for development (typically temperature). Thus, when spring temperatures are higher and degree-days accumulate faster

than they have historically, flowering may occur at a different time than pollinator emergence and flight does. Numerous studies have been conducted in a variety of systems to evaluate changes in phenology in response to climate, and in many systems plants appear to have advanced their phenology more than pollinators have, but there are some exceptions (see below).

Along an elevational gradient in the Rocky Mountains during three growing seasons, Forrest and Thomson (2011) used reciprocal transplants of artificial nests (trap nests) to show that bees and wasps generally have higher threshold temperatures for termination of diapause and development than plants require to initiate growth in the spring, and thus were less likely to advance their phenology in response to temperature than were plants. In grassland habitats in Germany, Kehrberger and Holzschuh (2019b) observed differences in temperature sensitivity between the first-flowering plant, *Pulsatilla vulgaris* Mill (Ranunculales: Ranunculaceae), and two of its pollinators, *Osmia cornuta* and *Osmia bicornis*, that produced differences in phenology. They found that for each 0.1°C warming, *O. cornuta* males advanced their abundance-weighted mean emergence by 0.4 days, and *P. vulgaris* advanced flowering by a mean of 1.3 days, while temperature had no effect on the emergence of female *O. cornuta* or *O. bicornis* of either sex. However, plants other than *P. vulgaris* did not show a change in flowering time due to temperature. This study demonstrated a stronger effect of temperature on plant flowering than on bee emergence, and authors suggest in future there may be a greater risk of pollinator limitation for early-flowering plants like *P. vulgaris* (Kehrberger and Holzschuh 2019b).

In Japan, Doi et al. (2008) used historical data (1953 – 2004) on flowering dates of four *Prunus* (Rosales: Rosaceae) tree species and the emergence of a butterfly, *Pieris rapae* Linnaeus 1758 (Lepidoptera: Pieridae, a proxy for potential pollinators) to demonstrate that the plants were strongly affected by temperature in the 30 to 40 day period prior to flowering. At the same time, butterflies were weakly affected by temperature during the 15 days prior to emergence. The temperature during plants' sensitive period of plants (i.e., the 30 - 40 days prior to flowering) has increased substantially since 1953, but temperature during the 15-day period prior to emergence of butterflies has not, leading to an increasing mismatch between plants and pollinators in this system (Doi et al. 2008). Along an urbanization gradient in northern France, Fisogni et al. (2020) found that the flowering of 70 plant species advanced in response to urbanization (and associated urban warming), while the flight phenology of pollinators (154 species of bees and hoverflies) did not. This was considered indicative of asymmetric responses of regional plants and insects to warming and an altering of the plant-pollinator network (Fisogni et al. 2020).

In some systems, pollinators have shown larger shifts in phenology in response to climate change than plants have. In California, Olliff-Yang and Mesler (2018) found that temperature was the best predictor of flowering for silky beach pea (*Lathyrus littoralis* Nutt; Fabales: Fabaceae) as well as the emergence of its main pollinator, the ground-nesting solitary silver bee (*Habropoda miserabilis* Cresson 1879; Hymenoptera: Apidae). However, they also found that bee nesting time was more sensitive to temperature than pea flowering, which means nesting will likely advance more quickly than flowering as temperature increases. Though at

present these species have high phenological overlap, the authors indicate that this difference in response to temperature may produce a mismatch as climate change progresses (Olliff-Yang and Mesler 2018).

There also may be substantial differences among species in their response to altered seasonal changes in temperature linked to differing life-history characteristics. There is some evidence that species that overwinter as adults may be more sensitive to changes in phenology than those that overwinter as prepupae. Slominski et al. (2019) used an experiment to evaluate how fall and spring temperature, fall duration, and the timing of spring onset affected eight cavity-nesting solitary bee species that overwinter either as adults or prepupae. Species that overwinter as adults (*Osmia* spp.) responded to the timing of spring onset, and when spring occurred earlier, pre-emergence mortality and weight-loss were increased, and longevity was decreased. Species that overwinter as prepupae (*Megachile* spp.) responded to spring temperature, and warmer springs were associated with less weight loss and increased longevity.

For some species or ecosystems, temperature sensitivity may not be the only contributor to asynchrony between plants and pollinators; aridity has been linked to asynchrony with plants for butterfly as well as bee emergence. On the Iberian Peninsula, Donoso et al. (2016) documented the interactions of 17 flowering plant with 12 butterfly species over a 17-year period (1996 – 2012) in a coastal wetland, and found that asynchrony between butterflies and host plants was greatest in the spring, and that dry winters exacerbate spring asynchrony. Dry springs promoted asynchrony between summer-flowering plants and butterflies that feed on them, and while the authors did not find a climatic trend over the period of the study, both specialist and generalist species of butterflies were similarly likely to experience an asynchrony with plants (Donoso et al. 2016). In ecosystems with Mediterranean characteristics, aridity might be more important than temperature in promoting asynchrony between plants and pollinators (Donoso et al. 2016). Boggs and Inouye (2012) found that early snow melt reduces the fecundity of butterfly *Speyeria mormonia* Boisduval 1869 (Lepidoptera: Nymphalidae) by limiting floral resources – this reduced fecundity causes smaller population sizes in subsequent years. Stemkovski et al. (2020) showed that, for 67 bee species in the Colorado Rocky Mountains, bee emergence generally advanced with snowmelt timing over a nine-year period, but subsequent life-history events were better explained by traits like overwintering stage and nest location. In comparison with plants in this system, bee phenology appears less sensitive to climate change, which the authors suggest favours the development of phenological mismatch in the future (Stemkovski et al. 2020). Over a 10 – 14 year period, Kudo and Ida (2013) found that snowmelt time and spring air temperature influenced the onset of flowering in spring ephemeral *Corydalis ambigua* Cham. & Schltldl (Ranunculales: Papaveraceae) and the first detection of overwintered queen bumble bees. In years when spring came earlier, *C. ambigua* seed production was lower because flowering occurred prior to bee emergence (Kudo and Ida 2013). The authors identified phenological mismatch as a significant factor limiting reproductive success in spring ephemeral plants in particular (Kudo and Ida 2013). In a follow-up study, Kudo and Cooper (2019) used historical records and a snow removal experiment to show that earlier snowmelt caused earlier flowering of *C. ambigua* by increasing soil surface temperature, and

the resulting mismatch with bumble bee emergence reduced seed production due to lower pollination success.

Some studies show that the phenology of plants that require insect pollination may respond differently to climate change than wind-pollinated plants. Rafferty and Ives (2011) demonstrated this by manipulating the flowering phenology of 14 perennial plant species, six of which have shown a shift toward earlier flowering over the past 70 years. Most of the species with a history of advancing phenology (five of six) had more pollinator visits when flowering was artificially advanced, while most of the species (seven of eight) that had no history of advanced flowering received fewer pollinator visits when their flowering was advanced. For most of the species studied, there was little evidence of phenological mismatch: species constrained by pollinators did not advance their flowering, while species not constrained by pollinators did (Rafferty and Ives 2011). Ulrich et al. (2020) manipulated the presence of insects for an artificial grassland community, and found that plants flowered earlier when insects were more abundant, and that plant communities were less diverse when insects were absent.

What are the consequences of phenological mismatch?

In theory, phenological mismatches could lead to reduced pollination of plants and insufficient resources for pollinators; theoretical models also predict differences in pollinator population sizes. Memmott et al. (2007) simulated the consequences of the phenological shifts that can be expected with a doubling of atmospheric CO₂ in a network of interactions between 1420 pollinator and 429 plant species. They found that earlier flowering is likely to make the quantity of floral resources available to 17 – 50% of all pollinator species to decline. If insect emergence dates do not change, no food plants will be available during as much as half of the current typical activity period of many species. Pollinators' diet breadth is also expected to be narrower due to a reduction in overlap between plants and pollinators (Memmott et al. 2007).

On sites in Greenland, Hoyer et al. (2013) demonstrated that from 1996 to 2009 the flowering season has shortened, which they linked to declines in flower visitor abundance over the same period. In a two-year study conducted in a Mediterranean habitat, Flores et al. (2019) found that when the flowering period was shorter, *Apis mellifera* hives gained less weight, had fewer occupants, and had smaller reserves of both honey and pollen, indicating that bees were food-stressed. Petandjou et al. (2014) studied a plant-pollinator network comprised of 132 plant species and 665 pollinating insect species in a Mediterranean community, and found that when the onset of flowering occurred earlier, pollinator richness was lower. In this system, earlier flowering was also associated with longer duration of flowering, but it is unclear whether this could compensate for flowering before many pollinator species had emerged (Petandjou et al. 2014). However, in manipulating the flowering onset of *Mertensia ciliata* (James ex Torr.) G. Don (Boraginales: Boraginaceae), Gallagher and Campbell (2020) found that seed set did not differ among plants blooming at different times – over the course

of the season, the number of effective pollinators (bumble bees) increased, which compensated for reduced overall pollinator visitation rates as the season advanced.

Farzan and Yang (2018) followed individually marked blue orchard bee (*Osmia lignaria*) females from cohorts with experimentally manipulated emergence dates to show that the earliest cohort had greater survival to nesting (20.4% higher than the second) but reproductive output peaked in the second cohort. Overall, females manipulated to emerge earlier had greater fitness than those emerging at historical norms or later in the season. In supplementary observational data, they found that earlier emergence was typically associated with conditions of increased resource availability and reduced parasitism risk, indicating that selection may be occurring for earlier emergence in this population. Similarly, Lee et al. (2018) noted that in areas of the eastern US where the introduced pollinator, the Japanese hornfaced bee, *Osmia cornifrons* is expected to require more days for development in the future (by the year 2100), populations may be exposed to pests and parasites for longer than in the past – which could have deleterious consequences. Nurnberger et al. (2019) used reciprocal translocations to assess how phenological shifts may affect *Apis mellifera*, and found that colonies with delayed phenology had smaller worker populations early in the spring and stored less honey over the course of the season, but also had lower parasite loads (*Varroa destructor* Anderson & Trueman 2000; Arachnida: Mesostigmata). Because colonies with typical phenology had three times as many varroa mites, and worker population growth was negatively correlated with mite loads during the summer, colonies with earlier emergence and developmental phenology may benefit from greater floral resources but may also be subject to greater parasite loads, which constrain their productivity (Nurnberger et al. 2019).

Some evidence against catastrophic consequences of phenological mismatch comes from Britain, where Balfour et al. (2018) found that extinct pollinators were more likely to be late-summer flying bee species, which the authors attributed to declines in late-summer resources (herbaceous plants), linked to land-use change, rather than climate *per se*. In fact, the authors suggested that trees and mass-flowering crops (shrubs), which typically bloom earlier in the year, and seasonal variation in intraspecific competition, may have buffered spring flying pollinators from climate change in the past several decades. Korosi et al. (2018) manipulated the phenology of apple trees, and found that though there was no effect on the overall abundance of pollinators, more wild bees and hoverflies, and fewer honey bees (*A. mellifera*), visited trees that flowered earlier. This indicates that, in some systems, the advancement of flowering may provide resources to wild pollinators that are resource-limited early in the season.

Can pollinators and plants avoid phenological mismatches?

There are some systems that show similar changes in the timing of plants and their pollinators, or flexibility in the response of some plants or pollinators to permit successful pollination and/or timely emergence. In a transplantation experiment, Dai et al. (2017) found that lousewort (*Pedicularis siphonantha* Don; Lamiales: Orobanchaceae) in China was able

to shift phenology in response to local conditions: plants moved to lower elevations flowered earlier, had shorter flowering periods, smaller flowers and lower pollen production, but received more pollinator visits and produced more seeds. Sevenello et al. (2020) found that flowering of two spring wildflowers (*Anemone* spp. L.: Ranunculales: Ranunculaceae; and *Trillium grandiflorum* Schreb: Fabales: Fabaceae) and emergence of ground-nesting bees (species in the genera *Nomada* Scopoli 1770 [Hymenoptera: Apidae], *Andrena* Fabricius 1775 [Hymenoptera: Andrenidae], and *Lasioglossum* Curtis 1833 [Hymenoptera: Halictidae]) in a hardwood forest understory responded similarly to changes in snowmelt date and spring temperatures over a period of six years. However, because *Trillium* flowering showed greater sensitivity to photoperiod (which will not be affected by climate change) than temperature, the authors note that differences among taxa in their phenological responses may produce diverging temporal shifts in the future (Sevenello et al. 2020). In Japan, over the past 40 years, evidence of changes in the emergence dates of 14 insect species is mixed: some species and sites showed a trend of later emergence, while others emerged earlier in warmer conditions; however, the shifts in emergence timing of insects were generally less than those found for plants in these ecosystems, suggesting a phenological mismatch is likely, and may worsen in the future (Ellwood et al. 2012). Monitoring *Bombus* in an alpine habitat in Japan, Kudo (2014) found that when spring came early, colony development was slowed: queens emerged ten days before the first flowering date, but workers emerged later than expected; additionally, there was an earlier conclusion to the flowering season (two weeks) due to earlier snowmelt. As a result, most of the *Bombus* spp. monitored experienced a phenological mismatch, being most abundant after flowering peak had occurred, but one common species, *B. hypocrita sapporoensis* Cockerell 1911, showed flexibility in responding to changes in flowering date.

There is evidence that some communities may escape phenological mismatch by having sufficient pollinator biodiversity. In the northeastern US, Bartomeus et al. (2013b) found that over a 46-year period, peak apple bloom and bee activity were highly synchronous, largely because bee species differed in their responses to warming, and produced a trend of stable abundance in pollinators over time. In this system, high biodiversity in native pollinators ensured sufficient provision of pollination services despite changes in the timing of flowering (Bartomeus et al. 2013b). In a related study, Bartomeus et al. (2011) found that the shift in emergence dates of 10 northeastern US bee species is similar to what has been documented in the flowering dates of plants, so for some generalist bee species in this region emergence dates may in fact be keeping pace with alterations in flowering phenology. If sufficient diversity is present, phenological shifts may alter which pollinators are most important, without compromising pollination success, as Wall et al. (2003) found when evaluating an endangered plant, *Clematis socialis* Kral (Ranunculales: Ranunculaceae). For this species, when spring blooming was later *Anthophora ursina* Cresson 1869 (Hymenoptera: Apidae) provided more pollination than *Bombus pennsylvanicus* De Geer 1773 (Hymenoptera: Apidae), while when spring blooming occurred earlier, the opposite occurred (Wall et al. 2003).

Key findings relevant to Bulkley-Nechako and Fraser Fort George

1. The earlier onset of spring is likely to lead to earlier flowering of plants. In many systems, pollinators appear to be less sensitive than plants are to temperature increases that signal the onset of spring. This difference in temperature sensitivity may cause plants to bloom before pollinators have emerged from overwintering. Crops that bloom early in the spring are most at risk of pollination deficits due to phenological mismatch.
2. Pollinators' ability to adapt to shifts in plant phenology is likely to be species-specific, and may depend in part on life-history characteristics (particularly whether a species overwinters as a pupa or prepupa). Some evidence indicates that bee species may be more likely to experience phenological mismatches with plants than hoverflies. More generally, pollinator species with limited flexibility in phenology are likely at greater risk of emerging when insufficient floral resources are available.
3. While the growing season will be longer than it has been in the past, there is little evidence from other studies to suggest pollinator flight seasons will also become longer. As a result, pollination services may not be available for the entirety of the growing season. Pollination services will likely be most limited at the start and end of the season (i.e., early-spring and late-summer/fall).
4. Areas with high pollinator diversity may be buffered against the effects of shifts in the growing season, as those pollinator species that are able to shift their emergence phenology may provide sufficient pollination services to compensate for species that do not shift their phenology.

Theme 3: Precipitation

In the BNFFG, climate projections predict more precipitation in all seasons other than summer, but due to warmer temperatures more will likely occur as rain, and less as snow, than in the past (BC Agriculture & Food Climate Action Initiative 2019). It is also likely that precipitation will occur in shorter, more intense episodes than in the past (BC Agriculture & Food Climate Action Initiative 2019). While snowmelt dates and the ensuing effects on spring freshet are also likely to differ, this will vary among watersheds, and better understanding requires more specialized hydrological modeling (BC Agriculture & Food Climate Action Initiative 2019). There are two ways in which changes in precipitation during the growing season may affect pollinators. First, during rain events pollinator visitation is likely to drop sharply, but this represents a relatively transient effect on activity, and may not have enormous effects on overall pollination. Second, plants may respond to more precipitation by altering investment in floral rewards, which may affect the success of plant-pollinator interactions even after the rain is over (i.e., indirect effects).

How does precipitation affect pollination?

During the growing season, rates of pollination visitation may transiently decline during rain events, but in most systems both plants that are water-stressed and those that receive too much water seem to be less preferred by pollinators and often have reduced pollination success (as indicated by seed set). Sanderson et al. (2015) generated time-series models of pollinator activity from the exit and return records from 36 captive *B. terrestris* colonies in northern England over a 78-day period, finding that foragers were more active when air temperature and solar elevation increased, and rainfall, humidity, and wind-speed negatively affected forager activity. In an assessment of pollination of native lowbush blueberry (*Vaccinium angustifolium* Aiton; Ericales: Ericaceae) in Maine from 1990 – 2012, Drummond et al. (2017) found that an increase in precipitation has reduced the number of foraging days for bees while plants are in bloom, and they suggest that providing supplemental forage for pollinators may be needed to retain biodiversity. Gallagher and Campbell (2017) used an experiment to show that while pollinator visitation (largely by bumble bees) of the plant *Mertensia ciliata* (borage family) peaked at intermediate water levels, seed set and other floral traits increased with water availability. In an experiment on roseleaf bramble (*Rubus rosifolius*; a relative of raspberries) in Brazil, Antiqueira et al. (2020) found that increasing precipitation caused a decrease in seed set due to reductions in the number and richness of pollinators visiting flowers, though the composition of species visiting flowers was unaffected. In a mesocosm study conducted over three seasons in northern Germany, Walter (2020) documented lower pollinator visits and reduced pollinator species richness at plants receiving too much (wet or temporarily flooded) or too little (dry) water relative to controls (plants receiving intermediate quantities of water), but dry plots had the lowest number of visits from pollinators in addition to the lowest pollinator richness overall. One species, *Trifolium pratense* L. (red clover; Fabales: Fabaceae), responded to permanently wet conditions by decreasing nectar sugar content, and this study found that the seed and flower weight of this species also declined in both flooded and dry plots, relative to controls, likely because of decreased attractiveness to pollinators (Walter 2020). In Panama, Peters and Carroll (2012) found that although coffee plants (*Coffea arabica* L.; Gentianales: Rubiaceae) require precipitation to trigger flowering, in a year when precipitation occurred in greater volumes and later than expected, the abundance and richness of bees visiting flowers were lower, as was pollination success.

There may also be significant differences among ecosystems in the effect of water availability on pollinators. For example, in a semi-arid region (Mohave floodplain), Andersen and Nelson (2013) found that, over a two-year period (2010 and 2011), richness and abundance of floral visitors to a riparian plant, *Tamarix* sp. (saltcedar; Caryophyllales: Tamaraceae) differed but this was unrelated to depth to groundwater (an indicator of water stress). In contrast, in a tropical zone (Bangalore, India), Mukherjee et al. (2019) found that lower rainfall was associated with general decreases in the abundance of four main pollinator species, and the yield of all pollinator-dependent crops over six years. They also found that the species responsible for most pollination differed depending on rainfall – in non-drought years, *Trigona* spp. bees were the most important influence on fruit-to-flower ratio (an

indicator of pollination services), but during drought periods *Apis dorsata* Fabricius 1793 (Hymenoptera: Apidae) compensated for the decline of *Trigona* in two-thirds of farmlands (Mukherjee et al. 2019). Minckley et al. (2013) studied a bee-plant community in the Chihuahuan Desert, and found that drought did not affect generalist bee species or bee specialists that relied on drought-intolerant plant species; only bee specialists that rely on drought-sensitive plants were absent during a drought year. Under conditions of extended drought, the authors predicted that bee communities will become progressively more species-poor and dominated by specialist species reliant on drought-intolerant plants, as are currently found in arid desert regions of North America (Minckley et al. 2013).

Reduced precipitation may limit pollination due to changes in rewards for pollinators. Glenny et al. (2018) and Burkle and Runyon (2016) both found that drought had negative effects on visual floral traits (flower size and display) of four forb species (three native and one invasive: *Campanula rotundifolia* L. [Asterales: Campanulaceae], *Heterotheca villosa* Pursh [Asterales: Asteraceae], *Phacelia hastata* Dougl. ex Lehm. [Boraginales: Boraginaceae], and *Potentilla recta* L. [Rosales: Rosaceae]). However, while Glenny et al. (2018) found that drought was linked to reduced pollinator visitation rates to one species (*P. recta*) and no effect on the rest, Burkle and Runyon (2016) found that drought caused decreased visits to a different plant species (*C. rotundifolia*), increased visits to another (*P. hastata*), and no effect of drought on pollinator visitation to the remaining two species. When Rankin et al. (2010) reduced water availability for tomcat clover, *Trifolium willdenovii* Spreng. (Fabales: Fabaceae) they found that it produced nectar of poorer quality and in lesser quantities, and pollen with lower protein content in response. Honey bees and bumblebees fed diets composed of this poorer-quality nectar and pollen had lower survival and productivity than those fed control diets that contained optimum sugar:glucose:fructose ratios and higher protein content (Rankin et al. 2010).

Because of their frequent requirement for aquatic habitats to produce larvae, Diptera may be disproportionately affected by drier conditions. In a two-year study that quantified floral visitation in two tall-grass prairie communities that differed in water availability Robson (2008) found that Diptera were the most abundant floral visitors, followed by Hymenoptera. Seasonal changes in visitation largely tracked temperature, but more Diptera were observed at the site closer to aquatic habitat.

Key findings relevant to Bulkley-Nechako Fraser-Fort George

1. Precipitation generally causes a decrease in floral visitation by pollinators. In all seasons other than summer, rainfall is projected to increase in the BNFFG, which may alter pollination services by reducing the amount of time that pollinators are active while plants are blooming, particularly in the spring. Some research suggests that providing additional floral resources (planting in field margins, etc.) may help preserve wild pollinator biodiversity when periods of inclement weather become more frequent.
2. Many pollinators appear to prefer plants that are neither over-watered nor water-stressed, likely due to the effects of inappropriate watering schedules on plant energy allocation,

which alters floral rewards. In the summer, given the projected increase in temperature and similar or lower quantities of precipitation, it is possible that plants may become water-stressed, which could lead to reduced floral rewards and lower pollination success due to reduced visitation by pollinators. Additionally, pollinators that consume lower-quality pollen and nectar may experience increased mortality and decreased productivity, which could have consequences for populations and ecosystems more broadly.

3. Pollinators that require bodies of water to complete their life cycles (primarily some flies) may be more vulnerable to drought than to increased precipitation. For crops or regions where aquatic flies provide a significant portion of pollination services, it may be prudent to limit the use of water from wetlands and ephemeral sources in the spring and summer.

Theme 4. Extreme events

In the BNFFG, as in many other regions, is expected to experience more extreme weather events with the advance of climate change. Current predictions are for twice as many days >25°C (at present, the average is nine days per year), and seven times as many days >30°C (at present, 1 day per year is normal) by 2050 (BC Agriculture & Food Climate Action Initiative 2019). Additionally, this increasingly dynamic weather may be associated with high winds as intense high-pressure weather systems move through the region. There are a few studies that deal specifically with how heat waves affect pollinators, most of them conducted on bees, and only one study that examined how wind alters pollination services. Generally, extreme weather events are likely to have negative consequences for pollinators and pollination, either by directly causing mortality, or indirectly, by preventing pollinators from foraging.

How do heat waves affect pollinators and pollination?

There is some evidence that heat waves may cause some plants to become more dependent on pollination services for maximal seed sets, but that these events may also reduce pollinator activity and have adverse physiological consequences for pollinators. Following acute heat stress (five days of elevated ambient temperatures), faba bean (*Vicia faba*) yields decreased significantly unless insect pollinators were present (Bishop et al. 2016), and outcrossing (insect pollination) increased following heat stress treatments (Bishop et al. 2017).

Experimental manipulations of honey bee colonies by Bordier et al. (2017) showed that during simulated heat waves, foraging activity increased by ~ 70%, but this was largely due to an increase in water foragers – pollen and nectar foraging continued at similar levels. This strategy resulted in increased recruitment of colony members and physiological changes in individuals, both signals of a stress response, which could have later consequences (Bordier et al. 2017). Among bumble bees, manipulations of temperature have shown that investment in offspring generally declines when colonies are subjected to simulated heat waves (Vanderplanck et al. 2019). Additionally, smaller colonies are generally more sensitive to heat stress than larger colonies, but better-quality nutrition may buffer the effects of increased

temperatures (Vanderplanck et al. 2019; Zaragoza-Trello et al. 2020). For Australian temperate-zone fig wasp *Pleistodontes imperialis*, survival to emergence and adult longevity decline significantly when individuals are exposed to temperatures above 35°C (5°C above current daily mean maximum) for as little as three hours (Sutton et al. 2018), showing that in some cases, even short extreme heat events may dramatically affect some species.

What other extreme weather events may affect pollinators?

In California, Brittain et al. (2013) found height stratification (spatial complementarity) in pollinator communities attending almond trees, with honey bees preferring higher sections of trees than wild pollinators, which attended flowers lower to the ground. However, during periods of high winds, orchards with low pollinator diversity (i.e., only honey bees) had substantially lower floral visitation rates, indicating that wild pollinators were less affected by inclement conditions than were honey bees (Brittain et al. 2013). In this system, when conditions are windy, pollination services may be buffered by biodiversity in pollinators.

Key findings relevant to Bulkley-Nechako Fraser-Fort George

1. Periods of exceptionally hot temperatures are likely to occur more often in the future, and during these periods pollination services will decline, as pollinators mount stress responses, engage in alternative activities (especially water foraging), or die.
2. High winds may not be problematic for wild pollinators, so areas with high native pollinator biodiversity are unlikely to experience reduced pollination in windy conditions, unlike sites that rely on domesticated honey bees for pollination. As a consequence, areas with high pollinator biodiversity may be less affected than sites that rely more on honey bees to carry out pollination.

Theme 5. Other co-occurring changes

While temperature and precipitation are expected to be affected by climate change, numerous other abiotic and biotic factors will also differ from historical norms. The primary driver of climate change is greenhouse gases, and carbon dioxide (CO₂) in particular is expected to continue to rise. Because CO₂ is essential to plant metabolism, some have suggested that increased levels may spur increased vegetative growth, and potentially investment in floral resources, providing a net benefit to pollinators. One biotic factor that may also be affected by climate change is the success and prevalence of parasites of pollinators – several studies suggest that climate change may facilitate the spread of parasites to new areas, or provide the conditions that favour increases in parasite population sizes.

How will increased CO₂ affect pollination?

For some plants, vegetative growth may increase with the rise in ambient carbon dioxide (CO₂) expected under climate change, but how this might affect pollinators remains unclear. In a study that manipulated ambient CO₂ levels for faba beans (*Vicia faba*), Osborne et al. (1997) showed that increased CO₂ caused plants to produce more flowers, and to flower for longer than those grown at ambient. However, the volume of nectar produced per flower did not differ, so plants grown at higher CO₂ would likely provide more resources to pollinators. Glenny et al. (2018) also found that visual and olfactory floral traits of four forb species, as well as pollinator visitation, were increased when CO₂ was elevated. An experiment by Amanina et al. (2016) tested how an oil palm pollinator, the weevil *Elaeidobius kamerunicus* Faust 1878 (Coleoptera: Curculionidae), might respond to increased CO₂, and found that ambient CO₂ did not affect the number of weevils emerging from cuttings taken from palms that were incubated under varying CO₂ regimens (400 and 800 ppm). In an experiment, Hoover et al. (2012) showed that pumpkins (*Curcubita maxima*) exposed to increased CO₂ (700 ppm) had delayed flowering, and produced smaller flowers with less nectar, and the longevity of bumble bee (*Bombus terrestris*) workers consuming this nectar was lower than that of conspecifics consuming nectar produced by plants kept at lower ambient CO₂ levels (360 ppm). The authors suggested that the effect on longevity was likely caused by changes in the sugar and amino acid composition of the nectar produced by pumpkins grown in increased CO₂ (Hoover et al. 2012).

Will climate change affect parasites?

Climate change may also indirectly alter pollinator abundance and diversity by influencing parasite populations. Cornelissen et al. (2019) predicted substantial risk of invasion by small hive beetle (*Aethina tumida* Murray 1867; Coleoptera: Nitulidae), an endemic parasite of honey bee colonies in sub-Saharan Africa, under current and future climatic conditions, particularly in temperate regions of the Northern hemisphere. In Tanzania, Giliba et al. (2020) found that areas currently at high risk of *Varroa destructor* (a parasitic mite of honey bees) invasion are predicted to have reduced risk in the future due to climate change, while those currently at moderate and marginal risk of invasion may have increased risk by 2085, indicating that expansion of *V. destructor* to new areas in future is likely. They identified mean diurnal range in temperature, mean temperature and mean annual rainfall as the most important climatic determinants of risk of *V. destructor* (Giliba et al. 2020). Forrest and Chisholm (2017) found that for a pollen specialist bee, *Osmia iridis*, in subalpine habitats, temperature increased rates of brood parasitism (by the wasp *Sapyga pumila*), which indirectly decreased the reproductive output of *O. iridis*. Though temperature directly benefited *O. iridis* (see Theme 2, above), the benefits were canceled out by the indirect costs due to increased brood parasitism (Forrest and Chisholm 2017). Over 22 years in the Arctic (Greenland), Kankaanpää et al. (2020) documented a shift in larval host use of parasitoids – on sites with more summer warming than winter warming, parasitoids of Lepidoptera (herbivore and pollinator) dominated, while in areas with more winter warming, parasitoids

of dipterans (pollinator) were more numerous. For crops with effective dipteran pollinators, this could have substantial consequences for pollinator abundance.

Key findings relevant to Bulkey-Nechako Fraser-Fort George

1. There are unlikely to be direct effects of increased CO₂ on pollinators, but it is possible that changes in the metabolism of host plants may affect resource availability for pollinators. However, the reviewed studies show that plants are not all expected to respond in the same way to increased CO₂, and thus the consequences are likely to differ depending on the species of crop plant pollinators are interacting with. While pumpkins and other related *Curcubita* (squashes and allies) may flower later and produce lower-quality nectar, legumes (like faba beans) may produce more flowers and bloom for longer periods of time, providing more resources.
2. Pests and parasites of pollinators are likely to become more numerous in the BNFFG as climate change continues, because historically cold winters may have functioned to control their spread. As winter temperatures rise, it is likely that more hive parasites and other natural enemies of pollinators will persist and spread.

Synthesis: potential targets for adaptation or mitigation

Reliance on a single species for pollination is a poor strategy, particularly while climate change causes increasingly unpredictable conditions during the growing season: it makes large-scale pollination failures far more likely (Berecha et al. 2013; Nielsen et al. 2017). Greater pollinator biodiversity results in more reliable pollination, even when conditions are poor, because more diverse communities have greater redundancy in functional roles among pollinator species. When numerous species are capable of pollinating a given plant, a ‘bad year’ for one pollinator species may not disproportionately affect pollination because other species in the community can compensate (e.g., Mukherjee et al. 2019). Additionally, numerous studies suggest that preservation of pollinator biodiversity enables greater resilience in pollination services because diverse pollinator communities are composed of species that have complementary emergence dates, spatial distribution, and foraging habits (Bartomeus et al. 2011, Brittain et al. 2013, Rader et al. 2013, Straka and Starzomski 2015, Drummond et al. 2017). This complementarity also contributes to resilience in pollination services, because these species will not all be affected in the same way by a shift in conditions, due to their differing life history characteristics and ecology (e.g., Brittain et al. 2013). Biodiverse pollinator communities are more robust to climate change, and the literature suggests three general areas of focus that are likely to promote pollinator biodiversity, detailed below.

Target 1: Maintaining non-crop floral resources for pollinators

One way to preserve greater biodiversity in pollinators is to maintain diverse plant assemblages, and ensure floral resources are present for the entirety of the growing season. Numerous studies identify resource diversity as a key contributor to pollinator diversity. The number of pollinator species in flight is correlated with the richness of flowering insect-pollinated plants in numerous ecosystems (e.g., multi-species studies in Britain, Balfour et al. 2018, and South Africa, Adedaja et al. 2018; and global and continental study on the genus Colletinae, Bystriakova et al. 2018). In a simulation study, Memmott et al. (2010) evaluated whether planting nectar-rich flower mixtures in field margins could be used as supplemental forage to buffer bumble bee populations from the changes in resource phenology expected under climate change. They found that a non-crop wildflower mix could provide substantial protection from days without food, and that providing supplemental floral resources was most important at the start and end of the growing season, when the availability of other food was lowest (Memmott et al. 2010).

Ensuring sufficient resources are available to pollinators may also buffer the effects of temperature. Zaragoza-Trello et al. (2020) found that if sufficient resources are available, increases in the mean temperature (1.62°C) of *Bombus terrestris* hives increases the number of individuals in all castes of a colony, but when resources are insufficient, negative effects are observed. This is because when resources are available, colonies are able to develop faster and have sufficient resources to make the wax and silk needed to protect offspring and cool the colony (Zaragoza-Trello et al. 2020).

Target 2: Monitoring intensive use of domesticated pollinators

There is some evidence that domesticated pollinators (honey bees, *Apis mellifera*) can outcompete wild pollinators in agricultural settings, which may have consequences for wild pollinator abundance and diversity. Given that honey bees may be more sensitive to temperature than native pollinators in some regions, relying on domesticated pollinators may not be a climate change-resilient strategy. In California, Thomson (2016) found that, over a 15-year period (1999 – 2014), the abundance of *A. mellifera* was inversely correlated with the abundance of *Bombus* spp. in field surveys, *Bombus* spp. declines were associated with reduced floral resources and occurred in years with reduced spring rainfall or drought. Evidence of reduced diet overlap in years when *Bombus* spp. were less abundant suggests that these species compete for resources, indicating that areas with significant populations of domesticated pollinators may be at greater risk for loss of native pollinators (Thomson 2016). In a study conducted in Norway, honey bees out-competed bumble bees on raspberry (*Rubus idaeus* L.; Rosales: Rosaecae) farms – honey bees were 97% of floral visitors to farms, while wild pollinators constituted 55% of floral visitors in surrounding wildflower communities (Nielsen et al. 2017). However, honey bees were much more sensitive to temperature than bumble bees (temperature explained 46% of the variation in honey bee visits, but only 2.2% of bumble bee visits), and honey bee visit rates peaked at 24.1°C, while bumble bees showed

a positive linear response to temperature (Nielsen et al. 2017). Raspberry crops in this the BNFFG may be vulnerable, given their reliance on a single pollinator species (honey bees) that seems more sensitive to temperature than wild pollinators (bumble bees). Nielsen et al. (2017) did not evaluate whether, in the absence of honey bees, wild pollinators could provide sufficient pollination of raspberries. Similarly, in southwest Ethiopia, Berecha et al. (2015) found that increasing forest management intensity and fragmentation was associated with greater relative abundance of honey bees on coffee plantations, which they linked to the introduction of bee hives in the most intensively managed forests. However, increased management and fragmentation were also associated with declining taxonomic richness of flower-visiting insects, so that more intensively-managed farms have less resilience in pollination services, due to their reliance on a single species for pollination (Berecha et al. 2015).

Target 3: Maintaining landscape heterogeneity and habitat connectivity

The retention of semi-natural habitat fragments within areas used for agriculture, and the maintenance of connectivity among fragments are likely to aid in preserving pollinator biodiversity and abundance and ensuring pollination services. Gutierrez-Arellano and Mulligan (2020) estimated that 80% of tropical protected areas contribute to crop pollination in adjacent regions. They attributed this to the provision of habitat for pollinators by protected areas and noted in particular the importance of small protected areas located close to crops and other cultivated areas as providing high pollination service per unit area while simultaneously being under greater anthropogenic pressure than larger, more remote protected areas (Gutierrez-Arellano and Mulligan 2020). Vickruck et al. (2019) demonstrated the importance of wetland remnants in the Prairie Pothole region by sampling bees at varying distances from remnants in agricultural areas, and found that native bee abundance and diversity declined as distance from wetlands increased. In this system, retention of semi-natural habitat fragments, specifically wetland remnants, may be important to ensuring sufficient pollination services (Vickruck et al. 2019). In terms of adaptation within the agricultural sector, Polce et al. (2014) suggested selecting crops and varieties that can be cultivated in areas with higher predicted pollination availability in future, and reclaiming habitat to support wild pollinator populations, as potential mitigation strategies for changes in pollinator distribution.

Retaining semi-natural areas within agricultural areas may be important to mitigating temperature effects on the richness and abundance of wild bees (Papanikolaou et al. 2017a). Papanikolaou et al. (2017b) found evidence that landscape heterogeneity could buffer against the deleterious effects of temperature on wild bee abundance and diversity in Germany, but not of variability in precipitation. They identified the maintenance of landscape heterogeneity as an important contributor but not a complete solution to preserving wild bee populations and pollination services (Papanikolaou et al. 2017b). In a related study the same research group found that, after correcting for phenology, bee species richness declined as temperature

rose, but that greater proportions of semi-natural habitats on the landscape decreased this negative effect of temperature (Papanikolaou et al. 2017a).

In their study of native stingless bee (*Melipona subnitida*) in Brazil Giannini et al. (2017) identify maintenance of habitat connectivity, via the cultivation of particular pollen- and nectar-provisioning plants, as important to retaining sufficient gene flow to protect that species from declines. The retention of habitat heterogeneity is likely to have positive effects on broad-scale phylogenetic diversity as well: in an experiment that simulated increased habitat fragmentation, bees visited plants (*Sinapis arvensis*) located farther from semi-natural habitat less often, while flies visited plants at similar rates regardless of how far from semi-natural habitat they were (Parsche et al. 2011). The positive effects of habitat heterogeneity on biodiversity ensure there is redundancy in pollinators (more than one, but ideally many, species to pollinate plants at any given time), which will result in greater resilience in pollination services.

Conclusions

There are two key caveats that become apparent after reviewing studies in this topic area: first, many responses to climate change are likely to be species-specific; and second, that numerous abiotic and biotic pressures are predicted to vary with climate change (temperature, humidity/precipitation, atmospheric CO₂ concentration, and nitrogen deposition, at minimum) and their effects on pollinators will not be simply additive. It is evident that many pollinators are in decline or have shifted their range relative to the past, and that declines are worse in areas with greater anthropogenic influence due to urbanization or agriculture. It is clear that climate change interacts with land use change to cause these shifts and declines, and thus making decisions regarding land use with the provision of pollination services in mind may be one way to mitigate the effects of climate change on pollinators.

Species differ substantially in their resource needs, which may explain why even closely related species are expected to respond differently to climate change. However, there seems to be some evidence that species with larger body sizes and smaller geographic ranges are likely to be disproportionately affected by climate change. In the BNFFG, we may expect to see species with historically more southern distributions to become more common, so long as there are appropriate host plants available, and we may see species with northern or higher altitude distributions become less common. The abundance and diversity of bees (particularly bumble bees) is likely to change over time due to climate change; pollinators from other orders (Diptera, Lepidoptera, Coleoptera) may respond differently, but there is much less research to help anticipate trends.

There is good evidence that pollinator activity, both seasonal and quotidian, is sensitive to shifts in climate – pollinator visitation rates are likely to rise due to temperature, and to decline during periods of inclement weather (rain and wind) or extreme heat events during the growing season. However, the sum effect of these changes in activity on pollination services and seed set is less easily predicted. Studies that specifically examine climate change effects on pollination success, via effects on pollinators, are not numerous, and the issue is

complex – observational studies typically quantify abundance and diversity of pollinators collected in a given area, while experiments that manipulate growing conditions are often limited in scale and evaluate the effects of a controlled change in one or a few features of climate. The consequences of climate for pollinators and plants cannot be determined by measuring each of them in isolation as there is often feedback within plant-pollinator networks. It is difficult to generalize in terms of linking changing communities and/or individual behaviour to altered success in pollination of crops, but declines in the abundance and diversity of pollinators are unlikely to have positive effects on pollination success.

In terms of the main climate features that will change in our region, the studies outlined in this review show that temperature is an important structuring agent (perhaps the most important abiotic influence) on plant-pollinator interaction networks. More specifically, warmer temperatures may not always spur increased pollination. Visitation rates are in part dependent on how plants respond to increased temperature in terms of both flowering phenology and nectar and pollen provisioning, and in part on the tolerance of key pollinators to ranges of thermal conditions. The most numerous managed pollinator, honey bee *A. mellifera*, is sensitive to temperature, and preserving wild pollinator biodiversity will be critical to ensuring pollination services continue as climate changes because in temperate zones wild pollinators often have greater thermal tolerance than domestic bees.

In preparing for the future, it is clear that maintaining high pollinator biodiversity should be a priority, because high biodiversity promotes resilience in pollination services. When more potential pollinator species are present, variability in weather, the timing of flowering, or other ecosystem perturbations are less likely to cause pollination failures. The literature suggests three potential avenues to promote pollinator biodiversity, which will ensure pollination services remain healthy in the BNFFG: maintaining non-crop floral resources for pollinators, monitoring how domestic pollinators are affecting wild populations due to competition and spread of disease, and ensuring landscape heterogeneity and habitat connectivity to provide a variety of nesting, predator avoidance, and foraging sites.

Lastly, broad-scale trends in pollinator abundance and diversity may not reflect local realities. Innovative methods of species distribution modeling will enable better understanding and prediction of pollinator communities at a scale that is meaningful to farmers, but these techniques require highly resolved data regarding both the pollinator species of interest, land use, and plant communities. To make accurate, local-scale predictions regarding the future of pollinators within the BNFFG, high quality data collection at appropriate temporal and spatial scales is required.

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II. Best Practices Review

Aim: identify and summarize resources and programs relevant to pollinator habitat enhancement and agricultural best management practices (BMP) for central and northern BC.

Numerous organizations provide guidance regarding various aspects of pollinator habitat enhancement and agricultural best practices for pollinator conservation. The resources listed below have been roughly divided into several different topic areas (pollinator habitat enhancement, interactive tools for pollinator habitat enhancement projects, technical guides and BMP documents for land management and agriculture, BMP for pesticide use, multimedia resources, and programs supporting pollinator enhancement and conservation projects). Within each topic area resources are listed alphabetically according to the name of the sponsoring organization, with short descriptions and links to relevant materials.

Resources for habitat enhancement include guides to help landowners identify and conserve pollinator habitat, tools to help select plant species well-suited to supply forage to pollinators, and guidelines for supporting the provision of nesting habitat for ground-nesting bees. The technical guides and BMP documents for land managers and producers cover a variety of topics, from site preparation methods for pollinator meadows and recommendations for tilling and cultivation practices for a variety of crops, to guidelines for grazing. Numerous organizations have also produced guides to reducing harms to pollinators when using pesticides – both synthetic and organic varieties. All of the resources below were inspected to ensure they were produced by organizations that provide information about their motivations and funding; in the interest of transparency, the first time an organization appears on the list, a short precis regarding its aims, affiliation, and funding (for non-governmental organizations) has been provided.

Guides for pollinator habitat enhancement

1. The Delta Farmland and Wildlife Trust (<https://deltafarmland.ca/>) is a non-profit organization funded by a mix of government grants and private donations that works with farmers in the lower mainland of BC to promote sustainable practices and preserve wildlife habitat.
 - a. One of their programs promotes retention of hedgerows and grass margins in areas used for agriculture. The information and linked fact sheets found at <http://deltafarmland.ca/our-programs/hedgerow-grass-margin-stewardship-programs/> provide good background and summarize existing research on how hedgerow and grass margin retention may be used to support populations of pollinators and other beneficial wildlife.
2. Pollinator Partnership Canada (P2C; pollinatorpartnership.ca) is a Canadian affiliate of an international organization (pollinator.org) supporting protection and promotion of pollinators and their ecosystems via conservation, education, and research. They

are funded via a mix of government and foundation grants, individual donors, memberships, gifts, and corporate sponsorships. They have developed several resources to aid with pollinator habitat enhancement:

- a. Region-specific planting guides are downloadable for different ecoregions in the BNFFG: the Fraser Basin (Prince George & Quesnel), Fraser Plateau (Smithers), and Columbia Mountain Highlands (Robson Valley):
<https://pollinatorpartnership.ca/en/ecoregional-planting-guides>
- b. General guide to planting forage suitable for domesticated and wild bees:
<https://pollinatorpartnership.ca/assets/generalFiles/Planting-Guide-FINAL-ISBN-June-2017-for-Web-English.pdf>
- c. For individuals looking to receive further training in how to implement successful pollinator habitat enhancement, Pollinator Partnership also offers training modules on a variety of topics that lead to certification as a Pollinator Steward: <https://pollinatorpartnership.ca/en/pollinator-steward-certification>
3. The Province of British Columbia sponsors the Food for Bees initiative, which is primarily intended to educate and encourage individuals to cultivate and/or conserve native plants that are important food sources for pollinators.
 - a. General information about Food for Bees:
<https://www2.gov.bc.ca/gov/content/industry/agriculture-seafood/animals-and-crops/animal-production/bees/food-for-bees?keyword=pollinator>
 - b. Important plants for foraging bees in British Columbia:
<https://www2.gov.bc.ca/gov/content/industry/agriculture-seafood/animals-and-crops/animal-production/bees/food-for-bees/bee-forage-plants>
 - c. Summer nectar and floral resources (native plants in many areas of the BNFFG, that should not be mowed or otherwise destroyed, particularly during the growing season): https://www2.gov.bc.ca/assets/gov/farming-natural-resources-and-industry/agriculture-and-seafood/animal-and-crops/animal-production/bee-assets/api_fs905.pdf
4. The United States Department of Agriculture (USDA) National Resources Conservation Service (NRCS) Plant Materials Program has produced numerous documents related to pollinator habitat enhancement (see <https://www.nrcs.usda.gov/wps/portal/nrcs/detail/plantmaterials/technical/publications/?cid=stelprdb1044847> for a summary. Documents that may be useful for pollinator habitat enhancement projects in the BNFFG may include:
 - a. A guide on choosing seed mixes for pollinators:
https://www.nrcs.usda.gov/Internet/FSE_PLANTMATERIALS/publications/idpncbr13772.pdf. While the specifics of this guide are aimed at landowners in Idaho and Utah, the general principles laid out in the guide are relevant to seed selection for pollinator habitat enhancement in any location.
 - b. A summary of the value to pollinators of 92 plant species selected by the NRCS for use in enhancement projects:
https://www.nrcs.usda.gov/Internet/FSE_DOCUMENTS/stelprdb1042140.xls.
5. The Xerces Society (xerces.org) is a non-governmental organization based in the United States, with a mission to promote invertebrate conservation. They are funded

via dues from society members, grants from a variety of government agencies, philanthropic organizations, and corporate partners (<https://xerces.org/funders>). They have many resources in their publications library (<https://xerces.org/publications>) that may be useful to producers in the BNFFG interested in pollinator habitat enhancement, including:

- a. Habitat planning for beneficial insects:
https://xerces.org/sites/default/files/2018-05/16-020_01_XercesSoc_Habitat-Planning-for-Beneficial-Insects_web.pdf
- b. Guide to enhancing habitat to provide nest sites for native bees:
<https://xerces.org/publications/periodicalsarticles/enhancing-nest-sites-for-native-bee-crop-pollinators>
- c. Guidelines for habitat enhancement and conservation specific to bumblebees:
<https://xerces.org/publications/guidelines/conserving-bumble-bees>
- d. Guide to collecting and using wildflower seeds:
<https://xerces.org/publications/guidelines/collecting-and-using-your-own-wildflower-seed>
- e. Guide to interseeding wildflower mixes in grasslands to increase floral resources for pollinators (plant species recommendations are for the Midwest and Great Plains, but the procedures described for site preparation and sowing are relevant to the BNFFG, particularly for ranchers with areas of open pasture): <https://xerces.org/publications/guidelines/interseeding-wildflowers-to-diversify-grasslands-for-pollinators>
- f. Guide to organic site preparation for wildflower planting (includes timelines and checklists): <https://xerces.org/publications/guidelines/organic-site-preparation-for-wildflower-establishment>; the accompanying description of organic site preparation methods for wildflower planting may prove useful as well: <https://xerces.org/publications/fact-sheets/organic-site-preparation-methods-comparative-overview>
- g. The Xerces Society also has developed a form that can help in planning and monitoring the progress of pollinator habitat enhancement projects:
<https://xerces.org/publications/habitat-installation-guides/pollinator-habitat-installation-forms>.
- h. Guide to maintaining wildflower diversity in pollinator meadows (i.e., maintaining sites that have previously been subject to pollinator enhancement measures): <https://xerces.org/publications/guidelines/maintaining-diverse-stands-of-wildflowers-planted-pollinators>

Interactive tools for pollinator habitat enhancement projects

1. The Center for Plant Conservation (<https://saveplants.org/>) archives data and publications produced by a network of academic institutions and government agencies in the United States. They have a database that can be searched by pollinator or plant species names (<https://saveplants.org/pollinator-search/>); while their focus is on plants

and pollinators native to the US, many of these species are also found in central and northern British Columbia.

2. The Database of Pollinator Interactions (<https://www.dopi.org.uk/>) is run by a group of academics from the University of Sussex (United Kingdom), and financially supported by the British Beekeepers Association and the Eva Crane Trust (a private philanthropic foundation). This is a repository for data on pollinator-plant interactions that allows users to search for pollinator or plant species by name, and shows which plants a given pollinator is known to interact with or which pollinators are known to interact with a given plant species. While this site is primarily focused on plants and pollinators in Great Britain, some of the same pollinator and plant species are also found in the BNFFG, making this a useful resource for individuals seeking to determine what kinds of pollinators are needed for a given plant, for example.
3. Pollinator Partnership Canada (P2C; pollinatorpartnership.ca) have developed an interactive tool to select appropriate native plants for pollinator enhancement projects. The tool can be used to find plants suited to specific ecoregions, morphology (tree, shrub, forb or vine), sun exposure and soil moisture levels:
<https://pollinatorpartnership.ca/en/find-your-roots>

Technical guides and BMP documents for land management and agriculture

1. Agriculture and Agri-Food Canada's guide, Native Pollinators and Agriculture in Canada (https://publications.gc.ca/collections/collection_2014/aac-aafc/A59-12-2014-eng.pdf) has brief descriptions of the main groups of insect pollinators found in Canada, and outlines the steps landowners can take to conserve and increase native pollinator diversity. Though produced in 2014, this guide remains relevant to producers today. This resource includes information on site selection and planting for pollinator habitat enhancement as well as guidelines for grazing and pesticide use to limit effects on pollinators.
2. The Province of British Columbia's Environmental Farm Plan guidelines on terrestrial biodiversity (particularly the sections on plant biodiversity) are generally applicable to and will promote pollinator biodiversity: https://www2.gov.bc.ca/assets/gov/farming-natural-resources-and-industry/agriculture-and-seafood/agricultural-land-and-environment/environmental-farm-planning/efp-reference-guide/full_efp_reference_guide.pdf; the full guide to biodiversity planning is available here: <https://www2.gov.bc.ca/assets/gov/farming-natural-resources-and-industry/agriculture-and-seafood/agricultural-land-and-environment/environmental-farm-planning/biodiversity-guide.pdf>
3. Pollinator Partnership Canada has produced several technical guides that can be found at <https://pollinatorpartnership.ca/en/ag-guides>; they have also produced a guide for land managers to preserve and create pollinator habitat, produced for farms in Ontario, but the principles and procedures laid out are generally applicable for central and northern BC:

<https://pollinatorpartnership.ca/assets/generalFiles/LandManagerGuide.Ontario.Farms.FINAL.PDF>

4. Seeds of Diversity (seeds.ca) is a non-governmental organization that promotes seed-saving and other initiatives associated with sustainable agriculture. They commissioned a report on best management practices for pollination (https://seeds.ca/pollinator/bestpractices/images/Pollination%20in%20Agricultural%20Landscape_Woodcock_Final.pdf). While this is intended for producers in Ontario, it includes specific recommendations for several crops grown in the BNFFG, including alfalfa, clover (*Trifolium* sp.) and assorted berries.
5. The Province of Manitoba has produced a set of guidelines for farming to conserve pollinator biodiversity, that includes information on identifying and creating nesting habitat, tilling and burning procedures, as well as safe pesticide use:
<https://www.gov.mb.ca/agriculture/crops/insects/pollinators.html>
6. The State of Minnesota's Department of Agriculture has produced a guide for farmers that includes information on tillage and management of non-productive areas that is applicable to many farming operations in the BNFFG:
<https://www.mda.state.mn.us/sites/default/files/inline-files/pollinatorsagland.pdf>
7. The United Nations Food and Agriculture Organization's pollination management guidelines (<https://www.fao.org/agriculture/crops/thematic-sitemap/theme/spi/scpi-home/managing-ecosystems/pollination-management/pm-how/en/>) describe management at multiple scales (field, farm, and landscape) to protect and enhance pollinator biodiversity.
8. The United States Department of Agriculture (USDA) has produced a handbook outlining BMP for pollinators on federal lands (<https://www.fs.fed.us/wildflowers/pollinators/BMPs/documents/PollinatorFriendlyBMPsFederalLands05152015.pdf>). It describes how numerous practices (including tillage, row cropping, mowing, grazing, and management of non-productive land) may be carried out to preserve or increase pollinator biodiversity.
9. The USDA NRCS Plant Materials Program has a full list of all pollinator-related technical documents archived at <https://www.nrcs.usda.gov/wps/portal/nrcs/rpublications/plantmaterials/technical/publications/?ptype=PL>. Producers in the BNFFG may be particularly interested in:
 - a. Technical report showing increased success of pollinator seed mixes sown in September in comparison to later in the year:
https://www.nrcs.usda.gov/Internet/FSE_PLANTMATERIALS/publications/oprpmcsr13783.pdf.
 - b. Technical report showing no benefits from alternate seeding (spatial) configurations for pollinator seed mixes:
https://www.nrcs.usda.gov/Internet/FSE_PLANTMATERIALS/publications/idpmpcsr13819.pdf.
 - c. Technical report comparing the viability of six commercially available pollinator seed mixes intended for use in the western states (some with similar growing conditions to areas in the BNFFG) under a variety of growing conditions:

- https://www.nrcs.usda.gov/Internet/FSE_PLANTMATERIALS/publications/idpmcsr13413.pdf.
- d. Technical report investigating the use of no-till methods to establish forb plant species in northern Idaho:
https://www.nrcs.usda.gov/Internet/FSE_PLANTMATERIALS/publications/wapmcsr12865.pdf.
10. The USDA's Sustainable Agriculture Research and Education (SARE; sare.org) program also produces knowledge transfer materials to help producers farm sustainably.
- a. Guide to cover-cropping for pollinators, which includes guidelines for crop selection and rotation strategies: <https://www.sare.org/wp-content/uploads/Cover-Cropping-for-Pollinators-and-Beneficial-Insects.pdf?inlinedownload=1>
- b. Guide to managing for alternative pollinators, that includes step-by-step guides to managing for numerous native pollinators (bumblebees, mason bees, leafcutter bees, etc), and information regarding how to match crops to native pollinator species: <https://www.sare.org/resources/managing-alternative-pollinators/>
11. The Xerces Society has produced several guides that provide explicit direction regarding a variety of habitat interventions and/or land management topics that can benefit pollinators, including:
- a. General guidance on farming for pollinators:
https://xerces.org/sites/default/files/2018-05/08-006_01_XercesSoc_Farming-for-Pollinators-brochure.pdf
- b. BMP for western rangeland (while the pollinator species in the BNFFG differ, the recommendations regarding grazing, mowing, and prescribed burning are still relevant to ranchers in central and northern BC):
<https://xerces.org/publications/guidelines/best-management-practices-for-pollinators-on-western-rangelands>
- c. For individuals interested in establishing new pollinator habitat, there is also a guide to establishing wildflower meadows from seed:
<https://xerces.org/publications/guidelines/establishing-pollinator-meadows-from-seed>, which includes information regarding site selection and preparation, plant selection and sowing techniques, as well as ongoing management.

BMP for pesticide use

1. Health Canada information about pesticide use, including specific ways to prevent harms to pollinators: <https://www.canada.ca/en/health-canada/services/consumer-product-safety/pesticides-pest-management/growers-commercial-users/pollinator-protection.html> This site includes links to best management practices documents for reducing effects on pollinators across several types of pesticide use (i.e., specific BMPs for spraying, insecticide-treated seeds, and other uses). There is also a

description of how practitioners can assess the risk to pollinators of pesticide application and links to outside resources on this topic.

2. Pollinator Partnership Canada has produced a technical guide aimed at reducing harms to bees when using pesticides:
https://pollinatorpartnership.ca/assets/generalFiles/Reduce.Bee_Poisoning.CanadaGuide.FINAL_noCrops2.pdf
3. The Province of British Columbia's production guides for a variety of crops have information on pollination specific to each crop, and include instructions on safe pesticide use to prevent adverse effects on pollinators (separate documents for fruits, vegetables, and hazelnuts; <https://www2.gov.bc.ca/gov/content/industry/agriservice-bc/production-guides>).
4. The Western Forum on Pest Management (<https://www.westernforum.org/index.html>) is a group of pest management experts and researchers that advise provincial government policymakers in BC, Alberta, Saskatchewan, and Manitoba. It is supported with funding from several governmental and producer organizations (e.g., BC Grain Producers' Organization, Canola Council of Canada, etc.). The Western Forum aims to help develop and implement control measures for a variety of agricultural pests in western Canada. As part of this, they provide guidelines on pesticide use.
 - a. General guidelines for different crops are available at:
<https://www.westernforum.org/WCCP%20Guidelines.html>
 - b. Specific guidelines to reduce impacts of pesticides on bees:
https://www.westernforum.org/Documents/WCCP/WCCP_documents/WCCP_Guidelines/WCCP_12/2013Chapter18HAZARDS.pdf
5. Xerces Society guide to reducing harms to pollinators when using organic pesticides:
<https://xerces.org/publications/guidelines/organic-pesticides>

Multimedia resources (webinars)

1. The USDA NRCS archives webinars, many dealing with pollinator habitat enhancement and/or agricultural BMP for pollinators (<https://conservationwebinars.net/search?SearchableText=pollinators>). Webinar topics in this area include:
 - a. Farming for beneficial insects:
<https://conservationwebinars.net/webinars/farming-for-beneficial-insects-pollinators-predators-and-parasitoids/>.
 - b. Farming for pollinators (which includes discussion of pollinator habitat enhancement: <https://conservationwebinars.net/webinars/farming-for-bees/>).
 - c. Identifying and capitalizing on opportunities for pollinator habitat enhancement on working lands:
<https://conservationwebinars.net/webinars/inventory-and-design-opportunities-for-pollinator-habitat-on-working-lands/>

- d. A discussion regarding the cover crops, herbs, and cut flowers that may best support beneficial insects: <https://conservationwebinars.net/webinars/cover-crops-herbs-and-cut-flowers-for-pollination-and-pest-management/>
2. United States Environmental Protection Agency webinar - integrated pest management: strategies for pollinator habitat promotion and conservation in agricultural areas: <https://www.epa.gov/pollinator-protection/integrated-pest-management-strategies-pollinator-habitat-promotion-and>

Support for pollinator conservation and/or habitat enhancement initiatives

1. CanPlant is a database run by Dougan and Associates, an ecological consulting firm in Ontario. They maintain a list of grant opportunities to support projects that involve planting native species (<https://can-plant.ca/Community-Planting-Grants.htm>). They also directly partner with community groups and individuals carrying out habitat restoration involving native plants.
2. BC Hydro's Community Re-greening program (https://www.bchydro.com/community/in_your_region/regreening.html) funds small-scale community planting projects that enhance ecological networks. Re-greening grants are awarded to municipal and Indigenous Nations' governments within BC Hydro's service area. The maximum value of grants is \$7500, and applications are due in late January each year.
3. The Investment Agriculture Foundation offers BeeBC grants (<https://iafbc.ca/bee-bc/>) to beekeepers and beekeeping associations to support small-scale and community-based projects that will contribute to bee health in the province. Eligible activities include planting forage resources for bees; the maximum value awarded is \$5000, but no timeline for applications for the 2022/23 funding cycle have been announced.
4. Wildlife Preservation Canada is an affiliate of an international non-governmental organization, funded via government and private grants. They offer wildflower seed grants (<https://wildlifepreservation.ca/wildflowerseedgrants/>), funded by corporate sponsors, that support the restoration and/or creation of pollinator habitat in Canada. Grants of up to \$2000 are awarded to communities or individuals that submit seed planting and/or stewardship plans. Application details are made available in the spring.
5. The Environmental Farm Plan (EFP) Program, sponsored by the Province of British Columbia (<https://www2.gov.bc.ca/gov/content/industry/agriculture-seafood/programs/environmental-farm-plan>) provides free consultations with EFP advisors, who work with producers to complete an agri-environmental risk assessment, which includes consideration of biodiversity and land use management decisions relevant to pollinators. While this program does not directly fund pollinator habitat enhancement or changes in agricultural practices, having an EFP in place can help producers identify ways to maintain or increase pollinator biodiversity. Additionally, having an EFP can facilitate access to supplemental funding, as some grant programs that support climate adaptation and regenerative agriculture require producers to have an EFP in place.

Other useful resources

Province of British Columbia, guide to beneficial insects:

https://www2.gov.bc.ca/assets/gov/farming-natural-resources-and-industry/agriculture-and-seafood/animal-and-crops/plant-health/beneficial_insects.pdf

Bumblebee Watch, a citizen science initiative to monitor bumble bees

<https://www.bumblebeewatch.org/>

III. Sampling Protocol

Preamble

Insects pollinate numerous crops, and in some cases this is an essential part of producing food; for other crops, insect pollination may not be essential but can substantially increase yields (Bommarco et al. 2012, Klein et al. 2007). While pollinator declines have been documented in many ecosystems (Potts et al. 2010), to what degree this impacts pollination success and/or reduced yields remains an open question (e.g., Winfree et al. 2007).

Measuring pollination services is a complex task, as the amount of pollination that occurs is a function of the abundance of pollinators, their floral visitation rates, and their efficiency (the quantity of viable pollen transferred per visit). Quantifying the abundance and/or diversity of species that are likely pollinators is the first step in characterizing a pollination network and determining the adequacy of pollination services.

For the purposes of agriculture, insect pollinators may include numerous families of Hymenoptera (bees and wasps), Diptera (flies), Lepidoptera (moths and butterflies) and Coleoptera (beetles, particularly those from the family Cerambycidae). These groups vary substantially in their reproductive phenology, foraging style, and microhabitat preferences, and thus it is unlikely that a single method or protocol for collection will adequately survey all of them. However, given that the baseline surveys for this project should be easily replicable, a combination of simple sampling methods (pan traps and vane traps) that provide complementary data seemed to be the most prudent way to survey pollinators. The results from the different sampling methods were compared to make recommendations regarding a streamlined methodology for future studies (see Baseline data set, section IV).

Protocol aims

1. Quantify (likely/potential) pollinator diversity
2. Quantify relative abundance of (likely/potential) pollinators
3. Identify the pollinators closely associated with crop plants

Site selection

Inclusion criteria for farms included only a minimum contiguous area under cultivation of 3 acres (5 m spacing of pan traps x 24 traps = 120 m transect; this equates to a circle with an area of ~ 2.8 acres or a square field of 3.6 acres). Climate and Agriculture Initiative BC recruited the majority of landowners; a few more sites were added via word-of-mouth recommendation from landowners recruited earlier in the year. A total of 18 sites were sampled during the 2021 field season, fairly evenly distributed across the BNFFG (four in the Telkwa area, one near Francois Lake, three in the Fort Fraser/Fraser Lake area, five near Prince George, and five in the Robson Valley). Most (16 of 18) of the recruited sites grew crops at least somewhat dependent on insect pollination.

Sampling density & frequency

Greater density and frequency of sampling result in more statistical power to detect differences, and greater likelihood of detecting change over time if further surveys are carried out in subsequent years. However, more sampling also requires more labour to classify the insects collected. Our primary method for sampling diversity was pan traps, and LeBuhn et al. (2013) indicated that when employed at 25 sites, sampled three times during the growing season, the FAO protocol (similar to ours, see Method 1 below) has > 90 % power to detect a 5 % decline in species richness in five years (assuming $\alpha = 0.05$). However, in systems with a high coefficient of variation (LeBuhn et al. used 25 % in their calculation, but this may be greater in highly seasonal temperate climates such as those found in the BNFFG districts), statistical power may be lower, but more frequent sampling is likely to at least partially resolve this by reducing the coefficient of variation at each site. To assess differences in pollinator abundance and diversity between the edges and the centers of fields, and the influence of adjoining land use, duplicate samples at each site were required. We enrolled 18 sites, sampled at 2 - 3 week intervals from mid-May to late August.

Method 1: Pan traps

Aim: Measure diversity and relative abundance of likely pollinators

Pan traps are the method for pollinator surveys adopted by the United Nations Food and Agriculture Organization (FAO) in 2016, because they are broad-spectrum and not costly to implement (LeBuhn et al. 2016, Sheffield and Ngo 2020). Generally, pan trap collections accurately reflect the diversity and abundance of many bee and Diptera species (Popic et al. 2013). Additionally, this type of sampling regime does not significantly affect pollinator abundance in areas where it is employed (Gezon et al. 2015). Using the FAO protocol also assures that comparisons can be made with many other ongoing studies, as the FAO is involved with research worldwide. Though the FAO protocol uses one set of traps per site, the addition of a second set of traps at the edge of a field enables analyses of the effect of adjoining land use types on the diversity and abundance of pollinators captured.

Studies comparing pan traps with other methods of surveying pollinators show that pan traps can be less reliable in estimating overall pollinator abundance in some systems (relative to sweep-netting; Popic et al. 2013, Prendergast et al. 2020), and generally are biased towards smaller species (numerous studies, *reviewed in* Prado et al. 2017). Additionally, pan traps provide limited information regarding the function of insects collected – the species captured may not actually be providing any pollination services. Thus, future inquiries may consider employing a supplementary method that more directly targets pollinators interacting with plants to provide additional certainty regarding which of the species collected via pan traps are in fact responsible for providing pollination services.

Pan traps: protocol

Materials

Pan traps (painted one of three UV-reflective colours: yellow, white, or blue; 48 per site, 16 of each colour)

Water (~ 2.4 L/site)

Detergent/Soap
Sieve
Tweezers
2 Collection vials (1 for ‘edge’ and 1 for ‘interior’ sample)
95 % Ethanol

Method

1. Two ~ 120 m linear transects were designated at each site, one in the interior of an area used for cultivation (surrounded on all sides by crop plants), and one along the edge, between an area used for crops and other, non-cultivated land.
2. On each transect, 24 pan traps (8 of each colour, alternating yellow, white, and blue) were placed on the ground, at ~ 5 m intervals. Traps were placed in the open (i.e., not where they would be obstructed by vegetation).
3. Each trap was partially filled with ~ 100 mL soapy water.
4. Traps were left out for one rain-free 24-hour period for each round of sampling. At the time of collection, each set of 24 traps were pooled to yield one ‘interior’ and one ‘edge’ sample for each site on each collection date.
5. When collected, the contents of each set of traps were drained (using a sieve) and stored in 95% ethanol pending further processing (Droege 2015).

Note: This method is essentially what the FAO recommends, but with an additional replicate at the edge of each field.

Method 2: Vane traps

Aim: provide a complementary (longer-term) indicator of diversity of likely pollinators

Vane traps are increasingly used in studies of pollinators and can provide data reflective of temporal change in pollinator communities (Kimoto et al. 2012). Additionally, in some contexts blue vane traps in particular have been shown to capture more individuals as well as greater taxonomic diversity of Hymenoptera pollinators in comparison with other commonly employed passive sampling methods (pan traps and trap nests; Prendergast et al. 2020). In this study, vane traps were intended to provide a longer-term measure of pollinator diversity (and perhaps collect some species that are not found in pan traps; Hall and Reboud 2019), as they were left out on sites all season, and the contents collected each time the site was sampled using other methods.

Vane traps: protocol

Materials

Vane trap (blue; 1 per site)
Nylon cord
Pole
Propylene glycol, diluted to 65 % with water
Sieve

Tweezers
Collection vial
95 % Ethanol

Method

1. One vane trap was suspended ~ 1.5 M above the ground at each site and filled with biosafe antifreeze.
2. Each time the site was visited to collect samples via pan trapping (see above), vane trap contents were sieved and placed in a collection vial with 95 % ethanol

Note: The location of each vane trap at each site was somewhat opportunistic – locations were chosen that would not pose any inconvenience to landowners, but were on the margin between cultivated and uncultivated land.

Sorting specimens

Materials

Samples
Petri dish
Tweezers
Dissecting microscope
Light source
Identification guides
Sample vials
Labels
95 % Ethanol
Data sheet

Method

1. Samples from vane or pan traps were placed in a Petri dish with 95 % ethanol.
2. Items were sorted to order, and unlikely or known non-pollinator specimens were set aside for submission to museum collections in the future (including, but not limited to Orthoptera, Odonata, Hemiptera, all Araneae other than those from the family Thomisidae, and predatory Coleoptera). All other items from pan trap samples were separated by morphotype, counted, and put into vials, while in vane trap samples only likely pollinators (Hymenoptera, Lepidoptera, Diptera from the families Syrphidae, Anisopodidae, Bibionidae, and *Lucilia* spp., Coleoptera from the family Cerambycidae, and Araneae from the family Thomisidae) were counted and retained for further analysis.
3. Single specimens of unique morphotypes were put in separate vials that were sent for identification via DNA barcoding by the Center for Biodiversity Genomics at the University of Guelph.

4. Remaining specimens kept at UNBC were left in 95 % ethanol and labeled with the following information: sampling date, location (GPS coordinates), and unique sample ID (site number and a suffix).
5. Identifications were made using a combination of DNA barcoding results and identification keys (Williams et al. 2014 and discoverlife.org for *Bombus* spp., Packer Lab Bees of Canada guides at <https://www.yorku.ca/bugsrus/> for other Hymenoptera, Miranda et al. 2013 for Syrphidae).

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IV. Baseline inventory

Sample collection & processing

A total of 131 pan trap and 71 vane trap samples were collected between May 18 and August 30, 2021 from 18 field sites in the BNFFG. As of March 1, all of the pan trap samples and 55 of the vane trap samples have been processed as per the protocol (see section III; a general summary of the number of items and morphotypes recovered from samples is shown in Table 1).

Table 1. Summary of specimens collected in different trap types and locations within each site (edge or interior of field for pan traps; all vane traps were located at the edges of fields).

	Trap type		
	Edge	Interior	Vane
Number of samples	68	63	55 [†]
Total items	8787	6529	9522*
Items per sample ($\bar{x} \pm SD$)	129.2 \pm 100.3	103.6 \pm 90.1	173.1 \pm 141.5*
Morphotypes per sample ($\bar{x} \pm SD$)	26.8 \pm 10.6	23.2 \pm 10.6	20.3 \pm 10.8*

[†]A total of 71 samples were collected, but only 55 have been processed to date.

*Only likely pollinators were quantified in vane trap samples, so these tallies are not directly comparable to those from pan trap samples.

Morphotype identifications

After the initial sorting of samples to separate out unique morphotypes, representative specimens of morphotypes were identified using DNA barcoding. The specimens selected for barcoding were chosen based on the likelihood they were pollinators (i.e., Hymenoptera, Diptera, Lepidoptera, beetles from the family Cerambycidae and spiders from the family Thomisidae were prioritized). DNA barcoding and vouchering was carried out by the Canadian Centre for Biodiversity Genomics at the University of Guelph for 665 specimens (see the accompanying file, *pollinator_voucher_images.pdf* for photos of voucher specimens). DNA barcoding involves taking a tissue sample from each specimen, extracting DNA from it, and amplifying and sequencing the DNA from a specific region of the genome that is known to vary among species, but differs little within species. The specimens in this study were sequenced using a primer set (LepFolR/L) that targets a portion of the mitochondrial cytochrome oxidase I (COI) gene, which can be used for DNA barcoding in a variety of arthropods (Hebert et al. 2003).

Of the 665 specimens sent for barcoding, 586 produced usable sequences (≥ 400 bp). To determine the identity of these specimens, the sequences were compared to the Barcode of Life database (BOLD, boldsystems.org), a repository for DNA barcodes and voucher specimen data (photographs, geographic coordinates, etc.; Ratnasingham and Hebert 2007). Species identities were typically assigned to specimens that showed a $\geq 99.85\%$ match to a

sequence in the database. When more than one species identity matched to a given sequence, phylogenetic trees and other resources (primary literature) were used to determine the correct identity to assign. In ambiguous cases, the specimen was assigned an identity at a lower level of taxonomic resolution (i.e., genus or family). For the specimens that failed to produce a sequence, similar morphotypes were identified in the lab using dichotomous keys, when possible. Table A1 (in appendix_species_list.xlsx file) shows a complete list of specimens sent for DNA barcoding and their respective identifications; full records for each specimen can be accessed by searching for the Sample ID codes at boldsystems.org.

Specimens not sent for barcoding were, in most cases, identified based on their similarity to specimens that were identified via barcoding. For some groups with many species that are not easily differentiated (e.g., *Helina* spp. flies), identifications of specimens were often limited to the genus rather than species level (labelled as ‘species uncertain’ in Table A2). For some highly recognizable morphotypes (e.g., Canadian swallowtail butterflies, *Papilio canadensis*) and a few recovered after the barcoding budget was exhausted, no specimens were sent for barcoding and identifications were made using dichotomous keys or other reference materials.

Taxonomic summary

A total of 346 unique morphotypes were identified from barcodes, and a further 32 morphotypes were assigned identities based on morphological characteristics of specimens (though not all of these were identified to species level). Flies (order Diptera) were the most numerous items collected, and also the most diverse, with a total of 217 species from 39 families, while bees (order Hymenoptera) were also well-represented, with 96 species from 15 families collected over the course of the season. The remaining groups collected in samples included four species of spider, 28 species of beetle, 30 butterfly or moth species, and a handful of obviously non-target species (e.g., stoneflies, order Plecoptera, and harvestmen, order Opiliones).

Though the sampling protocol used was targeted at pollinators, it is likely that some species caught were not important pollinators, so prior to further analysis, a literature search was conducted to assign ecological roles to each species. Each species (or, when information was lacking, genus or family) was categorized as a pollinator, beneficial for other reasons (e.g., predator or parasite of crop pests), and/or a potential pest. Of the 378 morphotypes identified in the final tally, 224 were considered likely or known pollinators (summarized in Table 2; see Appendix Table A2 for a full list of all morphotypes identified, their ecological role(s), and summaries of quantities collected). While the majority of likely pollinator species collected were flies (109 of 224 likely pollinator species), 116 of the 217 species of fly caught by pan and vane traps were unlikely to be pollinators. In contrast, most of the bee species (72 of 96 species) and all of the moths and butterflies collected (30 species) were likely to function as pollinators.

Table 2. Number of families and putative species from each order collected during pollinator sampling in 2021 and categorized by likely ecological role, as determined by literature searches.

Order	Total specimens	Number of families	Total species	Number of species		
				Pollinator	Beneficial	Pest
Araneae	47	2	4	3	4	0
Coleoptera	580	13	29	10	9	8
Diptera	14587	39	217	109	95	7
Hymenoptera	8558	15	96	72	19	9
Lepidoptera	1052	12	30	30	0	19
Other	14	n/a	3	0	1	0

The dataset was limited to exclude species that were not pollinators and those that function as pests as well as pollinators, to produce a list of species that are likely to contribute to pollination but not to crop damage ($n = 189$ species; Appendix Table A3). A species accumulation curve generated from these data indicates that the methods employed and number of samples collected were sufficient to inventory most of the pollinator species present, as richness approaches an asymptote when the number of samples processed exceeds 150 (Appendix, Figure 1). For the purposes of evaluating geographic and crop associations, species that are difficult to tell apart (*Aedes* sp., *Bibio* sp., *Dilophus* sp., *Drymeia* sp., and *Lucilia* sp.) were lumped together at the genus level; further evaluation within subgroups and geographic zones is pending.

The most numerous likely pollinator species was a dipteran, *Sylvicola punctatus* (a total of 1339 individuals collected, in 22 samples, from 14 sites), but the most widely distributed and frequently observed pollinators were bumblebees. Of the 189 pollinators evaluated, only four species were observed on all 18 of the sites sampled, all bumblebees from the subgenus *Pyrobombus*: *Bombus flavifrons*, *B. mixtus*, *B. perplexus*, and *B. vancouverensis nearcticus*. An additional three species were found on 17 of 18 sites: *Bombus rufocinctus*, *Lasioglossum leucozonium* (a halictid bee), and *Xylota subfasciata* (a hoverfly, from the family Syrphidae). Collectively, these seven species accounted for ~ 18% of all items collected during the 2021 field season, and ~ 30 % of all pollinators collected. Other likely pollinators collected in large numbers were small flies from the families Bibionidae (*Bibio* and *Dilophus* spp.) and Muscidae (*Drymeia* spp.).

Geographic distribution and crop associations

Sites were clustered by location into three zones, based on longitude: the central zone included eight sites between Burns Lake and Upper Fraser (between -122.0° and -125.0°); the eastern zone included five sites located in the Robson Valley (longitude $< -121.0^\circ$); and the western zone included five sites west of Burns Lake (all with longitude $< -125.0^\circ$). Sites were also categorized according to whether they were used to grow forage crops and/or as pasture for livestock, or were used for production of fruits and/or vegetables.

The most common species (see above) showed broad geographic distribution, had little apparent preference in terms of the type of crop being grown, and were present for most of the time period sampled. A few species of pollinator, captured in sufficient numbers, showed distinct geographical patterns. In particular, rare bee *Bombus frigidus* was not found east of Fort Fraser, while *B. cryptarum* was found only on sites in the Robson Valley, and *B. ternarius* was not detected west of Fraser Lake. While the most speciose group of pollinators, hoverflies, seemed to be patchily distributed among sites, detection of systematic differences in hoverfly species along the east-west corridor requires further analysis.

Pollinator species richness was generally positively related to sample size, which was unequal among geographic zones and crop types (west $n = 44$, central $n = 89$, east $n = 53$; forage/pasture $n = 112$, fruit/vegetable $n = 74$; Table 3). However, there were a couple of noteworthy exceptions to this. The first was the similar species richness of hoverflies found for both the central and eastern zones: even though there were twice as many samples collected from sites in the central zone, hoverfly species richness was essentially the same as for sites in the eastern zone. Additionally, the species richness of hoverflies and three groups of solitary ground-nesting bees (Hymenoptera, families Andrenidae, Colletidae, and Halictidae) was similar between crop types (forage/pasture and fruit/vegetable). Though there were approximately 20% more samples collected from sites used for forage crops and/or pasture than from sites used for fruit and vegetable production, the species richness of hoverflies (54) on sites used for forage crop production or pasture was only marginally higher than the number of hoverfly species documented (50) in samples collected from sites where fruits and/or vegetables were being produced. Additionally, the number of species of Andrenid, Colletid, and Halictid bees collected did not differ between crop types. Collectively, these results suggest that the number of samples collected was likely sufficient to inventory syrphids and some groups of bees, at the very least.

Table 3. Pollinator species richness within each family, separated by geographic area and dominant crop type; note that sample size differed among geographic zones (west $n = 44$, central $n = 89$, east $n = 53$) and crop types (forage $n = 112$, fruit and vegetable $n = 74$).

Order	Family	Total spp.	Geographic zone			Crop type	
			West	Central	East	Forage/ Pasture	Fruit/ Vegetable
Araneae	Thomisidae	3	1	3	2	3	2
Coleoptera	Cerambycidae	8	3	6	6	7	5
Coleoptera	Orsodacnidae	1	1	1	1	1	1
Diptera	Anisopodidae	1	1	1	1	1	1
Diptera	Asilidae	1	1	1	1	1	1
Diptera	Bibionidae	5	4	4	2	5	3
Diptera	Bombyliidae	1	1	1	0	1	0
Diptera	Calliphoridae	5	4	3	5	5	3
Diptera	Culicidae*	3	1	2	1	0	3
Diptera	Drosophilidae	1	0	1	0	0	1

Diptera	Muscidae	2	1	2	1	2	1
Diptera	Polleniidae	2	1	1	2	2	2
Diptera	Stratiomyidae	3	2	3	1	2	3
Diptera	Syrphidae	63	35	48	47	54	50
Diptera	Tabanidae	1	0	0	1	0	1
Diptera	Tachinidae	14	4	11	7	10	10
Hymenoptera	Andrenidae	14	9	12	4	11	11
Hymenoptera	Apidae	15	11	13	11	15	11
Hymenoptera	Colletidae	3	2	3	2	3	3
Hymenoptera	Halictidae	15	9	12	10	13	13
Hymenoptera	Megachilidae	16	12	15	8	14	11
Lepidoptera	Drepanidae	1	0	0	1	1	1
Lepidoptera	Erebidae	1	1	0	0	1	0
Lepidoptera	Noctuidae	1	1	0	0	1	0
Lepidoptera	Nymphalidae	6	2	5	3	3	4
Lepidoptera	Papilionidae	1	1	1	1	1	1
Lepidoptera	Sphingidae	2	0	1	1	1	1
Total number of species		189	107	150	119	158	143

*Species in this group are very difficult to distinguish; true species richness within this group is likely higher.

Field edge effects

Differences in species richness between pan traps from field edges relative to those located in field interiors were minimal, except for leafcutter and mason bees (family Megachilidae, 4 more species in edge than interior samples; Table 4). Though differences in species richness were in most cases small or nonexistent, differences in the abundance of several pollinators were much more evident. Edge samples had substantially more flies from the families Polleniidae and Syrphidae, as well as greater quantities of Andrenid, Colletid, and Megachilid bees; 12 species of hoverfly (including 4 *Xylota* spp.) and 10 species of bee (including 6 species of Megachilidae) were exclusively found in pan traps close to field edges. Interior samples contained greater numbers of bumblebees (family Apidae) and sweat bees (family Halictidae); 11 species of hoverfly and 6 species of bee (including 3 *Andrena* spp. and 2 species of Megachilidae) were exclusively found in pan traps in field interiors.

Table 4. Number of pollinator species in each family collected from pan traps located at field edges and in field interiors; a few families with hard-to-differentiate species have been excluded due to the likelihood that species richness in these groups is underestimated.

Order	Family	Edge		Interior	
		Species richness	Number of specimens	Species richness	Number of specimens
Araneae	Thomisidae	3	18	2	14
Coleoptera	Cerambycidae	2	16	4	9
	Orsodacnidae	1	16	1	3

Diptera	Anisopodidae	1	1	0	0
	Asilidae	1	3	0	0
	Bombyliidae	1	4	1	6
	Calliphoridae	3	83	3	48
	Phoridae	1	2	1	1
	Polleniidae	2	277	2	71
	Stratiomyidae	3	20	3	19
	Syrphidae	49	454	47	366
	Tabanidae	0	0	1	1
	Tachinidae	12	61	9	38
Hymenoptera	Andrenidae	10	184	12	129
	Apidae	10	307	11	395
	Colletidae	4	35	2	5
	Halictidae	14	623	13	822
	Megachilidae	14	107	10	34
Lepidoptera	Nymphalidae	3	5	4	9
	Papilionidae	1	4	1	3

Haying

Of the 13 sites used for forage crop production or pasture, seven were hayed during the growing season, and six were not. Haying dates at each site were noted; haying generally occurred from late June to mid-July, so to generate comparable estimates of species richness, samples from sites that were not hayed were divided into two groups: early season (before July 1) and late season (after July 1). Early season samples were collected at the same time as samples collected before haying, while late season samples were contemporaneous with samples collected after haying had occurred. On sites that were not hayed, species richness increased modestly over the course of the season, while on sites that were hayed species richness declined substantially following haying (Table 5). Interestingly, on sites that would later be hayed, pollinator species richness was higher early in the season than on similar sites that were not slated for haying. As a consequence of this difference early in the season, late in the season the number of pollinator species on sites that were not hayed was similar to sites that were hayed. Both hayed and unhayed sites experienced declines over the course of the season in species richness of Andrenid and Megachilid bees, and minimal changes in the richness of most other groups, with the exception of hoverflies. On sites that were hayed, hoverfly species richness declined following haying, while on sites that were not hayed, hoverfly species richness increased modestly over time.

Table 5. Summary across all sites used for forage crop production or pasture of pollinator species richness, determined early (before July 1) and late (after July 1) in the season for sites that were not hayed, and before and after haying (which typically occurred at some point in July) for sites that were hayed.

Order	Family	Hayed?		Hayed?		Change	
		No	Yes	No	Yes	Late – Early	After – Before
Araneae	Thomisidae	3	3	2	2	-1	-1
Coleoptera	Cerambycidae	3	4	6	5	3	1
Coleoptera	Orsodacnidae	1	1	0	1	-1	0
Diptera	Anisopodidae	1	1	1	1	0	0
Diptera	Asilidae	0	0	1	0	1	0
Diptera	Bibionidae	6	6	1	1	-5	-5
Diptera	Bombyliidae	1	1	0	0	-1	-1
Diptera	Calliphoridae	3	4	5	4	2	0
Diptera	Culicidae	1	1	1	1	0	0
Diptera	Muscidae	1	3	2	2	1	-1
Diptera	Phoridae	0	1	0	0	0	-1
Diptera	Polleniidae	0	1	2	1	2	0
Diptera	Stratiomyidae	2	2	2	0	0	-2
Diptera	Syrphidae	27	37	34	31	7	-6
Diptera	Tachinidae	1	7	5	7	4	0
Hymenoptera	Andrenidae	7	11	1	5	-6	-6
Hymenoptera	Apidae	12	14	13	13	1	-1
Hymenoptera	Colletidae	2	2	1	2	-1	0
Hymenoptera	Halictidae	9	8	9	11	0	3
Hymenoptera	Megachilidae	10	10	7	8	-3	-2
Lepidoptera	Drepanidae	0	0	0	1	0	1
Lepidoptera	Erebidae	0	0	1	0	1	0
Lepidoptera	Noctuidae	0	0	1	0	1	0
Lepidoptera	Nymphalidae	1	2	2	3	1	1
Lepidoptera	Papilionidae	1	1	1	1	0	0
Lepidoptera	Sphingidae	0	0	1	0	1	0
Total number of species		92	120	99	100	7	-20

Relative performance: vane and pan traps

Blue vane traps were more successful than pan traps in capturing specimens from two groups of pollinators in particular: long-horned beetles (Coleoptera: Cerambycidae; eight of nine species in vane traps versus five in pan traps), and bumblebees. In particular, larger-bodied and less common bumblebee species (i.e., *B. appositus*, *B. cryptarum*, *B. frigidus*) were predominantly collected via vane traps, rather than pan trapping. Additionally, vane traps captured substantially greater numbers of calliphorid flies (particularly *Lucilia* spp.) than pan traps did, but both methods captured similar diversity in *Lucilia* spp.

In contrast, vane traps were generally ineffective for inventorying hoverflies, as only 32 of 63 species of hoverflies identified were found in vane traps, while 61 hoverfly species were collected in pan traps. For the vast majority of bees (other than bumblebees), most species (14 of 14 Halictidae, 13 of 14 Andrenidae, 16 of 16 Megachilidae) were collected from pan traps; these types of solitary, ground-nesting bees were much less reliably censused by vane traps (which captured only 6 Andrenid species, 10 Halictid species, and 10 Megachilid species).

Notes for future sampling in the BNFFG

The existing sampling protocol provides a good measure of pollinator diversity but does not explicitly identify which pollinators are closely associated with crop plants (Popic et al. 2013, McCravy 2018, Prendergast et al. 2020). While some associations may be identified by determining which pollinators are collected more often in areas with particular crop plants and literature searches of relevant databases, more highly structured information regarding which pollinators visit crop plants of interest may be required in the future. At the very least, future surveys should include some identification of the floral resources available during a given sampling period to help identify plant-pollinator associations that may be of use to agricultural producers. Future studies may want to consider methods that allow for more direct assessment of pollinator activity, such as direct observation, hand collecting from plants during the course of a transect walk, and/or systematic sweep-netting (Nielsen et al. 2011, Popic et al. 2013, McCravy 2018). Additionally, to help identify the most important pollinators for crops, specimens collected via these methods should be inspected to determine if they are carrying pollen, and perhaps have any pollen present identified by an expert.

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Appendices – see accompanying spreadsheet, appendix_species_lists.xls

Table A1. List of specimens sent for barcoding (see appended image gallery for photos), with the success of barcoding assays and resulting identifications. A successful assay resulted in a sequence of ≥ 400 bp that corresponded to similar specimens in the BOLD database (i.e., sequences were not used for identification if they matched to *Wolbachia* or other clearly incorrect organisms). For specimens that were not successfully barcoded, if identifications were assigned to similar morphotypes using dichotomous keys the result is marked with an asterisk (*) in the table.

Table A2. List of all species identified, organized by taxonomic group, with ecological roles (determined via literature searches) and summarized collection details (total number collected, and number of samples each species was present in).

Table A3. Summarized records for all species collected during the 2021 field season considered likely pollinators but not pests, including details regarding geographic distribution, trap type, crop type, and timing.

Figure 1. Species accumulation curves generated from all pollinators, at all sites, from samples collected during the 2021 field season. The header on each panel indicates the method used to generate the value for expected richness (shown in pink; S = closed system); observed richness is shown in blue.

Full records worksheet: a complete list of all specimens sorted and identified to date, with geographic information removed to protect the anonymity of cooperating landowners.

