ECOSYSTEM SERVICES, CONTEXT AND THE SOCIAL BASIS FOR CONSERVATION: THE CASE OF INSECT POLLINATION BENEFITS TO LOWBUSH BLUEBERRY

by

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For Sarah, the only one who can trace this work back to its origin.

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ABSTRACT

The conservation of ecosystems is increasingly justified on the basis of the human welfare they generate. Since many of the “services” supplied by ecosystems are not currently captured in markets, advocates of ecosystem service value (ESV) assert that the social basis for conservation can be revealed through estimating the hidden welfare benefits. But, despite a number of ESV-inspired initiatives, global ecological degradation continues to accelerate, suggesting that the social basis for conservation remains elusive.

I locate the possibility for conservation in the present through a critique of ESV. I begin by focusing on methodological problems associated with a well-defined ecosystem service: insect pollination of agricultural crops. My review of global scale pollinator valuations indicates that current efforts fail to: 1) decouple contributions of wild pollinator species (i.e., the focus of conservation) from those of managed species, 2) adequately assess pollinator yield benefits relative to the dynamic and heterogeneous character of agricultural production, and 3) consider the complex character of agricultural commodity markets leading to an exaggeration of the impact of pollinator declines on agricultural prices.

Next, I tested the assumption that yield in highly pollinator-dependent crops is strictly pollen limited through an empirical study of lowbush blueberry (*Vaccinium angustifolium*). Experimentally I demonstrate that although berry set is related to pollinator abundance, this is only the case if pests and diseases are controlled. Moreover, a two-year multivariate study of berry set and yield in commercial fields demonstrates that variation in pollinator visitation rate has an inconsistent effect on berry set and is clearly governed by other agronomic factors.

My initial focus on the methodological shortcomings of ESV implies that conservation would advance if only the technical deficiencies of current ESV calculations were addressed. Yet it is precisely this focus on the technical dimensions of ESV methodology that disconnects the practice of valuation from the various ways it is presently taken up in society. After reviewing various criticisms of ESV, I develop a critical theory approach that stipulates that the possibility for conservation cannot be separated from the social context that presently gives rise to ESV.

LIST OF ABBREVIATIONS USED

AIC	Akaike information criterion
B	Boron
CANPOLIN	Canadian Pollination Initiative
Ca	Calcium
cm	centimeter
df	Degrees of freedom
e.g.	for example
et al.	and others
ESV	Ecosystem Service Value
ESVn	Ecosystem Service Valuation
E.U.	European Union
F	F-value
GLM	General linear model
h	hours
ha	hectare
i.e.	that is
Inc.	Incorporated
K	Potassium
kg	Kilograms
km	Kilometer
Max	Maximum

Mg	Magnesium
mg	Milligram
m	Meter
min	Minimum
min.	Minute
Mt	Metric tonne
N	Nitrogen
n	Sample size
NSERC	Natural Science and Engineering Research Council of Canada
P	Phosphorous
P	P-value
ppm	Parts per million
r^2	Correlation coefficient
SD	Standard deviation
SE	Standard error
t	t-test statistic
U.S.	United States of America
z	z-test statistic
Δ	difference between a model's AIC from the best-fitting model
χ^2	Chi-square statistic
%	Percent
~	Approximately

<	Greater than
>	Less than
±	Plus or minus
≤	Greater than or equal
≥	Greater than or equal
°C	Degree Celsius

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My field research in 2013 and 2014 relied heavily on the assistance of Steve Javorek at Agriculture and Agri-Food Canada in Kentville. Steve is an inexhaustible fountain of ideas and techniques. I count it as one of my blessings that he took an interest in this project and decided to take me under his wing. The opportunity to work so closely with this wild bee authority is something I will draw on for years to come. An hour with Steve always generated dozens of wonderful research questions and his support of the field work in 2013 and 2014 was pivotal. Matt Grant also deserves considerable recognition for juggling the million tasks that field bee research requires (but without seeming to break a sweat). I owe a considerable debt to Steve and Matt.

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CHAPTER 1 INTRODUCTION

1.1 ECOSYSTEM SERVICE VALUE AND CONSERVATION

The launch of the journal *Ecosystem Services* in 2012 appeared to signal the maturation of the ecosystem service value (ESV) approach to conservation. As the incoming editor, Leon C. Braat (2012), would explain, the journal was necessitated by the momentum that had built since the Millennium Ecosystem Assessment (2005) and which continued around successor projects, such as The Economics of Ecosystems and Biodiversity (TEEB) (2010). These international coordinated efforts set the task of determining the state of ecosystems globally and their relevance to human welfare with ever-greater detail and precision. They were, in themselves, the product of mounting interest, beginning in the 1970s, in sustainable development, and its elaboration in the 1990s to include the idea that meeting “the needs of the present without compromising the ability of future generations to meet their own needs” (Brundtland and World Commission on Environment and Development, 1987: 8) would entail maintaining a threshold amount of ecosystem function (Costanza et al., 1997). This long gestation has enabled the accumulation of research, theoretical approaches and practical experience that has not only extended the scope of our understanding into how ecosystems contribute to human welfare, but also enabled the estimation of their monetary value with precision unimaginable even a decade earlier.

The growth of ESV approaches suggests that the prospect of bringing many conservation initiatives in line with the goals of society has become largely a technical issue[1] (Daily, 2000; Daily et al., 2009; Kremen, 2005; Kremen and Ostfeld, 2005). From this perspective, conservation priorities could be readily addressed by integrating more and better valuations into policy and institutional frameworks. In situations where the benefits of these services quite clearly outweigh the costs of alternatives, ESV, it is thought, will give previously languishing conservation priorities a new lease on life (e.g. Armsworth et al., 2007; Costanza, 1996; Daily, 1997; Liu et al., 2010; MEA, 2005). This approach, in other words, envisions that ever-more sophisticated and complex valuations will transform the ecologically-destructive character of society in the direction of sustainability.

Critics, however, have expressed pessimism over the capacity of valuations to accomplish their stated ends. The growing number of valuations and their influence in defining conservation initiatives, they point out, is implicated in a widening "sustainability gap" (Fischer et al., 2007). The proliferation of ESV, it would seem, does not mark our increasing capacity to shape our future towards sustainable ends, but rather the opposite, a sense of growing helplessness in the face of accelerating environmental

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[1] As Kremen (2005: 469) asserts: "Carrying out such an agenda will present an enormous logistical, financial and scientific challenge, but it is not outside our human potential. We have only to look at the enormous strides made in medicine or space exploration, to name a few, to realize that is possible, given careful planning and sufficient resources".

degradation. Norton and Noonan (2007: 665) go so far as to claim that the rubric of ESV does not represent any kind of progress, but rather a regression, a “wrong turn.” Ecosystem service value does not emerge from working through the challenges of integrating contemporary society with ecology, they assert, but through the *avoidance* of these challenges.

Although ESV is strongly connected to the interdisciplinary heterodox field of *ecological economics*, many comment that the growth of ESV in the late 1990s provoked a crisis within ecological economics itself (Gómez-Baggethun et al., 2010; Røpke, 2005; Sagoff, 2012; Spash, 2011). Ecological economics coalesced in the 1980s as an attempt to integrate social and natural sciences around the position that economic activity, while resulting in efficient allocation, was ultimately bounded within objectively measurable biophysical limits, within which the social problem of unfair distribution of surpluses would need to tackled (Costanza, 1996; Daly and Townsend, 1993). In this manner, ecological economics looked to distinguish itself from *environmental economics*, a sub-discipline of the mainstream neoclassical tradition that strove to theorize ecological degradation entirely in terms of nineteenth century marginal utility theory. A key application of marginal utility within environmental economics came in the form of cost benefit analysis of pollution abatement polices in the 1980s (Newbery et al., 1990), which significantly propelled the importance of valuation approaches (Vatn and Bromley, 1994). But the growing influence of environmental economics posed a considerable challenge to the aspirations of ecological economists who looked to radically restrict the scale of economic activity within biophysical limits (Sagoff, 2012). In fact, the growth of

ESV initiatives by ecological economists functioned to blur the distinction between mainstream environmental economics and ecological economics (Spash, 2011). The erosion of the differences between these two approaches has resulted in a growing sense within ecological economics that its own categories are insufficient to bring about transformation (Gómez-Baggethun et al., 2010; Gómez-Baggethun and Ruiz-Pérez, 2011; Kallis et al., 2013; Kosoy and Corbera, 2010; Muradian et al., 2010; Norton and Noonan, 2007; Pascual et al., 2014; Ruhl et al., 2008; Sagoff, 2002, 2012).

1.2 ECOSYSTEM SERVICE VALUE TO AGRICULTURE AND WILD POLLINATING INSECTS

The backdrop to the broader questions of this dissertation is the extent to which agricultural yield depends on the activity of wild pollinating insects. ESV approaches predict that investments in wild pollinator conservation would generate sufficient value to offset the costs of restoring or maintaining habitat and reducing farm practices that disrupt pollinator life history (Garibaldi et al., 2014; Winfree, 2010). Insect pollination of agricultural crops is ideal for exploring shortcomings associated with ESV approaches because it lacks ambiguity: pollination operates within well-defined markets (i.e., agricultural commodities), pollinator populations can be enhanced by conservation (Kennedy et al., 2013; Rundlöf et al., 2008), and recent widespread attention to the importance of these insects to agricultural production suggests strong prospects for generating political pressure for pollinator conservation (Peterka, 2014; Winfree, 2010).

Arguably, insect pollination services offer one of the best opportunities for motivating conservation using an ESV approach.

Wild insect pollinators, particularly bees (order Hymenoptera, superfamily Apoidea) and to a lesser extent flies (Diptera), beetles (Coleoptera), and moths and butterflies (Lepidoptera), generate agricultural value by vectoring pollen to the stigma of the same flower (autogamy), between flowers of the same plant (geitonogamy), or among flowers on different plants (xenogamy) (Free, 1993). As pollen grains germinate, they extend a pollen tube down through the floral style to fertilize ovules that develop into seeds. Although the number of seeds per flower can be strongly related with yield, not only in terms of number of seeds (e.g., oilseed crops) but also with respect to fruit set and quality (e.g., berry crops), this relationship is influenced by the capacity of the plant to carry fully-pollinated flowers to harvestable seeds and/or fruit (Free, 1993). Frequently, crop plants produce far more flowers than the plant has resources to carry to harvest, resulting in substantial fruit abortion during crop development (Bos et al., 2007b). Moreover, different insect taxa often have crop-specific capacities to transfer pollen of suitable quality to produce a seed (Garibaldi et al., 2013) and there is some evidence that high seed or fruit set may also depend on a combination of different pollinator species (Greenleaf and Kremen, 2006; Winfree et al., 2007). Consequently, the value of pollinator activity to agricultural production cannot be straightforwardly calculated, an issue that I address in Chapter 2. In addition, significant crop production can occur without insect pollinators. Some plants are pollinated either entirely or in part through a different mechanism (e.g., wind pollination of cereal crops), and others are primarily propagated using vegetative methods (Klein et al., 2007). Yet, given the substantive mediating effect of insect pollinators on yield, the Millennium Ecosystem Assessment (2005) categorized insect pollination as a *regulating service* linked to the primary *provisioning service* of food production.

My empirical research focuses on the largest source of pollinator-dependent yield in Atlantic Canada, eastern Quebec and Maine, lowbush blueberry (*Vaccinium angustifolium* Ait., syn. "wild blueberry"). Commercial cultivation of lowbush blueberry involves large-scale management of wild stands of plants. Blueberry plants in commercial fields establish naturally and are brought into cultivation by clearing the forest cover and limiting competition from other plant species with the use of selective herbicides and biannual mowing or burning of the fields. Plants spread radially through a network of underground rhizomes, which send up 9 – 27 cm-tall fruit-carrying stems (ramets) every 2-30 cm (Vander Kloet, 1988). Since the plants in a field constitute a wild population, ramets from a given plant frequently exhibit distinctive and uniform patterns of leaf, stem, and flower colour, enabling individual plants – termed clones – to be visually distinguished. After an extensive development period (10+ years) the plants in actively managed fields grow together forming contiguous cover and are intensively managed with agrochemicals, particularly nitrogen and phosphorous-based fertilizers, herbicides, fungicides for leaf diseases and blossom blights, and insecticides for insect pests (Yarborough, 2004). Production is managed along a two-year cropping cycle with alternating vegetative (syn. “sprout phase”) and fruiting (syn. “crop phase”) years. Although up to seven different *Vaccinium* species are commonly found growing under the forest canopy only few species proliferate in the absence of forest cover (Vander Kloet, 1988). Consequently, the majority of yield is obtained from a single species *Vaccinium angustifolium*, with a minor contribution from *Vaccinium myrtilloides* which is largely intolerant of both the biennial burning or pruning associated with commercial cultivation (Hall and Aalders 1961; Vander Kloet, 1988). Berries are largely sold frozen and the main centres of production are northeastern Maine, the four Atlantic Canadian provinces, and the Lac-Saint-Jean region of Quebec (Vander Kloet, 1988).

Blueberry plants are completely dependent on insects to move pollen among flowers to set fruit (Cutler et al., 2012a; Lee, 1958). Although *V. angustifolium* has hermaphroditic flowers, it is functionally self-sterile owing to the shape and relative position of the anthers to the stigma (herkogamy) and to being protoandrous (i.e, pollen is released a few days ahead of stigmas being receptive) (Vander Kloet, 1988). Moreover, even if self-fertilization is induced by experimental procedures, in most cases it results in aborted seeds (Bell et al., 2009a; Bell et al., 2010).

Blueberries are an early flowering plant often blossoming within a month of the earliest tree and shrub species (*Salix* spp., *Acer rubrum*, *Prunus pensylvanica*) that occur in the landscape. Consequently, pollination of blueberry depends on a narrow subset of early season pollinator species, primarily *Andrena* spp., *Protoandrena* spp. and newly emerged queens of early-season species of *Bombus* spp., with some influence being exerted by nocturnal taxa (Cutler et al., 2015; Cutler et al., 2012a; Fulton, 2012; Manning and Cutler, 2014). There is also a well-defined market for the rental of managed pollinators. In Nova Scotia this includes approximately 25,000 honey bee colonies (*Apis* *mellifera* L.), a managed species of bumble bee (*Bombus impatiens*), and the alfalfa leafcutter bee (*Megachile rotundata*). These managed and wild species vary with respect to the speed at which they travel between flowers and the amount of pollen they deposit per visit, with *Andrena* spp. and *Bombus* spp. queens contributing 3.6 and 6.5 times more pollen per unit time than honey bees (Javorek et al., 2002).

Lowbush blueberry represents an ideal model for contextualizing pollinator benefits in relation to other plant limitations. The crop is grown on relatively small fields so that wild pollinators nesting on the edges can often deliver pollination services across a large area of the crop (Cutler et al., 2015). In addition, as a perennial crop, pollinators nesting within the blueberry field are not disturbed by annual cultivation. Moreover, as a rare example of a native plant grown commercially across its natural range, the insect pollinator fauna of lowbush blueberry appears to synchronize its life history events relative to the crop's bloom period and is adapted to efficiently transferring pollen among flowers. For these reasons lowbush blueberry is arguably the most wild pollinator-dependent crop in North America. This was recently confirmed by Eaton and Nams (2012) who estimated that over a quarter of lowbush fields studied in Nova Scotia received adequate pollination from wild species alone, without growers having to rent managed honey bee colonies.

1.3 CRITICAL THEORY, SOCIAL CONTEXT AND TRANSFORMATION

> *No method can claim a monopoly of cognition, but no method seems authentic which does not recognize that these two propositions are meaningful descriptions of our situation: 'The whole is the truth,' and the whole is false.* (Marcuse, 1982 [1960]: 451)

Chapter 2-4 of the thesis contextualizes biophysical/agronomic dimensions of pollinator ESV using empirical data derived from the literature and field studies in lowbush blueberry. In contrast, Chapter 5 draws attention to the difficulties in contextualizing the subjective and social dimensions of ESV using the methodology of critical theory.

Critical theory is a complex methodology which seeks to recognize and make explicit how, in an integrated capitalist society, it is impossible to think outside of our societal context and how thoughts which *appear* to be outside this context are always within (Horkheimer 1972 [1937]). Starting from this premise critical theory seeks to specify the possibility that the existing state of affairs (and therefore the deeper structure of society) can be changed (transformed). For example, from what standpoint or criteria could we envision a less ecologically destructive *form* of social life if our thoughts about that transformation are situated squarely in the *present form* of society? Critical theory is concerned with the problem of how individuals and social groups might become conscious of the inherently contradictory structure of contemporary society without resorting to an Archimedean standpoint that purports to stand outside our social structure

(Horkheimer 1972 [1937]). Here I elaborate on this methodology using the Marcuse quote above.

I pursue the critical theory first developed by Marx (1818-1883) and then continued by members of the early Frankfurt School (e.g. Adorno, Benjamin, Horkheimer, Marcuse) in the early to mid-twentieth century. This critical theory approach can make sense out of the paradoxical process at work in the present, and which I develop as central to explaining the growth of ESV approaches in Chapter 5. In the present, our capacity to mitigate ecological degradation appears to fall farther and farther behind, in spite of that fact that society becomes more conscious of this degradation. In order to approach such a problem I will use a methodology known as *immanent critique*, which is the methodology with which critical theory confronts the twofold task of *critique* and *transformation*. Critical theory responds to how thought can grasp the possibility of a less ecologically destructive form of society ("the whole is false" and is in need of transformation) when the social context of this thought is characterized by increasing environmental degradation amid growing environmental attention and concern (i.e., without assuming the Archimedean standpoint outside of society). The critical theory approach I use in the dissertation proceeds from the premise that in such a society, thought which neglects its immersion in social context ("the whole is true") not only fails to transform society in the direction of being less ecologically destructive, but actively reflects this ecologically destructive context.

The critical theory approach I pursue is exemplified, as Rose (1976: 74) explains, in Marx's immanent approach to the theory of classical political economy (of e.g., Adam

Smith and David Ricardo), which, "involves deriving the state of society from its appearance in those theories and concepts." In other words, rather than attempting to theorize society directly—a process which, at best, describes the world as it *is*—Marxian critical theory seeks to determine the *missed potential* for society (e.g., the potential for society to be less ecologically destructive) from the way in which society *appears* to itself. Marx's critique of political economy (i.e., his working through the forms of *actual* consciousness *of society* embodied in categories like capital, labour, surplus value) resulted in his locating the contradictory moments in the most characteristic, or symptomatic thought about that society. *Critique*, in this sense, needs to be distinguished from *criticism*, which attempts to separate "true" from "false" thinking. *Critique* attempts to locate the conditions of possibility for transforming the whole social structure ("the whole is false") in and through the actual full scope of thought and practice in that social structure ("the whole is truth").

Just as Marx's *critique* of the categories of classical political economy became the grounds for thinking capitalist society *could* transform into a form of social organization beyond capitalism, a *critique* of ESV would need to locate the possibility for a more adequate relationship between society and ecosystems *within the social context that gives rise to ESV*. Put another way, the commitment to critique holds that the only means for understanding the possibility of transforming society into a less ecologically destructive form is through the *actual* thoughts and practices through which ecology is presently recognized by society, such as ESV and its criticisms. The ultimate aim of such work would be to understand the significance of the emergence of ESV indirectly; that is, by reinterpreting the shortcomings as unrecognized, yet integral, ciphers of a runaway, ecologically destructive pattern of development.

1.4 WORKING IN AND THROUGH POLLINATOR ECOSYSTEM SERVICE VALUE

The primary objective of this work is to clarify the reasons why sustainable transformation of society has remained elusive and how this problem is indicative of more general problems within society at present. A key assertion of this thesis is that the activity of estimating ESV, as well as the manner in which these estimates are taken up and used within society, opens a door to understanding not only how the biophysical/agronomic dimensions of our world interact with society, but how the form of this relationship might conceivably change. By regarding ESV approaches as symptomatic of this relationship (i.e., as uncritically reflecting the actual relationship) I will argue that the tasks of ecological and social theory research is to more clearly pose the question of why ecological degradation seems to accelerate in spite of our efforts.

For ecological research, ESV estimates present important hypotheses about the relationship between ecosystem services and human welfare. Using the example of pollinator ESV I will assert that these hypotheses are frequently not tested, but carried forward as a firm foundation upon which the social necessity of pollinator conservation is advanced. Research focused on questioning this foundation can provide much needed insight into how ecosystems *actually* interact with complex and dynamic social processes, such as agriculture. Although such work would invariably improve and transform the ESV framework, rendering it more descriptive of current conditions, I argue in this dissertation that it only provides a partial response to the pervasiveness of ESV estimates within society. A purely technical and methodological approach to the growth of ESV threatens to affirm the current form of society. An ecological approach must, therefore, be accompanied by a critical social theory that can not only explain the growth of ESV, but understand how it simultaneously describes both the current ecologically destructive character of society, as well as the social basis for overcoming its destructive features.

While there are a wide range of ecological economic critiques of ESV, I argue in this dissertation that they are insufficiently critical. While it may certainly be valid, for example, that ESV fails to capture ecosystem complexity or the plurality of social norms (Vatn and Bromley, 1994), these types of criticisms fall short of what is necessary. In order to reverse the "sustainability gap", critique would have to go deeper and understand *why* ecologists and environmental philosophers increasingly find themselves objectively forced to work within the ESV framework. In other words, while Spangenberg and Settele (2010: 327) may be correct in observing that ecologists "feeling that biodiversity is valuable (in whatever sense)... have taken resort to the economic definition of value to make their point understood and to provoke policy resonance," the objective reasons for them doing so, the necessity, is left to one side and the pressures associated with embracing an "economic definition" are never specified. If ESV comes at the cost of liquidating key ecological principles, how might we attempt to understand the substance and process *underlying* such liquidation?

For these reasons my thesis assumes the specific form that it does. My research ultimately looks to engage the problems of ESV in and through existing debates. So, rather than oppose sides of the debate to one another, I will attempt to understand why these positions are arising in the first place, or to consider these positions as symptomatic of a deep and, as of yet, unclear social context. This approach means that rather than take for granted ESV as adequate or the inability of an ESV approach to auger wild bee conservation as inevitable, I will attempt to understand why such divergent positions are arising in the present and, through this, draw attention to what is missing in our ability to understand what drives the "sustainability gap." Since these debates are presently scattered across a number of disciplines, I am forced to cross boundaries in order to properly take up my work. From the outset this kind of interdisciplinary work might be mistaken for an attempt to forge a synthesis. Many ecologists, economists and philosophers have advocated for such a synthesis as a way to get beyond the problems of the present (Castree, 2014; Fischer et al., 2007; Norton, 2005; Paehlke, 2005). Instead I attempt to push the debate across several disciplines in order to create the grounds upon which a deep absence might be sensed, one that will contribute to an urgency to intensify these problems rather than try to resolve them prematurely[2].

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[2] "One continually finds the word *critique*, if it is tolerated at all, accompanied by the word *constructive*. The insinuation is that only someone can practice critique who can propose something better than what is criticized…By making the positive a condition for it, critique is tamed from the very beginning and loses vehemence" (Adorno, 2008 [1963]: 287)

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CHAPTER 2 WHERE IS THE VALUE OF VALUING POLLINATION ECOSYSTEM SERVICES TO AGRICULTURE?

The following chapter has been published in Ecological Economics (2015: 109:59-70) with G. Christopher Cutler and Peter Tyedmers as co-authors.

2.1 ABSTRACT

Current national and global scale monetary valuation of pollination services do not accurately estimate the contribution of wild pollinators to agricultural production. First, ecosystem (wild) pollination services remain largely bundled with those of managed pollinators. This problem is compounded by the fact that the dependency of crops on pollination, a key parameter used in current valuations, do not reflect variation in pollinator density, crop cultivars and growing conditions that exist in practice. Over half of the €153 billion of estimated global pollination service value in 2005 is based on estimates of pollinator dependency from crops with fewer than three field studies that measure actual levels of pollinator activity and corresponding fruit set. The resulting uncertainty may be most distorting when applied to widely-planted intensive oilseed crops. Furthermore, current valuations are underpinned by simplistic assumptions regarding the likelihood of pollinator decline and the impact on agricultural prices. Although efforts to motivate wild pollinator protection through their ecosystem service value remain highly circumscribed by conceptual and empirical limitations, I identify the need to go beyond technical solutions and develop a critical framework that could

account for why pollinator conservation has come to be predominantly justified in these terms to begin with.

2.2 INTRODUCTION

Evidence has emerged of a global decline in a number of wild animal species that pollinate angiosperm plants (reviewed in Potts et al., 2010) even though conservation initiatives appear to offer potential for reversing this trend (Carvalheiro et al., 2013). A prominent approach to encourage the conservation of these species has been to calculate the economic value (hereafter simply termed "value" unless otherwise characterized) they contribute to the production of pollinator-dependent agricultural crops. It is estimated that in 2005 €153 billion (Gallai et al., 2009) of global food production would be lost if pollinating insects disappeared. Furthermore, it is asserted that the value of insect pollinators to global crop production is rapidly rising (Lautenbach et al., 2012).

Efforts to estimate the value of ecosystem services provided by wild species to human well-being originates in the 1980s and has grown into a widely asserted justification for international conservation initiatives (Gómez-Baggethun and Ruiz-Pérez, 2011). Underpinning the ecosystem service approach is the assumption that the diversity and density of wild pollinator populations surrounding agricultural land are key to current levels of pollinator-dependent seed and fruit crop yield (Garibaldi et al., 2013; Greenleaf and Kremen, 2006; Hoehn et al., 2008; Winfree et al., 2011). By quantifying the value attributable to current crop yields it is hoped that pollinator conservation will be encouraged, thus overcoming the traditional opposition between economic imperatives and conservation (Armsworth et al., 2007). In other words, it is anticipated that if the value of wild pollinator services is demonstrated, farmers, land managers and the public at large will be motivated to protect pollinator habitat and pressure policy-makers to implement agri-environmental programs that meaningfully target pollinators, and by extension, more generalized conservation (reviewed in Winfree, 2010).

In this chapter I argue that our capacity to judge the risk of lost agricultural value due to wild pollinator loss is severely impaired by our current inability to gauge the magnitude and importance of the services being provided by wild species to current production. I will demonstrate substantial problems with both the methods and data used to quantify the value of pollinators, specifically:

- The actual dependency of yield on the activity of insect pollinators, D, is frequently unknown (Assumption 1, Figure 2.1).
- The proportion of this pollination conducted by managed pollinators, ρ, and by extension that conducted by wild pollinators, is assumed but rarely assessed (Assumption 2).
- That both D and ρ ought to be considered as dynamic and context dependent, but are treated as static constants that apply generally regardless of variation in local growing conditions.
- It is assumed that the loss of pollinators will decrease agricultural value, but the responses of consumers, farmers, technological innovation and government policy to price signals arising from potential pollinator declines are poorly understood, as are

the actual risks of pollinator decline impacting price in the first instance (Figure 2.1 - Assumption 3).

A consequence of not recognizing these deficiencies may be a false sense of certainty that continued reporting of decline in wild pollinator populations will motivate pollinator conservation efforts. Although there has been much speculation about growing food insecurity associated with pollinator decline (Aizen et al., 2009; Breeze et al., 2014b; Eilers et al., 2011; Gallai et al., 2009; Garibaldi et al., 2011a), I will argue that the relationship remains difficult to judge given the substantial uncertainties underpinning current estimates of value.

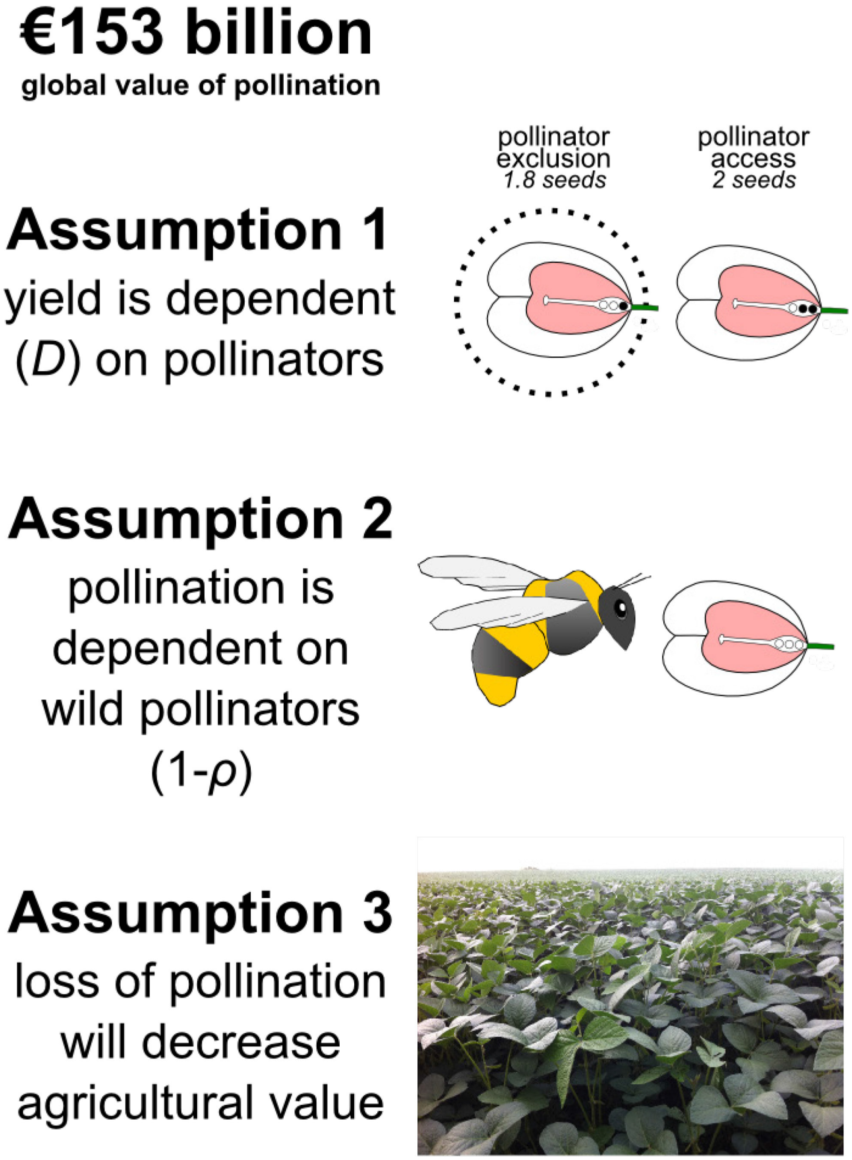

Figure 2.1 The key assumptions behind the value of insect pollination methodology(*IPEV*) technique to estimate the value of wild pollinators in relation to the €153 billion of value annually generated by insect pollinators (Gallai et al., 2009). The assumptions are illustrated in this figure for a single crop, soybean. The estimated dependency (*D*) of soybean on insect pollinators of 0.1 (Calderone, 2012) is depicted as a reduction of 0.2 seeds per plant when flowers are excluded to pollinators.

2.3 SO MUCH *DEPENDS* ON DEPENDENCY (*D*) (ASSUMPTION 1)

2.3.1 The history of calculating pollination value

The first pollinator valuations were published in the 1940s (Table 2.1), predating the concept of ecosystem services by almost forty years (Gómez-Baggethun et al., 2010). These followed the publication of broad compendia of the specific pollination requirements of different agricultural crops (Butler and Simpson, 1954; Free, 1960; Free, 1993; McGregor, 1976) and, as such, were not motivated by pollinator conservation concern, but rather interest in maximizing crop yields. Moreover, the focus of early valuations was almost exclusively on managed honey bees and on crops grown in Europe (Butler, 1943) and North America (Levin, 1984; Martin, 1973; Metcalf et al., 1962; Winston and Scott-Dupree, 1984).

Initially, value was crudely calculated as the total economic value (*EV*) of all crops in which at least some yield, however minimal, was facilitated by insect pollinators (Figure 2.2, Table 2.1). *EV* was the basis for all national-level valuations until 1987 and global valuations until 2009. Using this method the value of soybean pollination in the U.S. in 2010 would be assessed as $38.9 billion (Figure 2.2), or the *entire* value of the crop, even though the bulk of the pollination of this crop occurs through self-fertilization without insects (Free, 1993).

It is also worth noting that in some valuation estimates *EV* not only included crops directly utilized by humans, but also a portion (Levin, 1984; Winston and Scott-Dupree,

1984) or all of the value (Martin, 1973) of livestock meat and dairy production. The inclusion of livestock was justified on the grounds that they rely, in part, on leguminous crops, such as clovers and alfalfa, and the production of their seeds is facilitated, to one extent or another, by insect mediated pollination. Where livestock production value has been included in analyses, it represents the largest contributor to the total calculated value. While subsequent approaches have uniformly dropped livestock value from their calculations, debates over what to include or exclude in the scope of a valuation exercise have recently recurred in the context of oilseed-based bio-diesel production (Chacoff et al., 2010; Gallai et al., 2009).

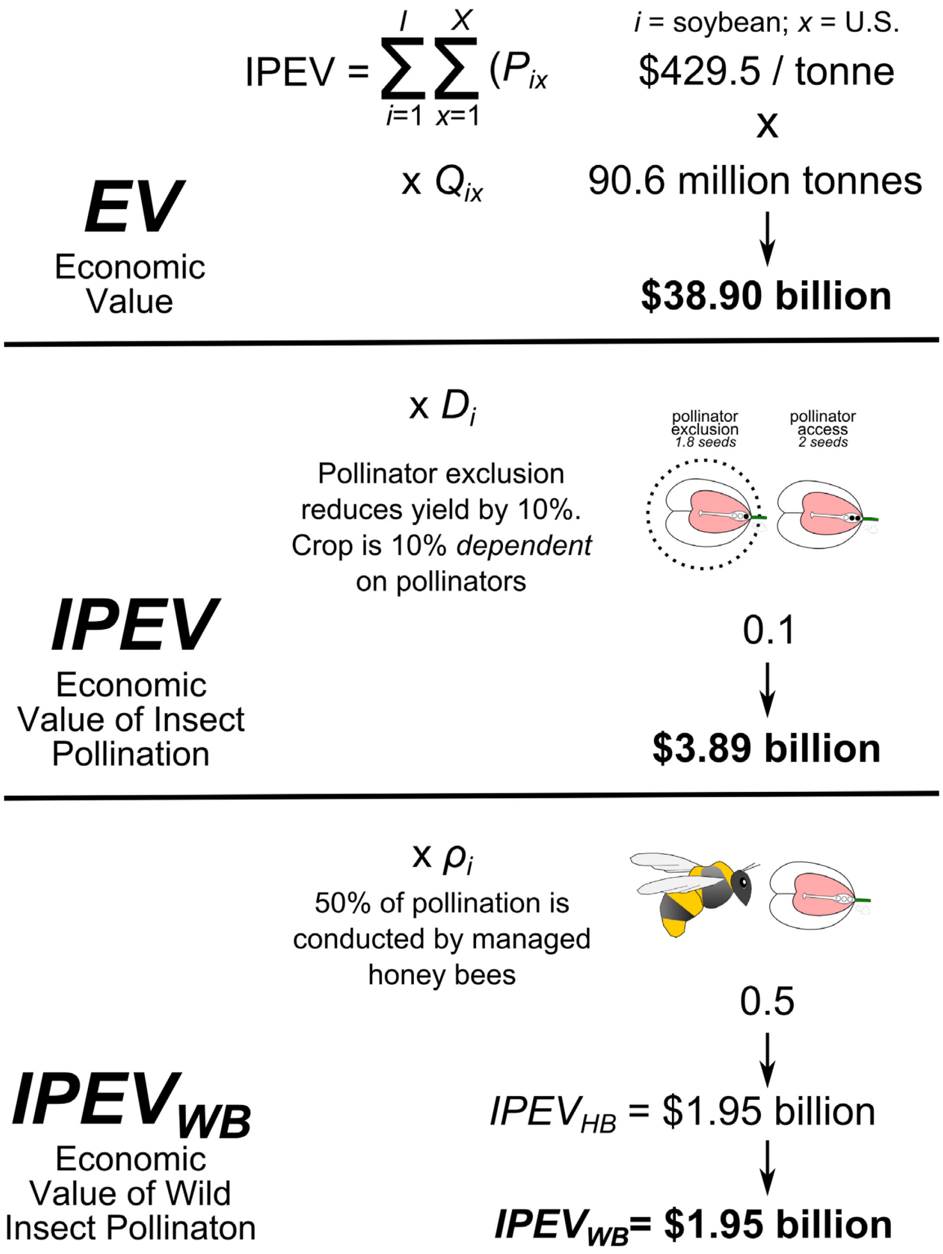

Figure 2.2 Different approaches for calculating the value of pollinators to agriculture using the example of soybean in the U.S. from 2010. For each region, *x*, included in the valuation, the quantity (*Q*) for each crop *i* produced was related to the dependency of that crop on pollination (*D*), as well as the proportion of pollination performed by honey bees (ρ). Values for *P*, *Q*, *D* and ρ from a U.S. valuation (Calderone, 2012).

Table 2.1 Valuations of pollination services

Method valuing pollinator contribution to crop value	Date of first use	Calculation[a]	No. studies that differentiate between wild and managed pollinators (total no. studies in brackets)	References	Key limitation
Production value 1. Economic value (*EV*)	1943	QP	0 (6)	(Butler, 1943; Fluri and Frick, 2005; Levin, 1984; Martin, 1973; Metcalf et al., 1962; Winston and Scott-Dupree, 1984)	Dependency of all crops assumed to be 100%, no differentiation between pollination by wild versus managed species, assumed perfectly elastic demand and cost of other inputs not accounted for.
Production value 2. Total economic value of insect pollination (*IPEV*)	1989	$QPD(\rho)$	8 (13)	(Ashworth et al., 2009; Calderone, 2012; Chacoff et al., 2010; Gallai et al., 2009; Kasina et al., 2009a; Lautenbach et al., 2012; Losey and Vaughan, 2006; Morse and Calderone, 2000; O'Grady, 1987; Robinson et al., 1989; Southwick and Southwick Jr, 1992; Winfree et al., 2011)	Dependency of all crops based on some empirical research, but frequently not extensive, frequently an arbitrary differentiation between pollination by wild versus managed species (but see Winfree et al., 2011), demand assumed perfectly elastic (with some exceptions (see Gallai et al., 2009; Southwick and Southwick Jr, 1992)) and cost of other inputs not accounted for (but see Winfree et al., 2011).
Replacement value	1995	Replacement of all pollination by labour or wild pollinators by managed bees.	2 (3)	(Allsopp et al., 2008; Mouton, 2011; Muth and Thurman, 1995)	The costs of replacing pollination with human labour have been shown to exceed total production value for the crop and, hence, not a suitable substitute (Allsopp et al., 2008). The shortfall in published managed bee stocking rates is limited to crops where managed pollinators are used and data is reliably collected.

[a] Q = quantity of crop grown, P = crop price, D = dependence of crop on animal-mediated pollination (see text)

Table 2.1 *continued...*

Method valuing pollinator contribution to crop value	Date of first use	Calculation[a]	No. studies that differentiate between wild and managed pollinators (total no. studies in brackets)	References	Key limitation
Contingent valuation method	2010	Willingness to pay for wild pollinator protection	1 (1)	(Mwebaze et al., 2010)	Evidence suggests that contingent valuation methods do not correspond to the actual amounts spent on conservation (Pearce, 2007; Vatn and Bromley, 1994).
Landscape service flows	2004	Relate landscape patterns to bee diversity and abundance and crop yields	4 (4)	(Chaplin-Kramer et al., 2011; Morandin and Winston, 2006; Olschewski et al., 2006; Ricketts et al., 2004)	Empirically robust measure of actual flows but limited by previous approaches with regards to being integrated with agricultural economic approaches.

[a] Q = quantity of crop grown, P = crop price, D = dependence of crop on animal-mediated pollination (see text)

2.3.2 Estimating insect pollination economic value (*IPEV*)

O'Grady (1987) proposed the Insect Pollination Economic Value (*IPEV*) methodology as a way to overcome problems associated with the *EV* methodology (Figure 2.2). It involved multiplying a crop specific coefficient of pollinator dependency (*D*) for each crop by the total value of the crop (*P* x *Q*) (Figure 2.2). Returning to our example of soybean in the U.S. (Figure 2.2) this approach takes the \$38.9 billion and allocates 10% ($D = 0.1$) of it to insect pollination, thus reducing the apparent value of the pollinator service flow to \$3.89 billion. *IPEV* currently is the basis of the most recent national (Ashworth et al., 2009; Calderone, 2012; Chacoff et al., 2010) and global valuations (Lautenbach et al., 2012) (Table 2.1).

In O'Grady's (1987) original formulation, crops were assigned one of three values for *D* depending on whether the crop could absolutely not set fruit without insect pollination (high dependency = 0.9), fruit set was enhanced by insect pollination (medium dependency = 0.5) or where fruit set is only slightly enhanced by insect pollination (low dependency = 0.1). In other words, the dependency of the crop was qualitatively deduced from the pollination biology of each species. Robinson et al. (1989) modified this approach by proposing that, where data existed, to calculate *D* from studies comparing fruit set under conditions of pollination exclusion (f_{pe}) to that of open pollination (f_p) or, where possible, under conditions of saturated pollination in which bees were stocked within exclusion cages (f_{pmax}, corresponding to the calculation of D_{max}). These parameters are used to determine the extent to which fruit set would decrease if pollinators were completely absent, using the equation:

$$D\ (or\ D_{max}) = 1 - \frac{f_{pe}}{f_p\ (or\ f_{pmax})}$$

It is important to note the implications of using studies measuring f_p versus f_{pmax} in this calculation. Using f_{pmax} values, or the yield resulting from artificial pollinator saturation studies, to calculate D_{max} estimates the *potential dependency* of a crop. The *IPEV* calculation, however, purports to calculate the *actual* or *realized value* of pollination in agriculture in a given year, or what the accompanying loss in *current levels* of agricultural value would be if the *current density* of pollinators active in these crops were drawn down to zero. Consequently, *IPEV* using D_{max} would have to be corrected to reflect real-world pollinator densities. In other words, in the absence of any correction for pollinator density, only experiments that measure f_p, or the yield realized in open pollinated plots, estimate the *realized dependency* (D), or the *dependency of actual crop value* (P x Q) on pollination. I can readily envision scenarios, for example, in which D would vary significantly simply by moving honey bee colonies into and out of the fields, effectively raising and lowering the differential between f_p and f_{pe}. To date, however, corrections for pollinator densities have either not been used (Ashworth et al., 2009; Chacoff et al., 2010; Gallai et al., 2009; Kasina et al., 2009a; Lautenbach et al., 2012) or applied coarsely using the strategy introduced by Robinson et al. (1989) in which value is uniformly adjusted by a factor of 0.1 if the crop routinely employs rented honey bee colonies (Calderone, 2012; Losey and Vaughan, 2006; Morse and Calderone, 2000). I will discuss a notable exception to this procedure in Section 2.3.3 (Chaplin-Kramer et al., 2011), but given the general practice of assuming pollinator densities to be static I proceed in the next section to examine to what extent f_p have been used to determine D in valuation. I discuss the consequences of not restricting the scope of studies to only those measuring f_p in Section 2.3.4.

2.3.3 Problems with estimating dependency on pollinators (D)

It is important to note that all subsequent valuations since Gallai et al. (2009), with the exception of efforts from the US that continue to use values derived by Robinson et al. (1989), have come to rely on values of D derived from a study by Klein et al. (2007). Yet Klein and colleagues (2007) did not assemble values of D for the purpose of economic valuation, but rather to determine the proportion of agricultural yield that was dependent on the activity of pollinators, as well as the degree of this dependence. This involved reviewing close to 400 pollination studies across 178 major crops or cropping groups and classifying each into broad categories of pollinator dependence (*essential* ($D > 0.9$), *high* (D = 0.9-0.4), *modest* (D = 0.4-0.1), *little* ($D < 0.1$) and *no increase* ($D = 0$)). Gallai et al. (2009) adapted this classification by taking the average of the highest and lowest value for the estimate of D for each crop (ie *essential* ($D = 0.95$), *high* ($D = 0.65$), *modest* ($D = 0.25$), *little* ($D = 0.05$)) as reported by Klein et al. (2007).

In order to determine the extent to which the studies comparing f_p and f_{pe} were used in the Gallai et al. (2009) valuation, I tallied the number of studies used to estimate D for the 46 crops and crop groups reported by Klein et al. (2007) in their electronic supplementary material and compared them to the value calculated by Gallai et al. (2009). In doing this I excluded citations that were not primary research as well as older reviews, notably Samuel McGregor's *Insect Pollination of Cultivated Crop Plants* (1976) or John Free's *Insect Pollination of Crops* (1993). Both Klein et al. (2007) and Gallai et al. (2009) recognized the limitations of these older reviews given the significant change in crop varieties and cropping systems since their publication.

There are very few studies, in fact, that directly measure f_p and f_{pe} for the crops that constitute the bulk of apparent value generated by insect pollination (Figure 2.3). Nearly half of the global value, or €74 billion, is derived from crops for which D is estimated from fewer than three studies that actually measure these parameters. The sparseness of these data is more apparent when considering only the crops with the ten highest pollination values, as defined by Gallai et al. (2009) (Table 2.2). While these crops contribute almost 65% of the total apparent value of pollination services globally, most of the studies used in support of their value calculation rely on experiments with a single cultivar, grown in a single field and assessed across only one season. Yet, pollinator-mediated yield is strongly conditioned by cultivar and their spatial arrangements in fields (Bell et al., 2012; Klatt et al., 2014), as well as environmental conditions and farm management practices (Boreux et al., 2013; Bos et al., 2007a; Bos et al., 2007b; Groeneveld et al., 2010; Hoover et al., 2012; Lundin et al., 2013; Melathopoulos et al., 2014; Racskó et al., 2007), suggesting current estimates of D largely do not account for variation within each crop to carry fruit and seeds to harvest. This points to a more profound problem; in the *IPEV* calculation D is regarded as a static parameter that is unresponsive to variation within and among fields, between regions, years and, importantly, in pollinator diversity and density. For *IPEV* to become a more robust methodology, pollinator exclusion experiments would need to be replicated temporally and spatially across crop growing regions and under a range of conditions.

The problem of inadequate replication in estimating D can be drawn to an even finer point when considering the type of studies cited by Klein et al. (2007). Given the global scope of their undertaking they included a significant number of studies that could be used to qualitatively assign a crop to different dependency categories. These include studies measuring density-dependent yield responses to pollinators (e.g. Stern et al., 2001), yield-increases resulting from the enclosure of crops with pollinators (D_{max}) (e.g. Chiari et al., 2005), pollination deficit studies (e.g. Morandin and Winston, 2005), surveys of varietal outcrossing (e.g. Kron et al., 2001), assessments of the pollination efficiency of different pollinator taxa (e.g. Stanghellini et al., 2002), and descriptions of pollinator fauna life-history associated with crop plants (e.g. Meléndez-Ramírez et al., 2004; e.g. Sakai et al., 2000). In fact, for eight of the top ten crops with the highest aggregate global value, only one or zero studies that directly measured f_p and f_{pe} were identified. The estimate for the most studied crop, soybean, ends up being based on data from four fields, using a total of three cultivars, in two countries (Chiari et al., 2005; Moreti et al., 2008), with one study conclusively demonstrating no difference between f_p and f_{pe} (Koelling et al., 1981).

To be clear, Klein et al. (2007) were aware of the limitations of the data they drew their estimates of D from and this is reflected in the very broad ranges encompassed in the two middle dependency categories, namely *high* (D = 0.9-0.4) and *modest* (D = 0.4-0.1). While the subsequent use of these categories for the purpose of estimating pollinator service value is sometimes accompanied by warnings of the potential inaccuracies in estimating *D* (e.g. Calderone, 2012; Gallai et al., 2009) these efforts do not use the upper and lower bounds of *D* reported by Klein et al. (2007), but rather mid-point values. To my knowledge only Chaplin-Kramer (2011) and Lautenbach (2012) have attempted to reflect the range of uncertainty and natural variability by reporting values estimated using the original upper and lower bounds of *D* as reported by Klein et al. (2007).

Despite the prominence of recent pollination valuation efforts, there has not been a complementary effort to systematically generate new and robust values of *D* for various crops from which objective estimates of global *IPEV* could then be rendered. There have only been small efforts towards this end, such as a valuation of wild bee pollination services in western Kenya, in which the authors base their estimates of *D* on pollinator-excluded and open pollinated plots across the region (Kasina et al., 2009a).

There are more nuanced problems associated with the use of *D* in *IPEV* calculations. For example, current approaches to estimating *D* only reflect differences in yield between pollinator exclusion and open pollination scenarios. This method reflects quantitative differences in yield between the condition of pollinator abundance and dearth, such as fruit set or total yield, but does not account for qualitative differences in the harvested crop, such as visual appearance, palatability or nutritional composition. Although two recent studies have proposed methods for incorporating qualitative dimensions into the *IPEV* methodology for apples (Garratt et al., 2014) and strawberries (Klatt et al., 2014), research along this vein simply does not exist for most crops.

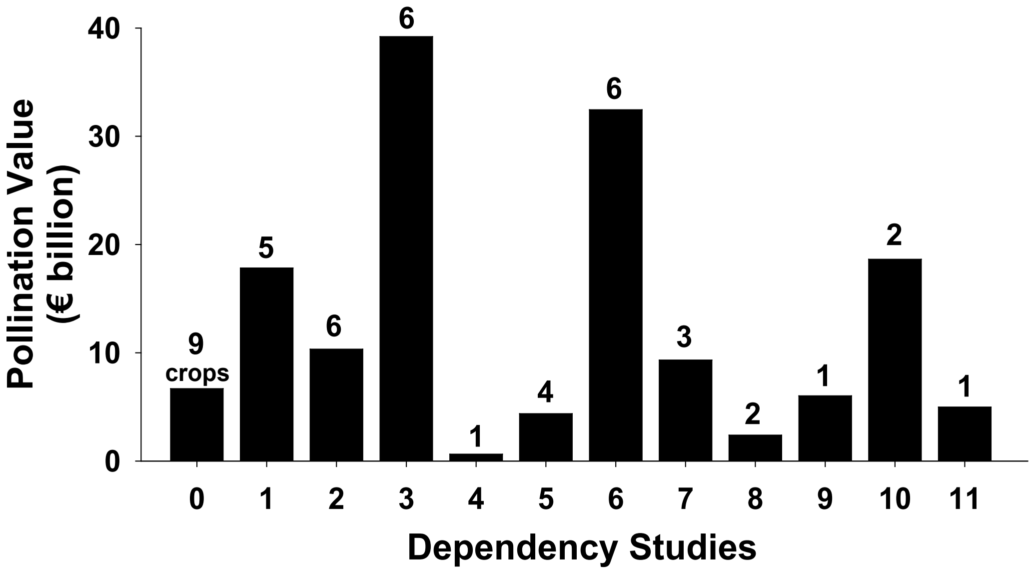

Figure 2.3 The total value of pollination of 46 crops grouped by the number of dependency studies used to establish these values in the most recent global estimate of €153 billion calculated by Gallai et al. (2009) with dependency values derived from Klein et al. (2007). These totals do not include references that were not original research, namely review papers or studies with methodologies that do not directly estimate dependency. The numbers of crops that are included in the summed value for each citation category appear above each bar.

Table 2.2 Uncertainty surrounding the calculation for dependency using figures from Klein et al. (2007) for the 10 highest valued pollination services in the most recent global valuation of pollination services by Gallai et al (2009).

Crop	Value of Pollination (€ billions)	No. studies demonstrating dependency[1]	No. studies included that do not directly measure dependency[1]	Total[2]	Total no. cultivars represented[3]	Total no. fields represented[3]	No. countries studies involve
Apple	15.72	1	7	11	1	1	1
Cucumbers and gherkins	15.35	0	1	1	-	-	-
Watermelons	14.48	2	0	7	1[4]	1	1
Soybean	10.88	2	1	6	4	3	2
Cottonseed	10.05	0	1	1	-	-	-
Guavas, mangoes, and mangosteens	9.05	0	1	3	-	-	-
Coconuts (incl. Copra)	6.67	1	1	2	na	1	1
Other melons (incl. cantaloupes)	6.23	0	1	2	-	-	-
Pears and quinces	6.01	1	2	3	2	2	1
Peaches	5.00	1	1	1	1	1	1

[1] – For definition of dependency see text. Common types of studies that did not fit this category were studies of *pollinator density dependence*, where researchers controlled the number of pollinators in plots, *pollinator efficacy*, where the relative efficacy of different pollinator species was compared and basic surveys of pollinators found in each crop.

[2] – The total number of cited studies may not add up to the first three categories because some studies either did not provide sufficient detail to conclusively categorize the methods as measuring dependency (i.e., measuring f_p and f_{pe}), or else the study was inaccessible.

[3] – Totals are for dependency studies only.

[4] – The cultivar was not specified.

2.3.4 Implications of unreliable estimates of *D*: The example of oilseeds with low *D*

The problems associated with current estimates of *D* are most pronounced with crops like soybean or oilseed rape, both intensive, widely-planted, mass flowering crops (Holzschuh et al., 2011) that have low estimated values of *D*: soybean $D = 0.25$ (Gallai et al., 2009) to 0.10 (Calderone, 2012); oilseed rape $D = 0.50$ (Calderone, 2012) to 0.20 (Gallai et al., 2009). The massive areas planted to these crops means they dominate estimates of *IPEV* in regions with intensive agriculture such as North America (Calderone, 2012; Morse and Calderone, 2000; Scott-Dupree, 1995), the EU (Breeze et al., 2011; Breeze et al., 2014b; Schulp et al., 2014) and Argentina (Chacoff et al., 2010). For this reason, soybean is ranked the fourth highest source of pollination value globally (Gallai et al., 2009). Consequently even small variations in *D* for these crops distort estimates of pollination value at national and global scales.

I speculate that the current value attributed to these crops is considerably overestimated given the evidence from the EU that there appears to be low densities of managed (Breeze et al., 2014b) and wild pollinators (Schulp et al., 2014) in areas where these crops are grown. As I argued in Section 2.3.2, where pollinator density is low, yield from open-pollinated plots (f_p) would approach that of plots with pollinator-exclusion (f_{pe}), essentially drawing the value of *D* towards zero. In other words, as pollinator density declines, so ought the value that pollinators contribute to output. While this may seem counter-intuitive, it is important to recall that *IPEV* estimates the value that would be lost if pollinators were to approach zero. In fact, in landscapes with already low densities of pollinators, yield losses associated with pollinator disappearance may be slim. I predict that actual measurements of f_p and f_{pe} would demonstrate that the production of these crops exists with near zero pollination value across most of their growing range.

If my prediction is correct it would significantly undermine forecasts that agricultural value will be lost due to the decline in managed and wild pollinator populations relative to the area planted to these crops (Aizen et al., 2008, 2009; Breeze et al., 2011; Breeze et al., 2014b; Garibaldi et al., 2011a; Schulp et al., 2014). All such forecasts fail to consider that current oilseed production may very well be produced under conditions of pollination deficit (Morandin and Winston, 2005). Consequently, the apparent lag in the growth of EU honey bee stocks following the 2003 biofuel standards, and subsequent growth in area planted to oilseeds (Breeze et al., 2014b), may be largely overstated. While the 4.2 M ha (32%) expansion of these crops since 2003 was only attended by a 7% increase in honey bee colony stocks, the implication of a "mismatch" between the two remains abstract without data of a *growing* pollination deficit on the ground. A similar mismatch was suggested for wild pollinators (Schulp et al., 2014) by virtue of the fact that 51% of EU cropland that was classified as having high pollinator dependency had a low supply of wild pollinator habitat.

Whether or not a mismatch actually exists has implications for pollinator conservation policy. In the EU the assessment of the high value of pollinators to oilseed production has drawn a strong focus on biofuel policy, namely to call for investment into pollinator habitat "green infrastructure" in areas where oilseeds are expected to be planted

(Breeze et al., 2014b; Schulp et al., 2014). Leaving aside the question of whether conservation strategies targeting highly intensified agricultural landscapes is the best use of conservation resources (Kennedy et al., 2013; Kleijn et al., 2011; Phalan et al., 2014), there are clear problems associated with the expectation that investments in "green infrastructure" will assist with the future retention of oilseed value. Breeze et al. (2011) assumed pollinator densities in these crops were once close to the recommended 1 (soybean) and 2.8 (oilseed rape) colonies ha^{-1}, but there no evidence for this and growers of these crops have historically not seen the need to create a market for managed pollinator stocks. Current cropping may have long operated under conditions of low pollinator densities and may be relatively insensitive to further pollinator decline, in other words at a ratio of f_p to f_{pe} approaching 1. The vast areas planted to these crops, even before the recent increases affected by the biofuel directive (13.8 M ha in 2005), would mean that to come close to these recommended stocking rates virtually every one of the 22.5 million honey bee colonies in the zone would have been deployed in the service of pollinating these crops. Given that such stocking rates do not exist in practice Breeze et al. (2011) suggest that wild pollinators may be delivering a considerable amount of unaccounted for pollination services. Yet this observation seems at odds with the fact that these crops are grown in highly intensified landscapes that are generally not conducive to robust wild pollinator populations (Schulp et al., 2014).

If pollinator stocking densities in biofuel crops are well below recommended rates, and values of D over-estimate the pollinator-dependent value of these crops, the deficit between pollinator densities and pollinator-dependent yield may be more an artifact of flawed assumptions than an actual biophysical phenomena. It is telling that despite such an apparent "mismatch" there is little evidence of lower yields (Aizen et al., 2008, 2009; Breeze et al., 2011; Breeze et al., 2014b; Ghazoul and Koh, 2010). Furthermore, the large weight attributed to pollinator value among crops such as soybean and oilseed rape risks distracting attention from cropping systems where current yield is potentially more reliant on the activity of wild pollinators. For example, given that most of the apparent value of wild pollinators in the U.S. resides in soybean (Calderone, 2012, see Section 2.3.1), it may become harder to argue for the need to focus pollinator habitat restoration around crops such as melons, berries and tree fruit, with ostensibly "lower" overall pollination value despite demonstrably higher levels of *D*. Yet, in an exhaustive meta-analysis of 41 cropping systems undertaken to demonstrate the superior capacity of wild pollinators to increase yield compared to honey bees, no studies were identified specifically for soybean (Garibaldi et al., 2013). That is to say the real “mismatch” may be between crops with the largest aggregate (though likely substantially overstated) *IPEV* and the crops ecologists deem most important from the perspective of yield dependence on wild pollinators.

2.4 SEEN ONE POLLINATOR SEEN ‘EM ALL (ASSUMPTION 2)

Calculating the value of agricultural insect pollination entails a unique set of problems not faced with many other ecosystem services. Most notably, managed bee species, like the domesticated Western honey bee (*Apis mellifera*), alfalfa leafcutter bee (*Megachile rotundata*) (Pitts-Singer and Cane, 2011), and some species of bumble bee

(*Bombus* spp.) (Velthuis and van Doorn, 2006), provide substantial amounts of agricultural pollination. While managed species may depend on natural and semi-natural habitat, which has prompted at least one author to classify them as an indirect ecosystem service (Mouton, 2011), they more closely resemble an agricultural input. These managed species are often non-native, have only a transient dependence on the habitat surrounding the fields they pollinate, and instead are highly dependent on manufactured inputs such as sucrose, processed plant proteins, pesticides, antibiotics, artificial nest structures, and substantial physical infrastructure (Kendall et al., 2013; Southwick, 1980; Southwick and Pimentel, 1981). This allows these species to function in highly intensified agro-ecological landscapes that would otherwise not support comparable levels of wild ecosystem pollination service flows (Ghazoul, 2007). Furthermore, the presence of managed pollinators can have negative implications for the viability of wild populations (Colla and Packer, 2008; Fürst et al., 2014; Goulson, 2010; Lye et al., 2011).

The problems associated with partitioning managed from wild pollination services has meant that the latter are poorly defined in valuation estimates. So far as I can tell, all global valuation of these services, from Costanza et al.'s (1997) widely-cited accounting of the value of global natural capital to the most recent attempts (Gallai et al., 2009; Lautenbach et al., 2012), fail to separate these two, very distinct, service flows. While many national-scale pollination valuations have attempted to separate honey bee from other pollination services, these have largely been motivated by efforts to determine the contribution of honey bees rather than wild pollinators (for an exception see Chaplin-Kramer et al., 2011). Moreover, there appears to be an important, yet often obscured, bifurcation of research goals inherent in various valuation studies between scientists working with managed pollinator species and those interested in the conservation of wild species. This division is evident in a recent exchange of letters (Aebi et al., 2012; Ollerton et al., 2012) that echoes an exchange that took place twenty years earlier in the same journal (Corbet, 1991; Morse, 1991), in which the overwhelming importance of honey bees to agricultural production is asserted by apicultural researchers against ecologists and entomologists who argue that wild species represent a significant source of agricultural pollination. The goals of these two groups may be incompatible. Valuation studies undertaken by apicultural researchers have historically been used to argue for the need to strengthen subsidies to beekeepers or government programs associated with apiculture (Muth and Thurman, 1995; Robinson et al., 1989). These programs have largely looked to stabilize beekeeper incomes or lower their cost of production as opposed to preserving or expanding pollinator habitat. Given that wild pollinator declines may lead to a growing dependence of agriculture on honey bee colony rentals, the interests of beekeeping firms might be rationally aligned with landscape practices that increase yields at the expense of wild pollination services. This may mean that the proliferation of methodologies proposed to measure pollination ecosystem services (Liss et al., 2013) may be motivated by more than simply the idiosyncrasies of researchers; they may reflect conflicting interests among pollinator service providers.

2.4.1 Partitioning *IPEV* between wild and managed pollinators using ρ

From the perspective of wild pollinator conservation perhaps the biggest problem with *IPEV* is the thorny issue of partitioning the role of wild versus managed pollinators. O'Grady (1987) proposed a solution to this problem early on, by the inclusion of a coefficient for the proportion of pollination conducted by honey bees (ρ) (Figure 2.2). This is demonstrated by returning to the soybean example from the last section. Including ρ further divides the \$3.89 billion of apparent value generated by insects into \$1.95 billion from honey bees (ρ=0.5) and \$1.95 billion by other insects. Although ρ is not included in recent global valuations (Gallai et al., 2009; Lautenbach et al., 2012), it has remained a component of all the U.S. valuations since 1985 (Robinson et al., 1989). Until recently (Losey and Vaughan, 2006) this was explicitly motivated by the focus on specifying the value of honey bees (Calderone, 2012; Morse and Calderone, 2000). This approach has also been used in other national valuation studies (Chacoff et al., 2010; Gibbs and Muirhead, 1998; Scott-Dupree, 1995). Since the contribution of wild pollinators is arguably less understood than the crop pollinator dependency (*D*), these calculations have assumed a relatively constant proportion of pollination is being conducted by managed honey bees, typically 80-100%.

There are some notable exceptions to the 80-100% rule acknowledged in valuation studies. Two high value crops, pumpkin and squash, were reassessed beginning with the 2000 U.S. valuation (Calderone, 2012; Morse and Calderone, 2000) as being 90% pollinated by wild bee taxa, notably by the squash bee (*Peponapis pruinosa* Say) a specialized pollinator of these crops (Hurd et al., 1974). Other exceptions that have been carried forward since the 1985 US valuation (Robinson et al., 1989) are alfalfa ($\rho = 0.6$) and, as previously mentioned, soybean ($\rho = 0.5$). Alfalfa is mainly pollinated with managed alfalfa leafcutter bees, so objectively the remaining 40% of its pollination is attributed to a managed pollinator (Losey and Vaughan, 2006).

The soybean ρ value is more difficult to explain. To our knowledge there has never been a study of the role of wild bee pollination in soybean in the U.S. Yet, given the vast area planted to this crop in that country, it becomes the single largest source of apparent value generated by wild bees, delivering an estimated $1.95 billion in 2010 (Calderone, 2012). This is larger than the total estimated value delivered by wild pollinators to the all other crops in the country *combined* ($1.51 billion).

The problem is that the proportion of yield increase attributable to wild and managed pollinators is not connected to any meaningful research on the activity of wild bees in these systems. What needs to be acknowledged is that our understanding of the proportion of pollination conducted by wild or managed species is based on an even narrower set of empirical data than current estimates of D. The relative contribution of wild pollinators can vary considerably even within the same crop and within the same region. For example, while honey bees were determined to be responsible for 88% of the aggregate pollinator-dependent increase in Michigan blueberry production, this was largely the effect of a disproportionate amount of production taking place on large fields (mean field size = 2.30 ha) (Isaacs and Kirk, 2010). In contrast, on small fields (mean field size = 0.02 ha) wild bees accounted for the majority of yield increases. A study from California on watermelons (Kremen et al., 2002) confirms the contextual basis of wild versus managed pollinator dependence. Here, native bees were able to provide all the pollination when fields were near uncultivated land whereas pollination was increasingly dependent on honey bees at locations at greater distance from such land. Clearly, generating robust estimates of ρ mirrors the challenge of estimating D given the enormous variation in relative and absolute numbers of managed and wild pollinators that may be available to service any one crop and in any given field.

2.4.2 Better approaches to value the activity of wild pollinators to crop yield?

Despite the quantification challenges, wild bees are undoubtedly providing significant value to agriculture, but perhaps not in the ways that current valuation efforts track. There is mounting evidence that wild bees are generally more effective pollinators than honey bees and their density on flowers is better related to crop yield (Garibaldi et al., 2013). There is also evidence of interaction effects, in which the pollination efficacy of honey bees is enhanced greatly by the presence of high densities of wild bees (Brittain et al., 2013a; Greenleaf and Kremen, 2006). Furthermore, in a number of commercial cropping systems, wild bee densities seem sufficiently high to render the rental of honey bee colonies superfluous (Eaton and Nams, 2012; Petersen et al., 2013; Winfree et al., 2007; Winfree et al., 2008). Despite these demonstrated benefits, the value of wild bees to agriculture, and their value relative to managed honey bees, is simply not reflected in current valuation estimates at either national or global scales.

One effort to address this problem is a revision to the basic *IPEV* calculation proposed by Winfree et al. (2011) that incorporates the observed proportions of wild and managed bees. This work is additionally remarkable as it incorporates a parameter that accounts for how much pollination can take place before yield reaches a maximum, in other words, where additional pollen deposition no longer increases yield. In doing so it corrects for the diminishing marginal value of additional managed pollinators (Muth and Thurman, 1995). Put another way, if wild pollinators are already present in a field the value they provide should be more than the additional yield increments that result from managed bees brought into the field later, assuming, that all bees are equally effective pollinators. The implications for valuing ecosystem services should be clear, especially in examples such as in New Jersey and Pennsylvania watermelon production where many of the 23 fields under study were found to be over-pollinated (Winfree et al., 2011). If the incremental value of additional honey bee pollination activity was factored in *after* the existing service flow from wild pollinators was accounted for, it resulted in wild pollinators contributing 14 times the value of managed pollinators, compared to just 1.6 times as calculated using the older *IPEV* method. The importance of marginal pollination effects has also been recently elaborated in another framework (Simpson, 2013).

Yet the key limitation that arises with the *IPEV* approach and modifications as proposed by Winfree et al. (2011) seems more than a problem of having the right framework; it is the problem of generating a sufficient number of datasets of wild versus managed pollinators to adequately represent the spectrum of crops, cropping systems and growing regions. This problem is compounded by the fact that existing datasets have attempted to measure pollination service flow in a myriad of ways (Liss et al., 2013). The most promising solution to this obstacle comes from recent attempts to estimate the "supply" of wild pollinators by assessing the quality of pollinator habitat surrounding agricultural land as a proxy. Although this technique was initially restricted to regional assessments of pollination value for single crop species, largely coffee (Jha and Dick, 2010; Olschewski et al., 2006; Priess et al., 2007; Ricketts et al., 2004; Scullion et al., 2011; Veddeler et al., 2008), it has been recently applied to a multi-crop valuation for the state of California (Chaplin-Kramer et al., 2011). In this latter effort, *IPEV* values were adjusted relative to proximity of wild bee habitat to each crop using a pollination service index (PSI). Moreover, unlike ρ, the PSI is based on an empirically derived relationship between the spatial distribution of wild bee habitat and the deposition of pollen for a crop with relatively high requirements for pollen deposition (e.g. watermelon). This approach would certainly resolve many of the problems described in Section 2.4.1.

Operationalizing the proximity to habitat approach for widely grown mass-flowering crops like soybean and oilseed rape, it may be reasonable to assume that for much of the area planted to these crops, wild pollinator habitat proximate to these fields has been greatly diminished due to intensive cropping practices (Schulp et al., 2014). Consequently, this methodology would introduce a downscaling parameter to appropriately reduce the pollination value associated with this crop. Yet even if land-cover proxies are effectively employed to better relate crop yield to wild pollinator density, as the authors of the California valuation point out, there remain significant problems with equating pollinator-mediated *yield* increases to increases in actual agricultural *value*.

2.5 THE PRICE IS RIGHT? (ASSUMPTION 3)

It is perhaps not in agro-ecology, but in the sphere of circulating agricultural commodities that current efforts to price pollination services are most problematic. The *IPEV* approach circumvents the complexity of this issue by essentially calculating the apparent loss of economic benefits if all bees suddenly vanished. Such a supply "shock" approach, as Muth and Thurman (1995) observe, starts from the premise that "if honeybees disappeared overnight, the yield per apricot tree [for example] would decline by 56 percent" (p 20). Two problems immediately arise from this conceptual framework. The first is that this approach says nothing about the likelihood that such a widespread collapse would occur and if it did, what might result. The second problem is that the supply shock approach is fundamentally a static analysis that overlooks the inevitable dynamic responses of consumers and farmers to declines in pollination services.

2.5.1 Is pollinator collapse possible and what would it mean?

It remains unlikely that all important pollinator taxa will uniformly decline throughout a given region or across the span of regions where any given crop is grown. For example, preliminary European studies of pollinator community responses to factors such as agricultural intensification and loss of pollinator habitat suggest that these do not so much lead to general pollinator decline, but rather, a homogenization in which a handful of pollinator taxa dominate (Biesmeijer et al., 2006; Carré et al., 2009). Moreover, in some regions where conservation efforts have been undertaken, species richness declines and biotic homogenization have slowed (Carvalheiro et al., 2013). The implications of such shifts are unclear but suggest the need to prefigure probable scenarios that pollinators are likely to face.

Experimental work by Cutler et al. (2012a) models the impact of complete bee pollinator collapse on lowbush blueberry yield, a crop that is essentially entirely insect pollinated and in which bees (both managed and wild) are understood to be the key service provider. Surprisingly, in plots from which all bees and all other daylight active pollinators were excluded, pollination was still affected by nocturnal insects though fruit set were reduced by approximately 10%. These hidden natural pollination resources, moreover, may be coupled with a reallocation of farm and managed pollinator resources. Monck (2008), for example, modeled the effect of a hypothetical introduction of an exotic honey bee pest to Australia. His work suggests that managed pollinator stocks would recover from initially high losses, as confirmed in the real world multi-national losses caused by Colony Collapse Disorder (Rucker et al., 2012b), but the recovery would be attended by a significant lag. These lags are predicted to result in a disproportionate number of remaining colonies being reallocated to pollinator-dependent crops that are either high value, freshly planted long-lived crops, such as orchard trees, or crops that are particularly capital intensive. In other words, the effects of a severe managed pollinator decline would potentially be felt unevenly through agricultural production, with some crops experiencing no decline in service provisioning, whereas others would experience considerable short-term disruptions.

It is important to note that even such short-term disruptions have not been demonstrated in practice, in spite of recent sustained losses of honey bees. For example, no relationship was found between the fees paid for honey bee colony pollination rentals across 12 different crops in the Pacific Northwest region of the U.S. and price fluctuations for those crops from 1987-2009 (Rucker et al., 2012a). Instead, rental fees were most strongly influenced by factors associated with the income and costs of beekeeping firms. For example, the strongest general effect on the price of pollinator rentals was associated with a largely external parameter, namely the price of honey. The authors hypothesize that beekeepers shift their colonies away from pollinating crops with low potential honey yield, such as cranberries, as the price of honey increases. This effectively increases the cost of renting honey bees and is independent of the kind of price increases associated with the loss of pollinator stocks. A notable exception was observed for the rental fees paid to beekeepers for the pollination of almonds in California, arguably the largest single market for insect pollinators in the world. From 2004-2009 the price of renting a single colony to pollinate this crop rose dramatically from $59 (USD) to $140. While this price increase is frequently attributed to the heavy loss in the stocks of US colony numbers following the onset of Colony Collapse Disorder, Rucker et al (2012a) point out, that the price increase occurred two years before these losses. Rather than a dramatic loss of colonies driving rental prices, their analysis reveals that the strongest effect on pollination rental fees was connected to the price of diesel fuel, such that a 10% increase in the cost of this input was associated with a $7.79 increase in the price of renting a colony. They suggest that this parameter largely explains why other crops in the region with lower transportation costs, for example apple, have remained stable across this period. This implies that current prices for pollination, even during a period of significant colony decline, have not been strong enough to generate the kind of agricultural disruption and reorganization forecasted by the Australian model (Monck, 2008). Thus, the vulnerability of crop yields to a pollinator supply "shock" is likely overstated, rendering the significance of the apparent €153 billion of global pollination service value unclear. The relevance of this figure is troubled yet further when considering what a hypothetical sudden pollinator shortage would have on the overall price of crops sold by farmers and paid for by consumers.

2.5.2 Responding to price signals

The dynamic response of managed pollinators to stock declines belies a deeper problem with current valuation estimates: their weakness in characterizing the response of both consumers and agricultural firms to any changing supply of pollinators. Although the response of consumers has been incorporated into the standard *IPEV* framework using historic estimates of the elasticity of price to demand for different crops (Gallai et al., 2009; Southwick and Southwick Jr, 1992), this approach remains problematic not only for methodological reasons (for an extended discussion see Winfree et al., 2011) but also because of significant gaps in information. For one thing it is unclear how consumers would respond to a changing supply of different pollinator-dependent crops independently of one another (Hein, 2009; Southwick and Southwick Jr, 1992). So, for example, a rise in the price of blueberries is considered as independent of the price of a potential substitute, like apples. In other words it is difficult to predict how pollination-mediated decline would impact consumers if the affected crops are broadly substitutable (Simpson, 2013). Oilseeds pose a particular problem in this regard, as different species are readily interchangeable for most food processing pathways (Friedmann, 1993).

Furthermore, the supply shock approach does not account for the possibility that a decline in pollinator supply may occur at different spatial scales or times. For instance, a decline of pollinators in one apple growing region of the world, for example in China where the bulk of global production occurs (Lautenbach et al., 2012), may have very little effect on apple prices beyond the short term, as it may induce production to expand among competitors such as India, Iran, and the U.S. These disruptions may be even less noticeable in an annual crop like soybean which is widely spread over multiple continents (Lautenbach et al., 2012). This level of analysis has yet to be conducted beyond very crude and static estimates, namely the percentage of current regional (Gallai et al., 2009) or national (Lautenbach et al., 2012) agricultural economic output that is dependent on pollinators. Consequently, forecasts of the risk of losing pollinator-mediated value remain highly speculative.

2.6 WHICH WAY FORWARD FOR CALCULATING POLLINATOR VALUE?

I have found that current approaches to valuing the contribution of insect-mediated pollination to agriculture are problematic on several levels. A key feature of these valuation exercises is that some fraction of crop value, D, is dependent on crop pollination. I have argued that D is incorrectly treated as a fixed constant, rather than as a dynamic parameter that is influenced by variations in pollinator density and the breadth of cultivars and growing conditions that exist in practice (Assumption 1, Figure 1). Moreover, to date, D has been derived from a relatively small base of empirical studies, leading to a particularly acute problem with global-scale cropping systems with low to moderate levels of pollinator dependence, such as soybean and oilseed rape. In turn, the dominance of these crops in many valuation estimates may substantially overstate the growing gap between the supply and demand for pollinators in agriculture (Aizen et al., 2008, 2009; Breeze et al., 2011; Breeze et al., 2014b; Garibaldi et al., 2011a; Schulp et al., 2014). I also suggest that extending valuation calculations to the specific role of wild pollinators is further troubled as current efforts tend to only coarsely discern pollination flows arising from wild versus managed insects, if they separate these flows at all (Assumption 2, Figure 2.1). Finally, valuation methodologies effectively overlook the countervailing activity of consumers, farmers, conservation agencies and beekeepers in the face of realistic scenarios of declining pollinator supply (Assumption 3, Figure 2.1).

A number of different strategies are emerging to deal with the assumption of static values of D. The first, outlined in the study by Kasina et al. (2009a) involves empirically measuring yield in open pollinated (f_p) compared to pollinator excluded plots (f_{pe}) for all the major pollinator-dependent crops across a growing region. Clearly replicating such an effort at national, let alone an international, scale is not operationally feasible. A productive line of research, however, might involve identifying the major sources of variation in these two parameters for crops with higher aggregate apparent pollinator-mediated value, such as apple and soybean. Such information would enable an objective basis to estimate the uncertainty associated the values for these key crops. As discussed in Section 2.3.4, I hypothesize that for crops such as soybean and oilseed rape there may be little difference between f_p and f_{pe} across the bulk of the area in which these crops are grown. Should this be borne out, it would result in a significant downward estimate in the global value of insect-mediated pollination.

Another strategy for addressing the shortcomings of current D estimates is to combine a restricted definition of the parameter, one that is derived by the comparison of yield in pollinator supplemented plots (f_{pmax}) to pollinator excluded plots (f_{pe}), and adjusting these values according to variation in the density and diversity of wild pollinator stocks in proximity to pollinator-dependent crops. As discussed in Section 2.4.2, the most promising approach for estimating wild pollinator density is using land-cover as a proxy for habitat (e.g. Chaplin-Kramer et al., 2011; Schulp et al., 2014). These spatial approaches might further be merged with an emerging understanding of crop-specific relationships between the density of wild versus managed pollinators with respect to crop yield (Garibaldi et al., 2013). Conceivably a more robust framework for estimating the relative contributions of wild pollinators than the widely used approach of using ρ is technically possible.

Perhaps the best way forward is an approach that avoids estimating D altogether, through the calculation of a crop's pollen deficit. Schematically, this technique compares how much potential yield is pollen limited by comparing the yield in pollen supplemented plots (f_{pmax}) to open pollinated plots (f_p). Moreover a comprehensive and easy-to-implement international framework for estimating pollen deficit was recently proposed (Vaissière et al., 2011). Valuation based on pollen deficit would mark a decisive departure from the *IPEV* methodology and, as such, offers a way around some of its key problems, for example its assumption of a "supply shock". Coupled with estimates of the marginal costs of pollinators, either in terms of the prices of managed pollinators or the cost of pollinator habitat rehabilitation, it could be used as a means to determine if current pollinator resources are managed efficiently. At this point in time, however, there are only small preliminary studies that calculate the marginal benefits of insect pollinators using pollen deficit data (e.g. Morandin and Winston, 2006).

2.7 WHY DO WE VALUE POLLINATOR VALUATIONS?

We are at a crossroads where we either continue with valuation calculations that are based on data that are neither systematically collected nor representative of actual agricultural practices, or else direct significant investment into generating new data specific to these ends. Perhaps we may already be starting down the second path. Independent of issues surrounding pollination there has been considerable momentum over the past decade towards ecosystem service valuation, culminating in the recent Intergovernmental Platform on Biodiversity and Ecosystem Services (IPBES). It is important to note that a major component of the IPBES 2014-2018 work program involves a Fast Track Assessment of the role of pollinators and their impact to food assessment (Deliverable 3a in IPBES, 2013) and a central element of this assessment is economic valuation. I anticipate a major conclusion of this work will be to call for investments into research to enable better valuations of pollination services. A comprehensive and systematic approach to collecting data specifically in aid of valuation efforts, while marking a radical departure with the past, may present additional problems. For all their shortcomings perhaps one of the great advantages of previous approaches to valuing pollination services was that they incorporated data from existing studies. This meant that few demands were placed by the valuation process on pollinator research itself.

Though technical solutions are attractive, I feel they avoid a more central question, namely why has valuation developed into the foremost grounds for arguing for pollinator conservation in the first place? It is important to note that the basis of justifying conservation though its economic welfare benefit has recent origins (Gómez-Baggethun and Ruiz-Pérez, 2011; Norton and Noonan, 2007; Sagoff, 2012; Vatn and Bromley, 1994) and has been associated with the broader social and economic transformations of the latter half of the twentieth century, namely the transition from Fordism into neoliberalism (Harvey, 1989). This transition has had far-reaching effects, one of which was the broad reconfiguration of how state-mediated agri-environmental schemes are justified (Potter, 1998). Although a few authors have attempted to understand the turn to valuation in relation to these broader historical changes (Sagoff, 2012; Spash, 2011), I feel the specific case of pollinator services may prove useful in clarifying what this turn ultimately means. Potentially the same historically-conditioned imperative that drove the U.S. beekeeping industry to value honey bee pollination benefits against the threat of losing a major price-support subsidy (Muth and Thurman, 1995; Robinson et al., 1989), operate in the current drive to value wild pollinator services against an overall indifference to publically funded conservation initiatives (Pearce, 2007). In other words what might be obscure in current debates over whether managed or wild insect species generate the bulk of agricultural value (see Section 2.5) is the extent to which these positions arise from the same set of social and political pressures. Moreover, these pressures may express themselves beyond the changing role of the state towards agriculture and conservation. I think, for example, it is significant that the explosive growth of pollinator dependent crops in the 1990s (Aizen et al., 2009) tracks record low levels of farm profitability, beginning in the 1980s, that accompanied widespread liberalization in agricultural trade (Friedmann, 1993). While the subsequent push to generate higher value markets through the cultivation of pollinator-dependent ‘exotic’ fruits, nuts and vegetables could be better quantified by a more sophisticated valuation methodology, such an approach, at best, can only describe these trends, not provide insight into their ultimate cause. Consequently a more robust framework would grasp the meaning of valuation in terms of the changing relationship of conservation to society in history. Such a framework could prove helpful in understanding the social basis of conservation, particularly in light of the most recent phase of restructuring following the global economic downturn of 2008. Developing such a framework is the focus of Chapter 5.

The issue of generating policy pressure, in fact, may not be the same thing as rendering an accurate picture of how pollination services concretely flow to society. This is reflected in the fact that there remain few examples whereby the economic benefits of wild pollinators result in farmers implementing pollinator conservation without state subsidies (Ghazoul, 2007; Sagoff, 2011). While this disconnection may signal the need for increased farmer education (Kasina et al., 2009b; Munyuli, 2011), it may also reveal the ways in which we lack the framework through which to understand the existing interactions among biophysical pollinator resources, farming operations and society.

In lowbush blueberry production in northeast North America growers appear to maximize their profit without engaging in meaningful rehabilitation of pollinator habitat. A recent survey of blueberry growers registered this disconnection; the widely perceived benefits of wild pollinators do not translate into investment in habitat conservation, but rather inexpensive and passive changes to farm management practices, such as withholding insecticide sprays during bloom (Hanes et al., 2013). Notably this is for a crop in which wild pollinators have been estimated to provide sufficient pollination across 25% of farms (Eaton and Nams, 2012). While investment has stubbornly resisted habitat protection initiatives, it has readily been available for the rapid expansion of commercially produced *Bombus impatiens* colonies, a species which is superior to honey bees for pollinating blueberries (Desjardins and De Oliveira, 2006; Drummond, 2012) but whose introduction is implicated in harming wild *Bombus* spp. populations through pathogen spillover (Graystock et al., 2013; Murray et al., 2013; Szabo et al., 2012). Ostensibly, even growers of crops that are highly pollinator dependent and that currently enjoy medium to high wild pollinator densities may opt for avenues for maintaining high productivity that ultimately undermine these populations (although see the findings in

Chapter 4 concerning the poor efficacy commercially produced *Bombus impatiens* colonies relative to wild *Bombus* spp. queens).

Spangenberg and Settele (2010: 329) point out that valuation has broadly proven ineffective: "the political argumentation (ecosystem functions as a basis for survival and development) has not been extremely successful in the communication to decision makers and the lay public, and it left hardly any footprint on the inner-science discourses". Fischer et al. (2007) agree, pointing out that the trend towards justifying the welfare gains associated with conservation has generally not coincided with the improvement of key biophysical indicators, but instead their deterioration, leading to a widening "sustainability gap". Consequently, this disconnection between means (valuation) and ends (healthy and diverse wild pollinator populations), may not be fully addressable at the level of technical innovation to current valuation methodologies. It is quite possible, in fact, that if the rising apparent value of pollinator services signifies anything, it is not the growing social necessity for conservation, but rather the opposite: our increasing inability to understand the social basis upon which conservation might be achieved which will be the focus of Chapter 5.

2.8 SUMMARY TO CHAPTER 2 AND TRANSITION TO CHAPTER 3

The focus of Chapter 2 was the broad and technical issue of how to estimate the economic value rendered by wild pollinating insects to agriculture. I identify three assumptions made in these calculations, namely that we know: 1) how much of the output of various crops is dependent on the activity of insect pollinators, 2) how much of this pollination is attributable to the activity of wild pollinators and 3) how the decline of wild pollinator populations will change the value of agricultural commodities. I demonstrate that there is insufficient evidence to support each of these assumptions. In Chapter 3 and 4 I explore the first two assumptions (crop output is dependent on pollination and the pollination is affected by wild pollinators) in experimental and survey work in the key pollinator-dependent crop in Atlantic Canada (lowbush blueberry). Chapter 3 specifically takes up the question of whether yield in blueberry is primarily pollen limited using a factorial experiment in which two levels of pollination are compared in plots in which insecticide and fungicide treatments are added or withheld.

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CHAPTER 3 CONTEXTUALIZING POLLINATION BENEFITS: EFFECT OF INSECTICIDE AND FUNGICIDE USE ON FRUIT SET AND WEIGHT FROM BEE POLLINATION IN LOWBUSH BLUEBERRY

The following chapter has been published in *Annals of Applied Biology* (2014: 165:387-394) with Peter Tyedmers and G. Christopher Cutler as co-authors.

3.1 ABSTRACT

Current approaches to determining the value of insect pollinators to crop yield assume plants are primarily pollen limited. This is particularly relevant in a crop such as lowbush blueberry, *Vaccinium angustifolium*, where no fruit will set without insect-mediated cross-pollination. However, such valuations usually ignore other factors that are necessary to maximize crop yields. I conducted an experiment to test whether yields of lowbush blueberry attributed to pollinator activity increased independently of pest management. The experiment was a 2×2 factorial design, incorporating two intensities of pollination (25 or 100% of flowers), and two levels of insect and disease management with recommended fungicide and insecticide sprays ('full inputs', or 'no inputs'). I demonstrated an interaction between these two factors, such that increased fruit set at harvest was only possible if 100% pollination was combined with the 'full input' treatment. Furthermore, increases in fruit weight among the remaining treatments were

only realized in the ‘full input’ plots. These results suggest that the value accorded to pollinator activity in blueberries is strongly dependent upon pest and disease management of the crop.

3.2 INTRODUCTION

Insect pollinators play a vital role in achieving high yields in a number of crops across the world (Klein et al., 2007; Lautenbach et al., 2012). In 2008 it was estimated that pollinators contributed $160 million to the production of blueberries in Canada, making it one of the largest beneficiaries of insect-mediated crop pollination in the country, alongside hybrid canola seed and apples (Campbell, 2008; updated from Scott-Dupree, 1995). This value is divided among different regions and includes the highbush blueberry sector, but a considerable portion of the value of pollination attributed to blueberries is generated in Quebec and Atlantic Canada on the lowbush blueberry crop (*Vaccinium angustifolium* Ait., syn. "wild blueberry").

Current methods for pollinator service valuation presumes the yield of pollinator-dependent crops to be primarily pollen limited (Calderone, 2012; Gallai et al., 2009; Morse and Calderone, 2000; Scott-Dupree, 1995; Winfree et al., 2011). This assumption is perhaps most stark in crops like blueberry where 100% of the fruit-set is attributed to pollinator activity. Although it is true that blueberries cannot set fruit without insect pollinators (Cutler et al., 2012a; Lee, 1958), this approach to valuation assumes that the marginal yield increase owing to other inputs, such as pesticides or fertilizer, is zero. This might indeed be the case among many crops where other limitations have been alleviated through intensive agriculture, at the expense of wild pollinator populations (Bommarco et al., 2012; Goulson, 2010; Winfree, 2010). In other words, as crops become more intensively managed, pollen flow tends to become the more pressing limitation on yield (Ghazoul, 2007).

The focus on pollen limitation in crop plants has not kept pace with the advances in our understanding of the interaction between reproduction and resource limitation in wild plants (Knight et al., 2005; McCall and Irwin, 2006). Such interactions, in fact, appear to be at play in crop yield (Bos et al., 2007b). Even if plants set large numbers of fruit initially – i.e. pollen limitation has been overcome – these fruits may subsequently abort due to other factors, such as moisture (Groeneveld et al., 2010), soil nutrient deficiencies (Boreux et al., 2013) or pest damage (Bos et al., 2007b; Lundin et al., 2013). Furthermore, the interaction between pollination and other plant limits may not be simply additive. For example, red clover seed yield increases due to pollination were disproportionately high if pest herbivory was suppressed, indicating a synergism between pollination and pest management (Lundin et al., 2013).

I suspect that similar interactions occur between blueberry yield, plant limitations, and pollination. While there is evidence that the agronomic factors that limit yield of commercial wild blueberries have dramatically decreased in some areas over the past twenty years, this effect appears quite uneven across the growing region. Up to four-fold increases in productivity observed in Maine across this period have been attributed to the increased use of honey bees, fertilizers, fungicides, insecticides and herbicides, but this same increase has not been observed in many parts of Atlantic Canada where similar management techniques are also applied (Yarborough, 2004). Furthermore, while yield increased 192 kg ha^{-1} for each honey bee colony added in Nova Scotia, Canada, the response was extremely variable and generally diminished above four colonies per ha (Eaton and Nams, 2012). The variation in response, as well as the apparent ceiling on returns to additional colonies, suggests that plant and pollen limitations remain quite interconnected in this system.

I am interested in clarifying the relationship between crop management practices, plant limitations, and the response of blueberry yield to pollination. In the current study I explored the response of blueberry yield to two levels of insecticide and fungicide use (full inputs, or no inputs), and two levels of bee pollination (full access to pollinators, or restricted access to pollinators). I hypothesized that pesticide treatments would work in concert with pollination, such that yield gains from pollination would not be realized in the absence of pesticides.

3.3 MATERIALS AND METHODS

3.3.1 Experimental Setup

The experiment was done in 2012 in a commercial wild blueberry field in Debert, Nova Scotia (45°26’35’’N, 63°26’57’’W). Twenty-four circular galvanized steel rings (76 cm diameter, 30 cm high), were laid out over blueberry plants in six blocks, giving four rings per block. Each ring constituted a treatment plot, which was randomly assigned within each block.

The experimental design was a 2x2 factorial, with two levels of pesticide treatment, and two levels of pollination treatment. For pesticide treatments, two plots per block were assigned recommended rates of insecticide and fungicide (treatment = 'full input'), and two plots per block were left without treatment (treatment = 'no input'). For pollination treatments, two plots per block received low intensity pollination, which consisted of covering the rings with 1.0 mm craft netting after the first 25% of the flowers bloomed. This netting remained over the aluminum rings until the blooming was complete, such that only a quarter of the flowers were pollinated (treatment = ‘25%’). In the other two plots per block, plants were available to pollinators through the entire bloom period (treatment = '100%') (Figure 3.1).

Pollination among the blocks was facilitated by placing a commercial bumble bee colony (*Bombus impatiens*) (Koppert Biological Inc., Scarborough, ON) between blocks 2 and 3, and another colony between blocks 4 and 5. Bees and plants were enclosed in a bee-proof cage measuring 1.8 x 1.8 x 12 m long covered with 30% shade cloth (Figure 1). The covering was installed immediately prior to the plants coming into flower (15 May) and removed following final petal drop (5 June).

The number of stems per plot (ring) was counted prior to bud break. Within each plot, four stems were randomly selected and the following parameters were measured: (a) the total number of blossoms during early bloom (15 May); (b) the number of fruit initially set (14 June); (c) the number of ripe and unripe berries at harvest; and (d) the total weight of ripe berries (2 August). Average ripe fruit weight was calculated by dividing the total weight of ripe berries harvested from the four stems by the number of ripe berries harvested. All berries in each plot were then harvested using a commercial wild blueberry hand rake and their weight per plot recorded using a digital scale (*plot yield*).

3.3.2 Pesticide Treatments, Insect Pests and Diseases

Topas® 250E (propiconazole, Syngenta Canada, Guelph, ON) was applied at the early bud break stage (8 May) at a rate of 0.5 L product·ha^{-1}. Pristine® WG (pyraclostrobin and boscalid, BASF Canada, Mississauga, ON) was applied during early (17 May) and mid bloom (23 May, 1 June) at a rate of 1.6 kg product·ha^{-1}. These fungicides were applied for control of *Monilinia vaccinii-corymbosi*, which results in a blight that kills blossoms and leaves, and various leaf diseases, primarily caused by *Septoria* spp. (Septoria leaf spot), *Thekopsora minima* (leaf rust) and *Valdensinia heterodoxa* (Valdensina leaf spot).

Decis® 5E (deltamethrin, Bayer CropScience Canada, Guelph, ON) was applied at a rate of 125 ml product ha^{-1} during mid bloom (17 May), and Delegate™ WG (spinetoram, Dow AgroSciences Canada, Calgary, AB) was applied at 200 g product ha^{-1} during mid bloom (23 May). These insecticides were applied to control blueberry spanworm larvae (*Itame argillacearia* Packard) (Lepidoptera: Geometridae).

Chemicals were applied separately in a spray volume of 400 L ha^{-1} using a hand-held CO_2 propelled 48 cm boom sprayer (Bellspray Inc., Opelousas, LA) with two flat spray XR8002VS TeeJet nozzles operating at 240 kPa. All chemical sprays were applied under sunny conditions with no wind at a distance of 40-50 cm above the plants.

In order to ensure plots experienced economically significant levels of insect pest pressure, 10 freshly-collected first-instar blueberry spanworm larvae were placed on different randomly selected stems in each plot on 15 May. Fungal diseases occurred naturally and were not artificially introduced onto the plants. The severity of damage from insects and fungal diseases to the blueberry plants in each plot was estimated immediately following the bloom period (14 June) using two methods. First, a sampling ring (20 cm in diameter) was randomly placed within the plot. Each stem within the ring (19.8 ± 1.1 (SE) stems/ring) was visually scored along a 1-5 scale based on the percent damage on leaves or fruit: '1' (0% damage);'2' (>0-25% damage);'3' (>25-50% damage);'4' (>50-75% damage); or '5' (>75-100% damage). The average severity score among all the stems within each plot was then calculated. Second, the number of flowers that had been lost to infection with the fungus *Monilinia vaccinii-corymbosi* were counted on the same four stems in each plot that were used for floral and fruit set counts.

3.3.3 Statistics

All analyses were conducted in R (R Development Core Team, 2013). I used a linear model to test the effect of each factor, namely pollination level (pollinating 25% or 100% of flowers), pesticide use (no or full inputs), their interaction and block (Gotelli and Ellison, 2013). Differences in the average disease and insect pest severity score between the 'full input' and 'no input' plots were compared with a non-parametric Mann-Whitney *U*-test (Gotelli and Ellison, 2013). Since the only combination of pollination and pesticide use were randomly assigned to blocks (Figure 1), the analysis of input levels on plant damage was conducted without consideration of the block structure. The percentage of *M. vaccinii-corymbosi* flower infection for between the two levels of pollination or input treatment was compared using a Student's *t*-test (Gotelli and Ellison, 2013). In order to normalize error variance before analyses, percentage data were arcsine-root transformed, and yield per plot and flowers per stem data were square root transformed (Gotelli and Ellison, 2013). Average fruit weight data were not transformed. Assumptions of normal error distribution and homoscedasticity were confirmed by an examination of residuals (Anderson-Darling (normality), $P > 0.05$; Bartlett's test (homoscedastity), $P > 0.05$).

3.4 RESULTS

The pollination treatment significantly affected both total fruit set and ripe fruit set, whereas inputs only affected the ripe fruit set (Table 3.1, Figure 3.2 A,B). The interaction between the pollination and input treatments, however, were significant for both measures of fruit set (Table 3.1, Figure 3.2 A,B). Moreover, the pattern of the interaction was similar for both total and ripe fruit set, namely that the positive effect of pollinating 100% versus 25% of the flowers was only realized in the presence of the 'full input' treatment (Figure 3.2 A,B).

I did not observe a similar increase in average fruit weight to the pollination and input treatments as I did for fruit set. In plots that received full pesticide inputs, the average weight of ripe fruit was greatest from stems receiving restricted access to pollinators (25%) rather than from stems where flowers received full access to pollinators

(100%) (Figure 3.2 C). In fact, flowers in ‘full input’ plots with restricted pollinator access (25%) produced ripe fruit that were 1.5-fold heavier than in plots with full pollinator access. In contrast, there was little difference in the average weight of fruit between the pollination treatments in the ‘no input’ plots (Figure 3.2 C). Average fruit weight, however, was not affected by pollinator or input treatment alone, but rather by interaction of these factors (Table 3.1).

Although pollination and input treatments affected both fruit set and average fruit weight, there was no evidence to suggest they were significant factors in determining plot yield (Table 3.1, Figure 3.2 D). Differences in plot yield among treatment groups were also not due to variation in the number of stems per plot or flowers per stem (stems: $F_{3,15} = 2.25$, $P = 0.12$; flowers: $F_{3,15} = 1.07$, $P = 0.39$).

The average disease and pest severity score was lower for plots treated with fungicide and insecticide (1.77 ± 0.15 (SE) severity score/plot) compared to the plots in which the treatments were withheld (2.16 ± 0.15 (SE) severity score/plot) ($W = 187$, df = 22, $P = 0.04$). The magnitude of protection afforded by the use of insecticides and fungicides could also be discerned by the percentage of stems displaying no visible damage. While over half the stems in the plots treated with fungicide and insecticide scored 1 (0% visual damage to leaves or flowers), this was relatively rare among plots where the treatment was withheld, where only 12% of the stems were free of damage. The most prominent source of damage in most plots was fungal diseases, mainly *M. vaccinii-corymbosi,* but insect damage was most pronounced in two plots. Fungicide sprays resulted in a six-fold decrease in the percentage of flowers killed by the fungus (full inputs = 1.31 ± 0.42% (SE); no inputs = 8.16 ± 2.47%; *t* = -2.92, df = 22, *P* = 0.008). The percentage of floral infection, however, was independent of level of pollination (*t* = -0.83, df = 22, *P*=0.42).

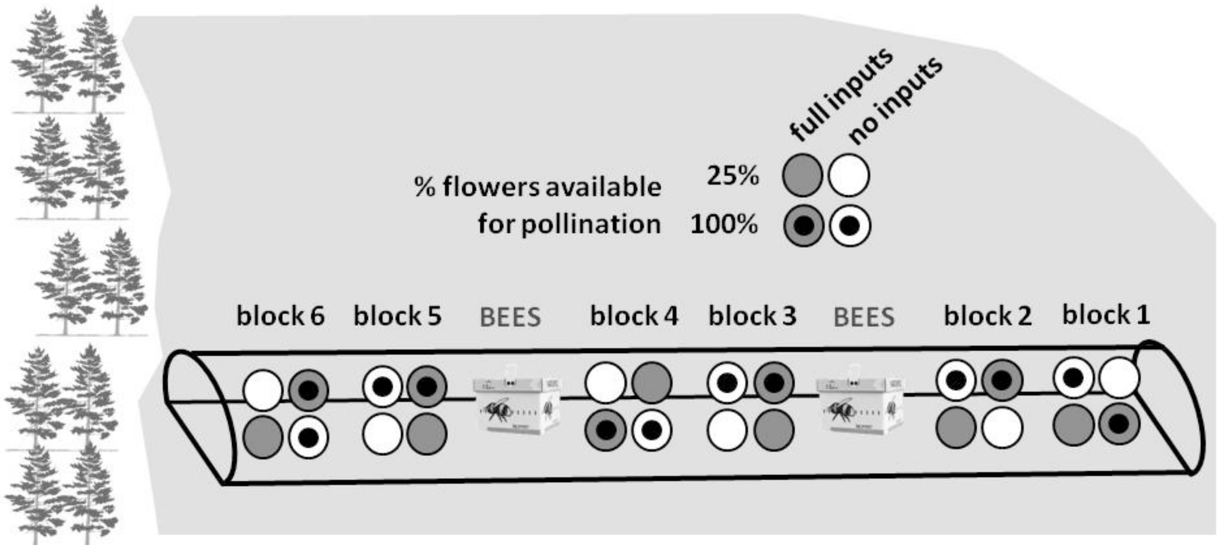

Figure 3.1 Experimental setup to examine effects of pollinator limitation and pesticide input in lowbush blueberry.

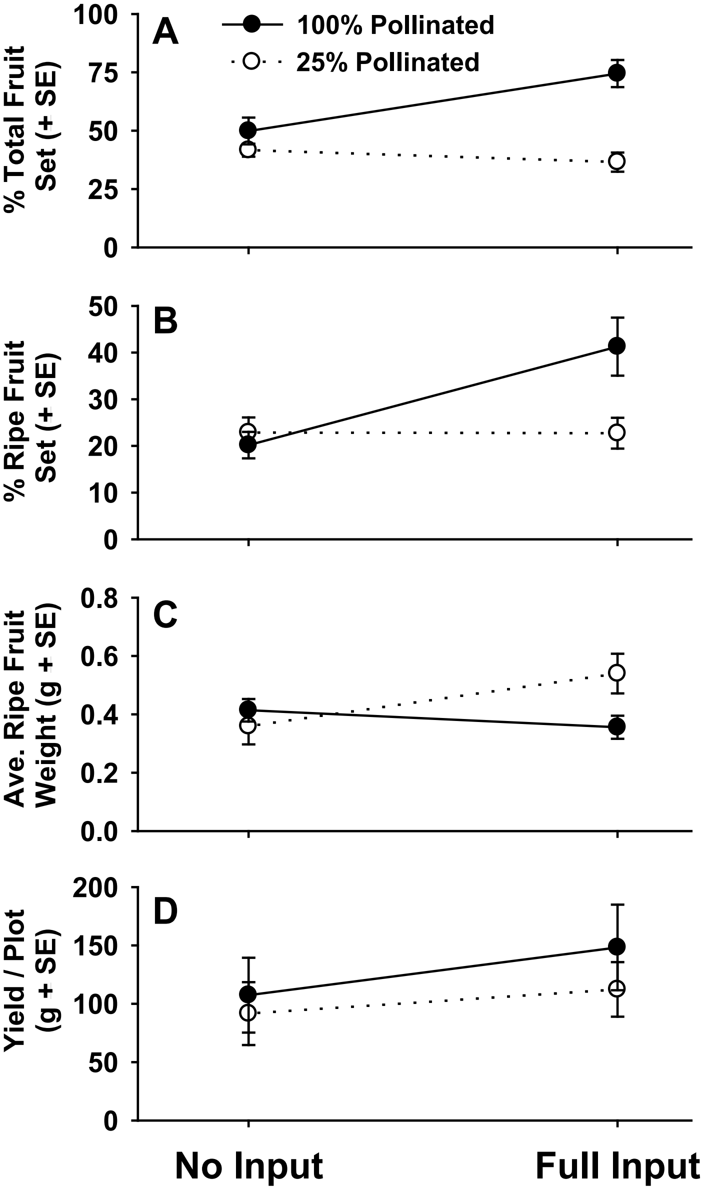

Figure 3.2 Effects of two levels of pollination (100% or 25% of flowers pollinated) and the effect of withholding ('no input') or using ('full input') insecticide and fungicide sprays on lowbush blueberry: (A) total percent fruit set at harvest; (B) the percentage of ripe fruit at harvest; (C) average weight of ripe fruit on four stems in each plot; and (D) total fruit yield per plot at harvest (n=6 plots per treatment).

Table 3.1 Results of linear models analyzing effects of insecticide and fungicide (input) and pollination (25% versus 100% of flowers pollinated) on lowbush blueberry percentage fruit set, average ripe fruit weight, and yield.

Factors, by variables	**df**	***F***	***P***
Total fruit set			
Input	1	4.04	0.063
Pollination	1	21.58	< 0.001
Input*Pollination	1	9.24	0.008
Error	15		
Ripe fruit set			
Input	1	8.69	0.010
Pollination	1	4.81	0.044
Input*Pollination	1	8.82	0.010
Error	15		
Fruit weight			
Input	1	3.81	0.070
Pollination	1	4.24	0.057
Input*Pollination	1	14.53	0.002
Error	15		
Yield per plot			
Input	1	1.79	0.200
Pollination	1	1.03	0.325
Input*Pollination	1	0.04	0.839
Error	15		

3.5 DISCUSSION

I found evidence that suppression of lowbush blueberry fungal diseases and insect pests interacted with pollination of the crop to significantly impact berry fruit set and average ripe fruit weight. This interaction had two dimensions. First, the response of plant fruit set to increased pollination was highest in the presence of fungicides and insecticides. Second, when pollination was limited, low fruit set resulted in increased fruit weight, but only when fungicides and insecticides were used. Both findings suggest that yield effect due to loss of pollination in this crop is greatest when insect pests and diseases are controlled, or in other words, that increased pollination cannot compensate for losses from pests and diseases (Lundin et al., 2013). The combined effect of the two types of interactions, however, were not reflected in overall plot yield. My results agree with those of another recent studies that suggests the benefits accrued from pollination are linked to the pest and disease management of the plant (Lundin et al., 2013).

The impact of leaf diseases and insect leaf herbivory on fruit abortion or the rate of fruit development is poorly documented for pollinator-dependent crops. Although early research suggested early blueberry defoliation could reduce fruit set (Aalders et al., 1969), very little work has been done on this problem. The question of the interaction between fruit set or fruit quality and plant limitations has only recently become prominent (Boreux et al., 2013; Bos et al., 2007a; Bos et al., 2007b; Groeneveld et al., 2010; Lundin et al., 2013). Consequently, my work is one of the first demonstrating the interaction between pollinator intensity and pest and disease pressure that goes beyond direct insect seed predation (Lundin et al., 2013).

There are several possible mechanisms driving the interactions between pest management and pollination in our experiment. One factor may be the loss of flowers to *M. vaccinii-corymbosi* when fungicide treatment was withheld. The blight caused by this fungus was the most predominant disease in our plots. Treated plots certainly had a lower percentage of flowers damaged by the blight compared to the untreated plots, but this alone cannot not account for all of the observed reduction in fruit set. The prevention of floral loss by the fungicide would be expected to increase the total number of fruit set (Figure 3.1A) as opposed to the percentage of those berries that are ripe. The fact that the effect of inputs was only significant for ripe fruit set and not for total fruit set (Table 3.1), suggests the benefits of combined sprays extend beyond protection of flowers from *M. vaccinii-corymbosi*. Consequently, I hypothesize that other unmanaged pest and/or disease problems that affect the capacity of plants to carry ripe fruit to harvest may have also accounted for some of the observed variation in ripe fruit set. An interactive effect of pollination and plant health on fruit quality, not only on fruit set, would complicate emerging research implicating the role of pollination on fruit quality (e.g. Garratt et al., 2014; Klatt et al., 2014; Ricketts et al., 2004).

The additive effects of pest protection and pollination level on the percentage of fruit set may have been even greater had the scope of our study extended beyond the cropping year. Commercial wild blueberry fields are typically pruned on alternate years to decrease continued vegetative growth and branching, while maximizing fruit yields the year after pruning (Penney et al., 1997). There is evidence that the impact of weeds, insect pests and diseases during the vegetative ‘sprout’ year of growth may have an even larger effect on the number of flowers that are available for pollination the following year (Kennedy et al., 2010; Roloff et al., 2004; Williamson and Miller, 2002; Yarborough et al., 1986). Consequently, the pollination-pest management interaction I observed in the crop year may be far exceeded if the sprout year is taken into account.

Given the small-scale of our experiment, in addition to the experimental character of the pollination and pesticide treatments, it is unclear what the implications are for cultivation under commercial field conditions. Although yields among six plots directly adjacent to the pollinator exclusion tent were within the range observed among plots within the tent (91.9 g ± 12.3 g (SE)), the treatments likely represents an extreme situation in commercial fields. First, it is unlikely that commercial fields would be left without insecticide or fungicide treatment as simulated by my ‘no input’ treatment. Similarly, the levels and timing of pollination in our treatments may only occur under peculiar circumstances in commercial cultivation. Although managed bumble bee colonies are routinely used in commercial fields, my stocking rate was a magnitude higher than the 2.5 colonies per ha recommended for this crop (Drummond, 2012). Consequently, the levels of pollination experienced in the 100% pollination treatment were far higher than that experienced under most field conditions. Moreover, the experimental design placed peculiar constraints on the restricted pollination treatment in that only the very first 25% of flowers were pollinated. Although this situation can arise under commercial conditions if pollinator flight becomes constricted for the latter part of the bloom due to cool weather (Tuell and Isaacs, 2010), it clearly is not equivalent to reducing pollination across the span of the bloom. There is evidence from commercial cranberry (*Vaccinium macrocarpon* Ait.) that fruit set among the first flowers are favored, such that the rates of fruit abortion for later flowers is higher (Brown and McNeil, 2006). Should lowbush blueberry similarly favor the development of early pollinated flowers, I predict that our restricted pollination treatment likely over-estimates yield compared to a treatment in which the pollination of 25% of the flowers was extended across the bloom. Consequently, I speculate that spreading pollination across the bloom would have led to more pronounced differences among the treatments. Chapter 4 will describe a study testing whether these findings can be generalized at a commercial scale.

One of the broader implications of my findings is to call into question current approaches to estimating the financial value of benefits delivered by pollinating insects to agriculture. The most widely cited global estimate (Gallai et al., 2009), which uses the same methodology as the estimates for Canada (Scott-Dupree, 1995) and the U.S. (Calderone, 2012; Morse and Calderone, 2000), prices the activity of pollinators by dividing the entire farm-gate value of the crop by an estimate of the crop's dependency on pollinators. As I outlined in Section 2.3.2, the basic experiment for estimating dependency involves comparing yield on groups of flowers that are open-pollinated compared to flowers from which pollinators are excluded. In cases such as blueberry, this leads to the entire value of the crop being attributed to the activity of pollinators, because no fruit is set in the absence of insect-mediated pollination. Rather than attempt to calculate the marginal benefits of pollinators in concert with the marginal costs associated with these benefits, the traditional methodology essentially calculates the economic benefits lost if pollinators suddenly vanished, in essence under the circumstances of a pollinator "supply shock" (Muth and Thurman, 1995). This approach completely discounts the contributions of other interacting inputs. While a recent revision of this approach incorporates the costs of other inputs (Winfree et al., 2011), it does not account for any dynamic and contextual interaction among pollinator activity, agricultural inputs, and crop yield. My research suggests the methods used for valuation may need to take into account other determinants of yield.

A dimension of pest and disease control not directly raised in this study is the potential negative effect of many pest control products on wild (Bommarco et al., 2012; Goulson et al., 2008) and managed pollinators (Johnson et al., 2013; Mullin et al., 2010; Pettis et al., 2013; VanEngelsdorp et al., 2010). The fact that the benefits of pollination may be bound up with the use of chemicals that harm pollinators may present farmers with a management conundrum. This pollination paradox (Ghazoul, 2007), whereby the very practices that maximize agricultural productivity, and thereby the need for pollination, undermine the flow of pollen among flowers, may be reduced by the adoption of less toxic options for controlling blueberry diseases and pests (Gradish et al., 2012; Ramanaidu et al., 2011). Bommarco *et al.* (2013) propose that many conventional approaches to overcoming limits on yield could be dealt with through a process of ecological replacement whereby a greater proportion of a limiting component of production is supplied by ecosystem services, such as the control of pests by predators or parasitoids (Cutler et al., 2012b). Part of the research associated with Chapter 4 (but not included in the thesis itself) attempts to characterize these trade-offs in commercial fields.

3.6 SUMMARY TO CHAPTER 3 AND TRANSITION TO CHAPTER 4

In Chapter 2 I described how current approaches to valuing pollinator ecosystem services assumes that yield in highly-pollinator dependent crops like lowbush blueberry are strictly pollen limited. In Chapter 3 I used a factorial experimental design to show that the full benefits of high pollination are largely realized in a certain pest and disease management context. The limitation of this study, as explained in Section 3.5, is that it does little to describe to what extent current output is pollen limited because, for example, it is unlikely that growers would leave their fields completely untreated for insect defoliators and leaf diseases. It is possible, given the levels of agricultural intensification, that most plant limits in lowbush blueberry are met, rendering yield predominantly pollen limited. In order to test this hypothesis Chapter 4 describes a two year study of yield variation in commercial lowbush blueberry relative to pollination levels and several key agronomic parameters.

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CHAPTER 4 POLLEN LIMITATION IS NOT THE ONLY LIMITATION ON YIELD IN A HIGHLY POLLINATOR DEPENDENT CROP (LOWBUSH BLUEBERRY)

4.1 ABSTRACT

Current approaches to valuing insect pollinator services to agriculture assume that yield, for highly pollinator-dependent crops, is limited by the number of pollinator visits. This assumption is also reflected in the practices of growers of these crops, who frequently stock their fields with a high density of managed pollinator species during bloom. I tested this hypothesis in a two-year study of commercial lowbush blueberry (*Vaccinium angustifolium* Ation: Ericaceae) in Atlantic Canada. I related variation in fruit set and yield along transects from different fields to that of pollinator diversity and visitation rate, as well as agronomic parameters, such as plant nutrient levels and pest, disease and weed pressure. Although managed honey bees (*Apis mellifera* L.) constituted the largest number of visitations to blueberry flowers during the bloom period, their abundance was not associated with high pollen deposition rates (> 50 pollen tetrads per floral stigma) unlike that of non-*Apis* bee species. There was evidence of limits on yield since less than a quarter of the initial flowers in both years failed to form fruit. Multivariate analysis of the pollinator visitation rate, accounting for other agronomic parameters, suggests there was little evidence that yield was strictly determined by pollinator abundance. Fruit set and harvestable ripe berries per stem along transects were

significantly related to pollinator abundance but only in the second year of the study, and only then in conjunction with patterns of variation observed in tissue micronutrients. The weight of individual fruit in the second year was best explained in relation to agronomic factors and not to pollinator abundance. Finally, pollinator abundance did not explain yield in experimental plots in any year. These findings challenge the assumption that yield in highly pollinator dependent crops are strictly pollen limited.

4.2 INTRODUCTION

The valuation of pollinator ecosystem services, as I explained in Chapter 2, assumes pollen flow mediated by insects to be the key limiting factor in the production of 23×10^8 Mt of food production annually (Klein et al., 2007). The assumption rests on the idea that intensive crop management practices have reduced most other limitations on crop yield to such a degree that only variation in pollinator visitation determines yield (and by extension, the aggregate value of those crops). Whereas the great agricultural discovery of the nineteenth century was that nitrogen was the limiting factor in crop yield and that declining yields could be predicted from the amount of fertilizer applied to fields (von Liebig, 1855), the growing output of pollinator-dependent crops since the 1990s has led many to hypothesize that a parallel shift has recently taken place, such that yield will increasingly be determined by the availability of pollinators (Aizen et al., 2008, 2009;

Aizen and Harder, 2009; Breeze et al., 2011; Breeze et al., 2014b; Garibaldi et al., 2011a; Schulp et al., 2014)[3].

Two recent discoveries have complicated the view of the straightforward pollen limitation presupposition held by such assessments. The first is that yield response of crops to pollinator abundance can be experimentally altered by manipulating crop plant management, for example the availability and timing of crop irrigation (Boreux et al., 2013; Klein et al., 2015), the availability of micro and macronutrients (Klein et al., 2015; Marini et al., 2015), cultivar selection (Hudewenz et al., 2013; Marini et al., 2015) and the reduction of insect and plant disease pressure (Chapter 3, but also Bos et al., 2007a; Lundin et al., 2013). Such work suggests that variation in agricultural management practices may strongly condition the pollinator yield benefits being accrued in any given crop year. The second discovery is that pollen deposition alone cannot account for the benefits that flow from pollinator activity. Instead there is a qualitative dimension that is broadly absent when pollination is conducted by commercial honey bees (*Apis mellifera* L.) as opposed to other pollinator taxa (Garibaldi et al., 2013). It remains unclear, however, how these two dimensions—plant limitations across the range of current crop management practices and qualitative differences in pollen deposition by different taxa—

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[3] As Achim Steiner, the current Under-Secretary-General of the United Nations states: "The way humanity manages or mismanages its nature-based assets, including pollinators, will in part define our collective future in the 21st century. The fact is that of the 100 crop species that provide 90 per cent of the world's food, over 70 are pollinated by bees". (AFP, 2011)

interact within various cropping systems, among different years, and across different regions, in realizing the benefits of insect pollination.

The lowbush blueberry (*Vaccinium angustifolium* Ation: Ericaceae) agroecosystem of Atlantic Canada, eastern Quebec and the Northeastern U.S. is ideal for investigating how variation in crop plant limitation and qualitative aspects of pollen limitation affect pollinator-mediated yield benefits because it represents an extreme case of pollinator dependency. First, blueberry plants are completely dependent on insects to move pollen among flowers to set fruit (Cutler et al., 2012a; Lee, 1958). Lowbush blueberry is also unique in North American agriculture because it is one of the few major crops that develops from wild plants. As such, the crop has coevolved with an endemic wild pollinator fauna that is abundant and effective. For instance, a number of species nest in and around blueberry fields and emerge coincidentally with the blueberry blossom (Javorek, unpublished findings; Cutler et al., 2015), resulting in densities in fields rivaling those of fields that are heavily stocked with managed pollinators. Consequently, over a quarter of lowbush fields in a recent study were shown to receive adequate pollination from wild species alone, without growers having to rent managed honey bee colonies (Eaton and Nams, 2012). Moreover, many wild bees are adept at removing pollen from the poricidal anthers of blueberry using sonication (i.e., vibration of the flight muscles) (Javorek et al., 2002) and are highly attracted to the flowers as their tongue length enables them to collect nectar rewards from the narrow bell-shaped flowers (Vander Kloet, 1988). For these reasons lowbush blueberry is likely the crop most dependent on wild pollinators in North America. Nevertheless, blueberry producers use massive numbers of managed pollinators, most prominently honey bees (Karmo, 1956), but also leafcutter bees (*Megachile rotandata* F.) (Stubbs and Drummond, 1996) and the common eastern bumble bee (*Bombus impatiens*) (Desjardins and De Oliveira, 2006; Stubbs and Drummond, 2001). The high density of wild pollinators and honey bees provides an excellent opportunity to examine the quantitative versus qualitative dimensions of pollinator service flow.

Variation in plant-centered limitations affecting blueberry yield are also likely high relative to other cropping systems. Many factors remain outside the scope of management (e.g., plants develop from wild populations and spread by underground rhizomes rather than cultivation), but there have been significant advances in crop management practices, particularly the use of selective herbicides and insecticides, the broad adoption of nitrogenous fertilizer, and the more extensive use of fungicides for leaf diseases (Bell et al., 2009a; Penney and McRae, 2000; Yarborough, 2004).

The objective of this study is to examine how variation in lowbush blueberry yield is explained by agronomic characteristics of blueberry plants alongside (and interacting with) qualitative difference in pollen deposition by honey bees versus other bee taxa. I test two broad hypotheses using data collected from two seasons from commercial lowbush blueberry fields: 1) while pollinator visitation rate increases floral pollen deposition, fruit set is more strongly related to variation in non-*Apis* species visitation and not the visitation rate of honey bees (as suggested by Garibaldi et al., 2013) and 2) variation in pollinator visitation is the primary determinant of fruit set and final yield in lowbush blueberry (e.g., Calderone, 2012; Gallai et al., 2009; Lautenbach et al., 2012)).

4.3 MATERIALS AND METHODS

4.3.1 Study sites

The study was conducted over two years (*n*=16 fields (2013), *n*=34 fields (2014)) using commercial blueberry fields located in Queens (*n*=7 (2013), *n*= 7 (2014)) and Kings (*n*=9 (2013), *n*=20 (2014)) Counties in Prince Edward Island, and Pictou County (*n*=2) and Colchester Counties (*n*=5) in Nova Scotia (2014) (Figure 4.1). These regions are characterized by mixed agricultural production, mainly cereals, soybeans and potatoes, as well as forests and livestock pasture. Fields averaged 10.0 ha in size (max=27.0 ha, min=1.6 ha) and were usually located at least 3 km apart from each other in each year. Only production fields were included in the study and most had been producing berry crops for more than five years. None of the growers used organic production techniques but relied on standard agrochemical and mechanical pruning practices. Production in both 2013 and 2014 was higher in these fields than in the previous two cropping cycles (Figure 4.2), indicating yields in both years of the study were relatively high.

All data were collected from 100 m-long transects that were established 25 m from the north end of each field (Figure 4.3). Transects were sited on sections of the field with complete coverage of blueberry that were also at least 25 m away from the east or west edges of the field. Average stem density along each transect was estimated by counting all the stems in a 10 × 10 cm^2 area at five randomly selected sections along each transect. Quadrats (1 m^2) were situated at the 0, 25, 50, 75 and 100 m points on each transect. Quadrats were established on all fields in 2013, but only 16 fields in 2014 (Figure 4.1). Placement of the quadrats were such that they contained stems (ramets) from a single *V. angustifolium* clone (see Section 1.2 for a description of the vegetative growth pattern of *V. angustifolium* plants).

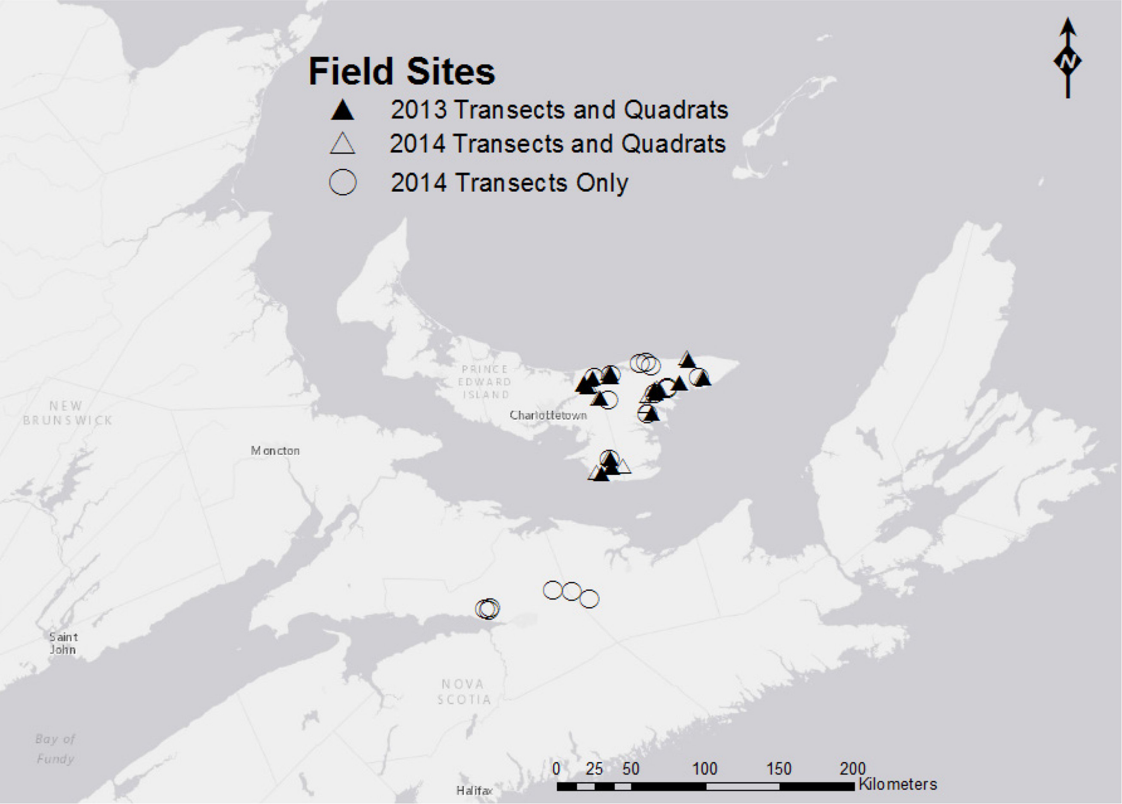


Figure 4.1 Location of commercial lowbush blueberry fields in Prince Edward Island and Nova Scotia used to study pollen limitation in 2013 (*n*=16 fields) and 2014 (*n*=34 fields). In all fields in 2013 both transect fruit set and quadrat yield were measured. In 2014 all locations had transect fruit set measurements but only a subset of the fields had quadrat yield. The global positioning coordinates for each study site are listed in Appendix A.

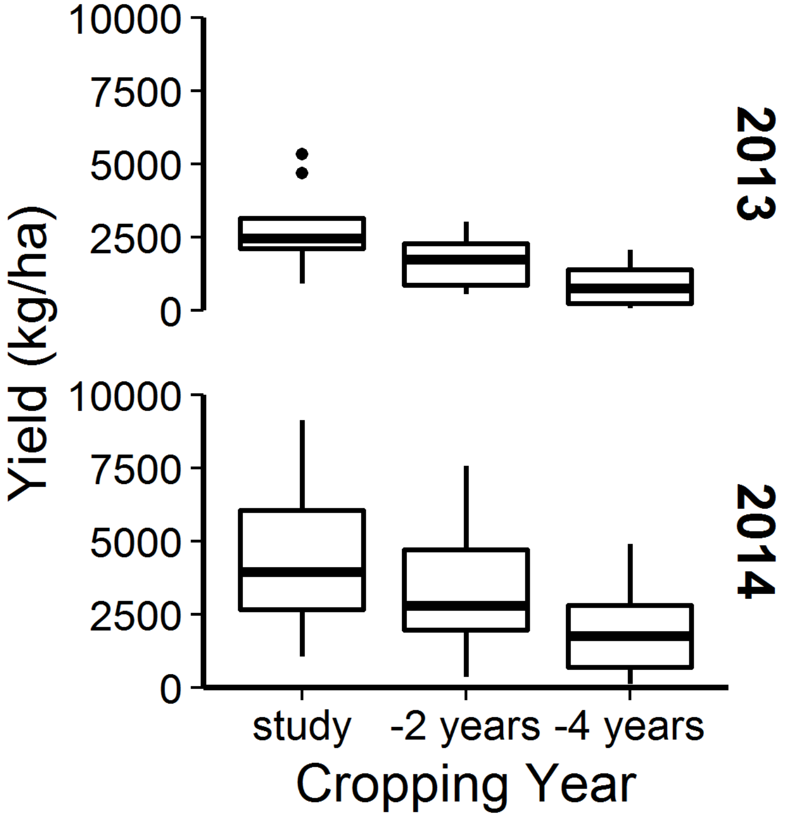


Figure 4.2 Box-and-whisker plots of grower-reported lowbush blueberry yield at a selection of field sites in Prince Edward Island and Nova Scotia where transects were situated to study pollen limitation. These include amounts for the year of the study 2013 (n=8) and 2014 (n=19), but also in two prior cropping cycles. Lowbush blueberry is a biennial crop so these yields where either two or four years prior to the study (-2 years or - 4 years). The horizontal line signifies the median field yield, boundaries the 25 and 75th percentiles and whiskers the most extreme data point that is no more than 1.5 times the length of the box.

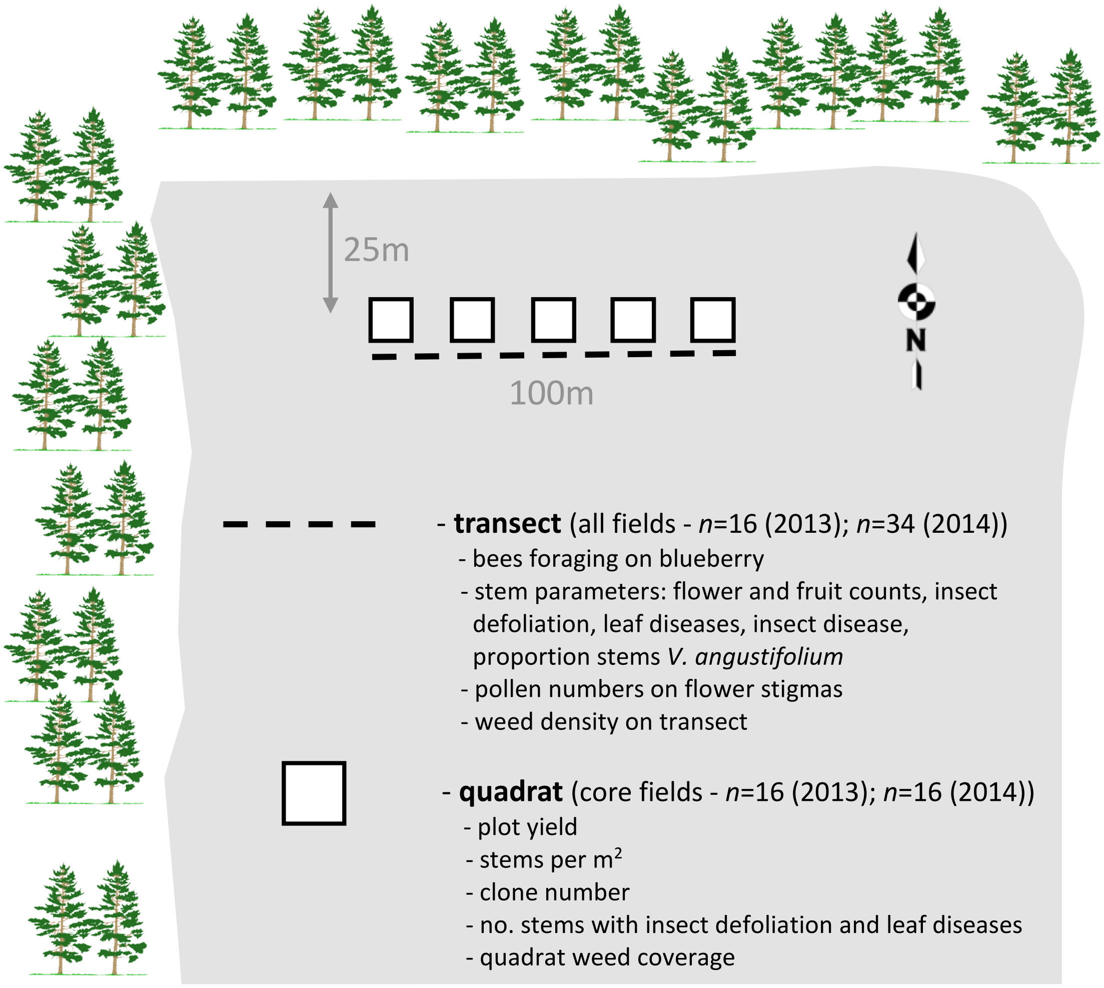


Figure 4.3 Experimental setup at each lowbush blueberry field in Prince Edward Island and Nova Scotia during pollen limitation study with the location of transects, quadrats and the parameters measured at each level.

4.3.2 Stem assessments

Stem assessments were modified from Eaton and Nams (2012) and involved clipping 25 stems at evenly spaced intervals along the transect during four inspection periods. Stems were brought to the laboratory and identified to species (*V. angustifolium* or *V. myrtilloides*), the severity of insect-defoliation and leaf diseases was assessed, and the number of flowers and fruit were counted. Sampling occurred at four dates during the season to determine the average of the following parameters per stem: 1) the starting total number of flowers (early June, early bloom); 2) the number of flowers open to pollinators during peak bloom (mid June); 3) the number of fruit initially set or damaged by diseases (late June - early July, two weeks after the end of bloom); and 4) the number of ripe berries (early-mid August). The last parameter, the number of ripe berries, is a proxy for the marketable yield per stem, though it fails to take into account the number of overripe berries which fall from the plant onto the ground before and during harvest (i.e., shatter) as well as the weight of the berries. In 2014, ripe berries harvested from each stem were weighed fresh and divided by the total number of harvested berries to estimate the weight per berry. The percentage of leaves damaged by insect pests or diseases was scored at each period using an ordinal scale based on the percent damage on the leaves: '0' (0% damage); '1' (>0-30% damage); '2' (>30-60% damage); and '3' (>60-100% damage).

4.3.3 Pollinator community assessment

The number of bees visiting blueberry flowers was assessed twice during peak bloom, with the exception of 18 fields in 2014 that were assessed once. Sampling in 2013 consisted of walking slowly (~3.5 m/min.) for 60 minutes along the transect length, capturing all bees encountered on blueberry flowers with an aerial net. Bees were then either placed in a killing jar if they could not be reliably identified to species visually, or counted and released from the net. In 2014 the sampling period was reduced to 30 min in order to include additional fields. Up to four different people were involved with sampling bees. Measures of bee diversity and abundance were equivalent among observers, as determined by periodic calibration tests when all individuals sampled on the same field. All sampling took place between 10:00 and 16:00 on warm (>16°C) sunny days with <20 km/h winds. Bees that could not be identified in the field were later identified in the laboratory. Counts from different dates and from the same field were averaged for analysis.

4.3.4 Pollen deposition

Variation in pollen deposition among transects at peak bloom in 2014 was estimated from stem samples collected on the days pollinator data was collected. Stems harvested from the fields were tightly bundled and placed into water-saturated vermiculite blocks positioned in an insulated cooler packed with ice for transport from the field to the laboratory. Later in the day, using fine forceps, two stigmas were harvested from each stem (i.e., 50 stigma per transect) and placed on a drop of basic fuchsin gel to facilitate counting pollen tetrads deposited per stigma with bright field light microscopy (Javorek et al., 2002). Each tetrad unites into a single structure four separate pollen grains, each of which can germinate and result in an independent fertilization. Only receptive stigmas were selected for staining, which was discerned visually by the presence of a creamy exudate on the stigma surface. To confirm that no pollen was deposited onto stigmas in transit, stems from control plots excluded from pollinators using a fine mesh (30% shade cloth), were included on each collection date.

4.3.5 Tissue analysis

The 25 stems collected at harvest for fruit set counts (Section 4.3.4) were stripped of all fruit, and the remaining stems and leaves were placed in paper bags and dried at 65°C for 24h before being shipped for analysis of nitrogen (N), phosphorous (P), potassium (K), calcium (Ca), boron (B) and magnesium (Mg) at the Prince Edward Island Department of Agriculture and Forestry Analytical Laboratories (Charlottetown, PEI) in 2013 and the Nova Scotia Department of Agriculture Laboratory Services (Truro, Nova Scotia) in 2014. The levels measured for the pooled sample of 25 stems (expressed in terms of percentage of dry tissue weight) were used to estimate the average across all plants in each transect. In 2014, tissue analysis was additionally done in late June-early July 2014 (initial fruit set) and in July 2013 (sprout phase).

4.3.6 Transect weed cover

The density of weeds on each transect was assessed during initial fruit set (late June - early July). Density was estimated by scoring each 1 m^2 quadrat along the transect (with >50% blueberry plant coverage) along an ordinal scale based on the percent weed coverage with a score of ‘0’ for weed free plots, ‘1’ for weeds growing through >0-25%

of the 1 m^2 quadrat, ‘2’ for weeds growing through >25-75% of the quadrat and ‘3’ for heavily weeded quadrats where weeds were interspersed with blueberry plants over >75% of the quadrat.

4.3.7 Yield quadrat assessment

In addition to estimates derived from the stem clippings along transects (i.e., fruit per stem) (Figure 4.3), the five yield quadrats positioned along each transect provided an additional measure of yield (i.e., weight of ripe harvested fruit per unit area). To avoid confusion, it is important to note that while many of the same measurements made on transects—weed density, insect pest pressure, stem density—were made within quadrats, they involved separate plot-level measurements with different methodology.

The number of stems in each yield quadrat was counted prior to leaf and flower emergence in early May. A few weeks later, boundaries between different *V. angustifolium* clones became distinctly visible as leaves and flowers emerged, making it possible to estimate variation in the number and relative size of clones bordering the quadrats. These estimates were made by counting the number of clones within a 5 m radius of each quadrat (from the center of the quadrat) and providing an approximate size of these clones by measuring their length and width. Although *V. myrtilloides* clones could not always be reliably distinguished, the total area (percent cover converted to m^2) of this species around each clone was also estimated. Yield quadrats were assessed for weed pressure and levels of insect and disease leaf damage in mid-July (i.e., as fruit were ripening). Unlike the transect assessments, weed coverage was visually estimated for each quadrat and the number of stems with >25% of their leaves damaged by insect as well as the number with >25% of their leaves damaged leaf diseases were counted. Immediately prior to commercial harvest, blueberry hand rakes were used to remove all berries from each quadrat and measured with a digital scale. The yield of ripe berries from each yield quadrat was calculated by multiplying the total yield by the ratio of ripe berry weight over total berry weight (e.g., ripe + unripe) measured from a ~ 200 berry subsample.

4.3.8 Data analyses

All analyses were conducted in the R statistical environment (R Development Core Team, 2013). Data from 2013 and 2014 were analyzed separately. The change in the number of potential fruit per stem—calculated as the total number of flowers in early June compared to the final number of ripe fruit per stem in early-mid August—was related to the overall density of bee pollinators using a repeated measures mixed linear model that incorporated 'field' as a random effect, assessment period (bloom, fruit set, or harvest) as a fixed effect, and bee density during peak bloom as a covariate. To account for within-field correlations across time, the repeated measure analysis incorporated an autoregressive function (AR1) (Zuur et al., 2009). Because of the predominance of honey bees and their distinctly lower pollinator efficiency relative to other bee taxa in blueberries (Javorek et al., 2002) and to allow direct comparisons to the findings of Garibalidi et al. (2013), for all other analyses bee density at peak bloom was partitioned into honey bees (*Apis*) and all other species (non-*Apis*).

The next set of analyses examined whether variation in 2014 *Apis* and non-*Apis* visitation during peak bloom explained levels of pollen on blueberry stigmas. The effect of the floral visitation rate of both groups of pollinators (and their interaction) in explaining the total amount of pollen deposited on flowers was fitted to a linear model that contained a covariate adjusting for variation in the density of open flowers at each field during the bee survey. The number of open flowers along the 100 m transect (the covariate) was estimated by multiplying the number of open flowers per stem by the number of stems in a 200 m^2 area, estimated as the number of stems counted in five quadrats, times 40 (i.e., the approximate area covered during a 60 min pollinator assessment).

The absolute amount of pollen counted on transects, however, may not reflect the proportion of stigmas receiving threshold levels of pollen required for enhanced fruit set. Consequently, I reanalyzed the data by fitting *Apis* and non-*Apis* floral visitation rate, the interaction of these factors, and the number of flowers open along the transect to the proportion of stigmas with effectively no (<4 tetrads – reflecting the fact that even control stigmas that were excluded from pollinators periodically had 1-2 tetrads), low (>4-10 tetrads), medium (>10-50 tetrads), or high (>50 tetrads) levels of tetrads during peak bloom using a general linear model with a quasi-binomial distribution (correcting for over-dispersion) and a logit link. Model selection involved dropping the least significant terms as described by Zurr et al. (2009: 220-223), except for the *Apis* and non-*Apis* variables, which were retained in all models regardless of significance. The variance explained by each model was determined from the residual and deviances from the fitted relative to the null model using McFadden's pseudo-r^2 (Faraway, 2005).

A central hypothesis of this paper is that blueberry production is pollen limited. To determine if the pollen deposition was related to fruit set (i.e., the total number of berries set per stem immediately after pollination) and ripe berries per stem at harvest, the relative linear fit of the four levels of pollen deposition described from the previous analysis (no, low, medium, or high) and the number of initial flowers per stem to fruit set was compared using Akaike's information criterion (AIC). Since many of these models had variances that increased along with the number of initial flowers per stem, a generalized least squares (GLS) model was used with a variance structure that was fixed to the number of initial flowers (Zuur et al., 2009: 74-75).

The final set of analyses examined the relative contribution of *Apis* and non-*Apis* pollinator density to fruit set and berry yield relative to other factors such as weed density, the average severity of leaf diseases, the average intensity of insect defoliation and levels of nutrients in plant tissues. Since these analyses involved a large number of potentially multicollinear explanatory variables, all variables were examined using principal component analysis (following varimax rotation) to visualize potentially redundant variables. The full linear model for fruit set contained *Apis* and non-*Apis* densities, their interaction, and a number of variables that potentially explained variation in fruit set. The best model was selected using the Multi-Model Inference model averaging techniques featured in the MuMIn package in R using the dredge function. Model averaging acknowledges that there can sometimes be dramatically different configurations of explanatory variables with relatively similar levels of fit using traditional stepwise methods of model selection and consequently generates an average model using the relative probability of different model configurations (Burnham and Anderson, 2002). The average result for predictor variables for all models was calculated from a subset of models with AICc scores within 2 of the best fitted model ($\Delta i < 2$). The absence of significant multicollinearity was confirmed by estimating variance inflation factors and ensuring they were less than 3.

A linear mixed effect analysis was used to determine the relationship between pollinator densities and various plant parameters on quadrat berry yield. Quadrat measurements were nested within field which was modeled as a random effect. Although flower density, clone diversity, area of adjacent *V. myrtilloides*, weed, pest and disease pressure parameters were measured directly at each quadrat, *Apis* and non-*Apis* density and plant tissue parameters were measured at the level of the transect such that the same values were applied across all five quadrats per field. The full model included *Apis* and non-*Apis* density (and their two-way interaction), the total initial number of flowers per quadrat, the total number of distinct *V. angustifolium* clones, their total area and the area covered by *V. myrtilloides* within a 5 m radius of each quadrat, the percentage area of each quadrat covered by weeds, the number of stems with insect and disease damage and levels of tissue micro nutrients at harvest (2013 and 2014) and at initial fruit set (2014). The manual backward selection method for mixed models described by Zurr et al.(2009: 120-122) was used for model selection.

4.4 RESULTS

There was considerable variation across fields in both the number and diversity of bees visiting blueberry blossoms during peak bloom (Figure 4.4A), but three taxa were most prominent: honey bees, wild bumble bee queens (mainly *Bombus impatiens* and *Bombus ternarius*), and female *Andrena* (Figure 4.4B). Honey bees were used in all but one study field in 2013 and four study fields in 2014, and visitation rates were largely equivalent to the combined visitation rate of all wild bee species (Figure 4.4; 2013 - 51.8 ± 55.8 (± SD) honey bees/h vs. 51.5 ± 22.8 wild bees/h; 2014 – 55.3 ± 43.3 honey bees/h vs. 51.5 ± 16.2 wild bees/h). *B. impatiens* workers, which were assumed to be entirely from managed colonies as wild workers should not have yet emerged, were encountered infrequently (Figure 4.4) in both years despite colonies being deployed in over half of fields at an average stocking rate only slightly below recommended rates of 7-10 colonies/ha (Stubbs and Drummond, 2001) (2014 mean stocking rate; 6.64 ± 2.23 (± SD) colonies/ha, *n* = 12 fields). In contrast, the two fields in 2013 and four fields in 2014 stocked with alfalfa leafcutter bees had relatively high visitation rates by this species compared to honey bees (Figure 4.4).

The number of blueberry pollen tetrads observed on stigmas during peak bloom in 2014 was strongly related to the floral visitation rate of honey bees (*Apis*) and non-*Apis* bee species (Figure 4.5). A significant negative interaction between the visitation rates of these two groups of bees and pollen loads found on stigmas indicates that non-*Apis* species deposited more pollen per bee visit than *Apis* (0.37 vs. 0.19 pollen tetrads per bee, respectively) (Figure 4.5). The increased pollen deposition rate associated with the non-

Apis visitation was also evident when comparing visitation rates to the proportion of stigmas collected from transect with no (<4), low (4-10), medium (10-50), or high (>50) amounts of pollen tetrads. Only the visitation rate of non-*Apis* species, and not *Apis*, was associated with an increasing proportion of stigmas with some pollen (> 4 tetrads) (Table 4.1) but model fit improved when non-*Apis* visitation rate was compared only to the proportion of stigmas with high pollen loads (>50 tetrads) (Table 4.1; $r^2 = 0.41$ for model with proportion of stigmas with > 50 tetrads vs. $r^2 = 0.29$ for model with proportion of stigmas < 4 tetrads; r^2 values for other models < 0.21 and reported in the table). The percentage of stigmas with high pollen deposition (> 50 tetrads) increased by 2.3% with each additional non-*Apis* visitation per hour, although this effect was offset by the total number of flowers blooming per unit area (Table 4.1). Moreover, 2014 fruit set per stem and number of ripe berries per stem was best explained by the proportion of stigmas on a given transect with high pollen deposition (Table 4.2). This model, which also included the number of initial flowers per stem, predicted that a 10% increase in the proportion of stigmas with high pollen loads would result in an additional 3.2 berries set per stem, but only 0.9 harvestable ripe berries (Table 4.2).

In spite of the positive effects of non-*Apis* visitation on pollen deposition in 2014, I observed that fewer than a quarter of the initial flowers per stem in early June resulted in harvestable ripe fruit by August (Figure 4.6). In both years the largest drop in potential fruit occurred immediately following pollination, when 3- (2013) and 2.1-fold (2014) fewer fruit set compared to the starting number of flowers. Yet there were also 1.8- (2013) and 1.9-fold (2014) reduction in harvestable ripe fruit per branch in August compared to the number of set fruit in late June (Figure 4.6). To determine if variation in pollinator visitation rate or other agronomic factors explained these observed patterns of transect fruit set, model selection was applied to the full models outlined in Table 4.3. These full models included concentrations of nutrients in stem and leaf tissue which were potentially multicollinear. Principal component analysis indicated the following nutrients predicted patterns of variation in other nutrients and were then removed in subsequent model refinements (Figure 4.7): a) 2013 harvest - P indicated K and Ca (PCA1) and N (PCA2), B (PCA3) and Mg (PCA4) were modeled as independent parameters; b) 2014 fruit set ('fset') - N indicated P and K (PCA2) and Ca indicated B (PCA4) and Mg (PCA1) was modeled as an independent parameter; c) 2014 harvest – N indicated Mg and Ca (PCA1), B indicated P (PCA3) and K (PCA2-4) was modeled as an independent parameter. Moreover, I found that in Mg at fruit set in 2014 indicated N at harvest, Ca at harvest and Mg at harvest (PCA1) and, consequently, Ca at fruit set was included in the model explaining the number of harvestable berries in 2014.   These nutrients demonstrated a pattern of change relative to pollination and harvest in 2014 with NPK declining and Mg and Ca increasing (Figure 4.8). Boron followed the pattern of Mg and Ca, but declined in tissue immediately after pollination. Levels at harvest in 2013 and 2014 were similar, with the possible exception of P.

Model selection for determinants of fruit set on transects resulted in the following number of models with AICc scores within 2 of the best fitted model ($\Delta i < 2$): 5 (2013 fruit set); 5 (2013 harvestable ripe fruit); 8 (2014 fruit set); 9 (2014 harvestable ripe fruit); 5 (2014 harvestable ripe fruit weight). While the averaged models (Table 4.4) seem to partially confirm the hypothesis that pollinator visitation rate increases initial fruit set and harvestable ripe berries per stem, this increase was: a) not consistent (i.e., significant effect of visitation was observed only in 2014) and b) it was only associated with visitation by non-*Apis* bees, not *Apis* (Table 4.4). Moreover, in both 2013 and 2014 initial fruit set and harvestable berries per stem were explained by factors not associated with visitation rate of either pollinator group (Table 4.4). In 2013, for example, increases in fruit set and harvestable number of fruit per stem were associated with transects in which stems had larger numbers of flowers. Fruit set in 2013 was additionally related to transects with low insect defoliation and weed cover in July, as well as transects that experienced delayed bloom (Table 4.4). Also, in 2014 increases in fruit set and harvestable berries per stem were not only predicted by higher rates of visitation by non-*Apis* pollinators, but also higher average flower counts per stems (fruit set and ripe berries per stem) and magnesium levels (multicollinear with nitrogen, calcium and magnesium levels in tissue at harvest, Figure 4.7) in stem and leaf tissue at fruit set (ripe berries per stem only) (Table 4.4). Significantly, while non-*Apis* visitation was related to 2014 fruit set and harvestable berries per stem, it was not retained in any of the best models predicting the average weight of berries (Table 4.4). Instead, higher berry weight in 2014 was associated with low levels of leaf disease, elevated level of blossom disease, increases in tissue boron at harvest (multicollinear with phosphorus at harvest, Figure 4.7) and decreased levels of nitrogen at fruit set (multicollinear with phosphorous and potassium at fruit set, Figure 4.7).

The importance of factors other than pollinator visitation was even more pronounced among models explaining overall yield in quadrat plots. Neither *Apis* nor non-*Apis* visitation parameters were retained in any of the best models predicting weight of hand-harvested ripe berries per plot (Table 4.5). Moreover, factors related to plot yield were different than those explaining fruit set and harvestable berries per stem on transects, with the number of stems with moderate leaf disease levels associated with lower plot yields in 2013 and, in 2014, the initial number of flowers per plot was associated with higher yields whereas higher levels of magnesium in tissue at fruit set (multicollinear with nitrogen, calcium and magnesium at harvest, Figure 4.7) was associated with lower yields (Table 4.5).

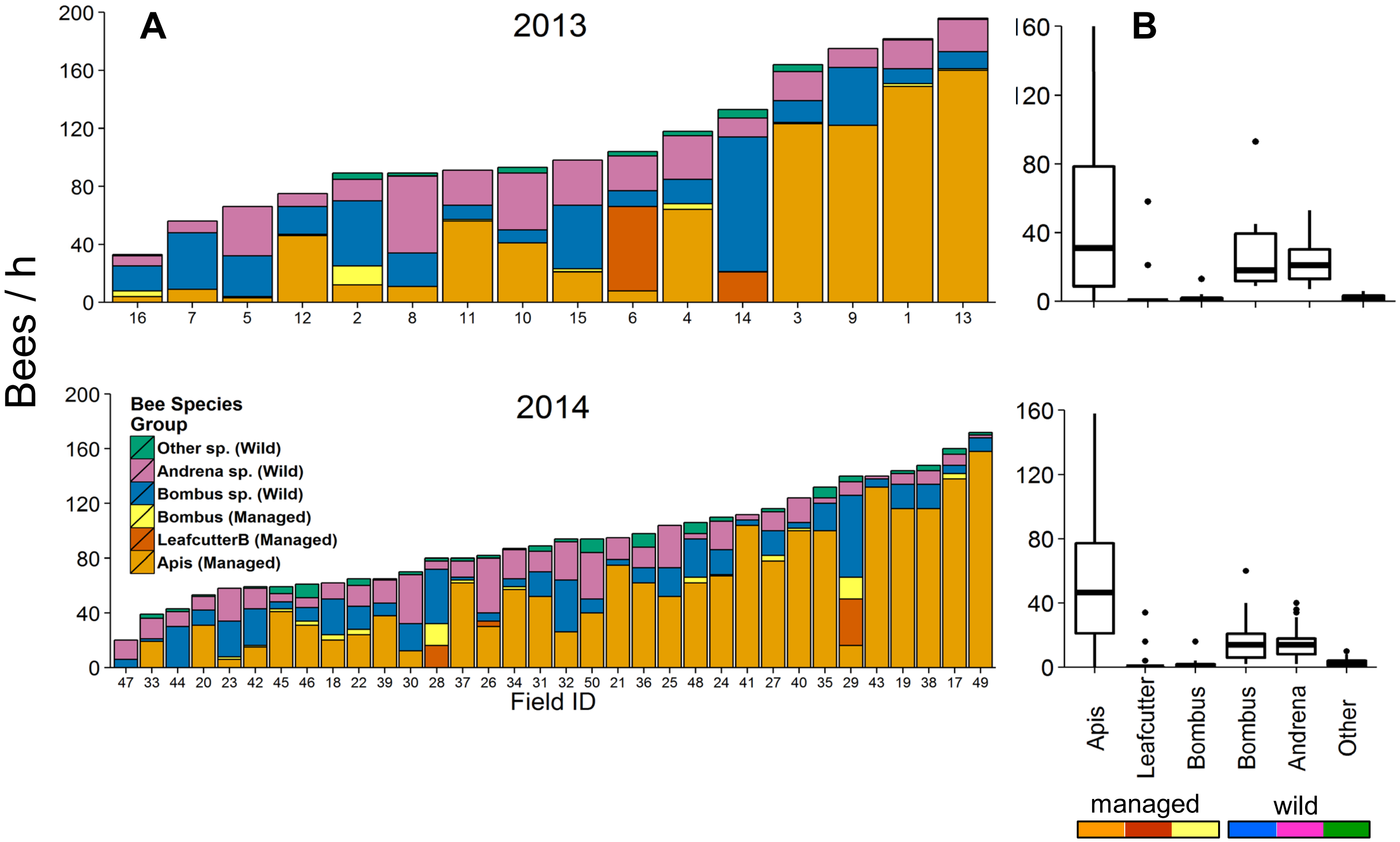
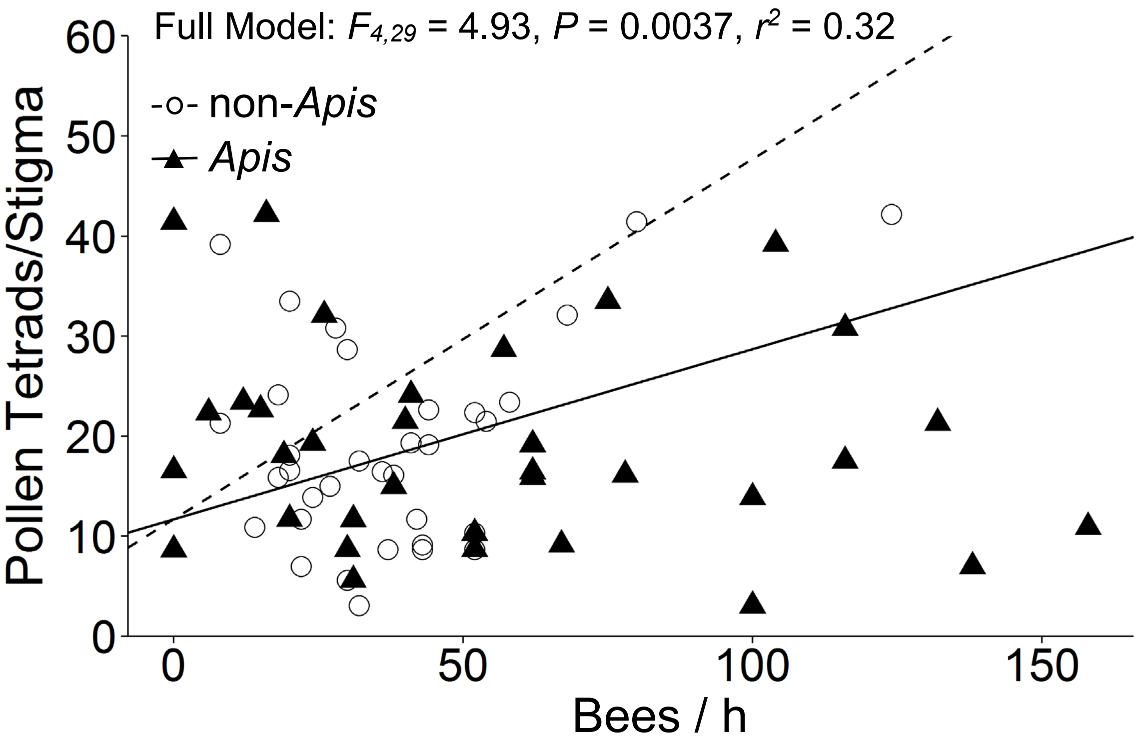
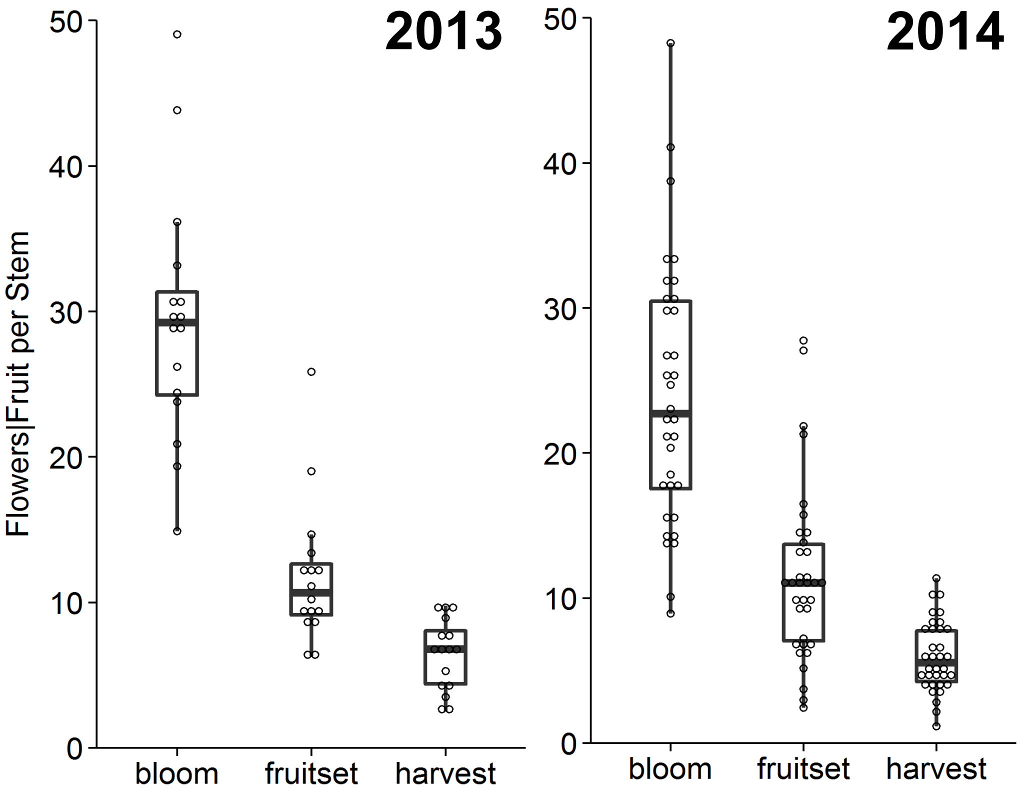


Figure 4.4 The density of bees encountered foraging on blueberry plants over one hour along transects at peak bloom in 2013 (*n*=16 fields) and 2014 (*n*=34 fields) across field sites in Prince Edward Island and Nova Scotia. The number of bees at each field is broken into six species groups (three groups of managed species and three groups of wild species) by field (A) or across all fields in a given year using box and whisker plots (B). In box and whisker plots the horizonal line within each box indicates the median, lower and upper box boundaries the 25 and 75th percentiles, respectively, and whiskers the most extreme data points that are no more than 1.5 times the length of the box.



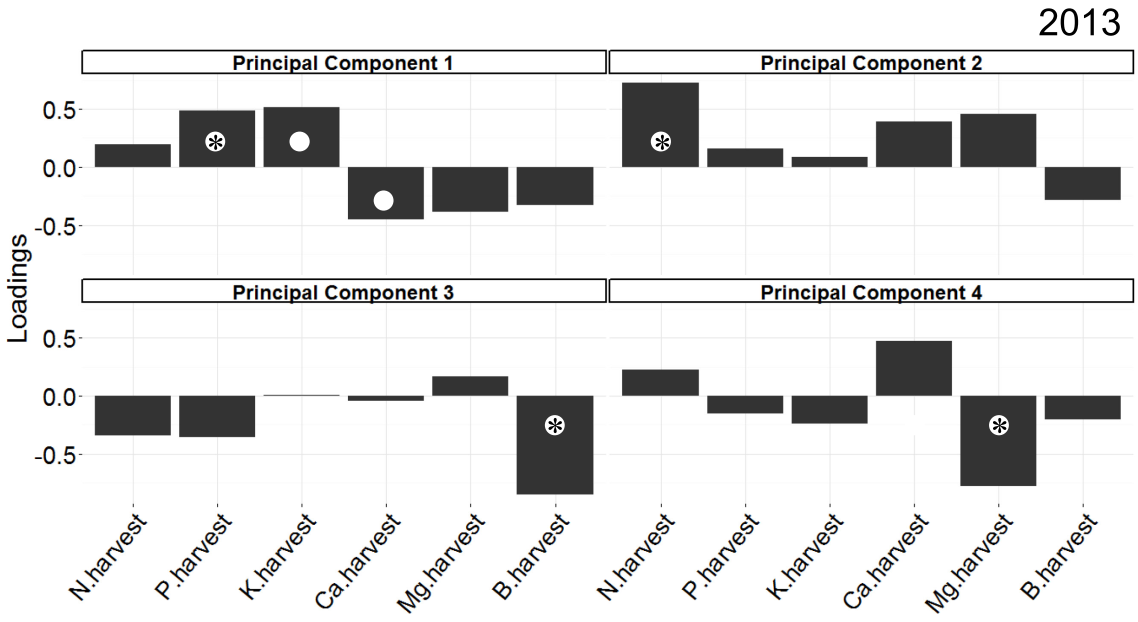
Predictor variable	**Estimate**	**± SE**	***t***	***P***
(Intercept)	11.7	5.29	2.20	**0.0344**
Apis (bees/h)	0.17	0.07	2.60	**0.0144**
non-*Apis* (bees/h)	0.36	0.09	3.85	**0.0006**
Apis × non-*Apis*	-5.4×10^{-3}	2.4×10^{-3}	-2.23	**0.0316**
flowers open on transect	-1.1×10^{-5}	4.3×10^{-6}	-1.91	0.0662

Figure 4.5 A linear model explaining the number of pollen tetrads on blueberry stigmas extracted from flowers along transects at peak bloom in 2014 (*n*=34 fields) by the number of honey bees (*Apis*) or bees of other species (non-*Apis*: including all wild bees as well as managed bumble bees and leafcutter bees) across field sites in Prince Edward Island and Nova Scotia. The full model included all two and three-way interactions but only the interaction between the two groups of bees was retained in the final model due to a lack of significance of other factors (P<0.05). Parameter estimates (± SE) are for the two bee groups along with a parameter representing the number of flowers open along the transect. Significant estimates (*t*-test) are bolded in the table.



	2013				2014			
predictor variable	**estimate**	**±SE**	***t***	***P***	**estimate**	**±SE**	***t***	***P***
(intercept)	29.17	1.41	6.22	<0.0001	24.81	1.12	22.09	<0.0001
fruitset	-17.42	1.40	-12.43	<0.0001	-13.29	1.30	-10.23	<0.0001
harvest	-22.72	3.77	-15.86	<0.0001	-18.84	1.32	-14.29	<0.0001

Figure 4.6 Box and whisker plot of the number of flowers or flowers per stem per field (2013 *n*=16, 2014 *n*=34) across sites in Prince Edward Island and Nova Scotia. The horizonal line within each box indicates the median, boundaries the 25 and 75th percentiles, and whiskers the most extreme data point that is no more than 1.5 times the length of the boxhighest and lowest values of the results. The average value per stem at each field is indicated by a dot. Flowers were counted at the beginning of the bloom period (early June) and fruit at initial fruit set (late June) or harvest (early August). Parameter estimates for fixed effects in a repeated measures mixed effects model (field as a random effect) appear below – SE represents the standard error of the parameter estimate and *t* the *t*-test statistic.



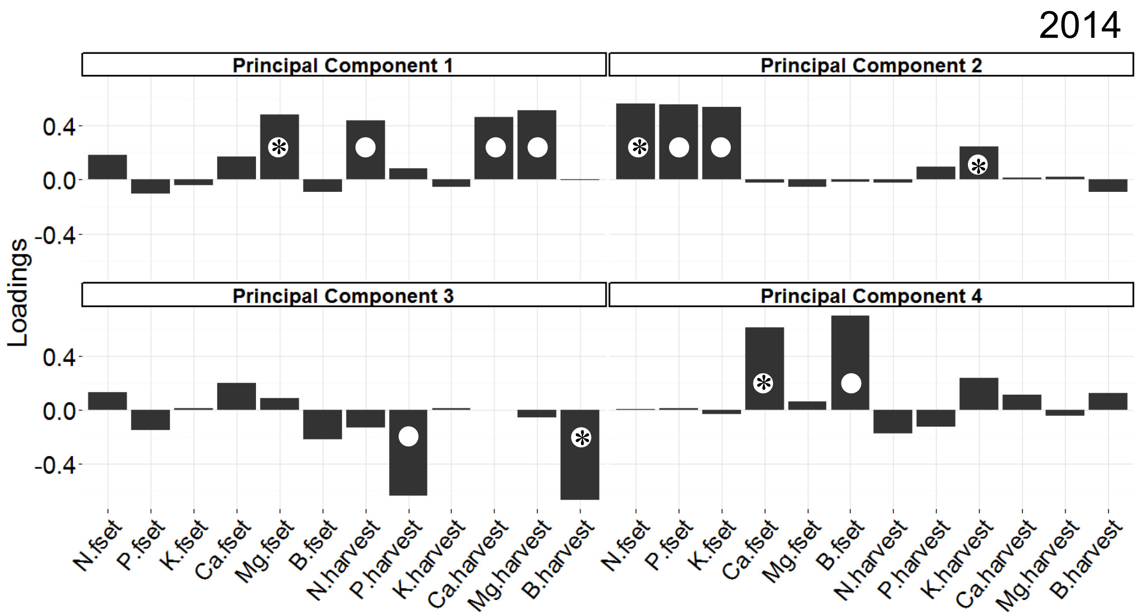


Figure 4.7 Varimax-rotated loadings from principal component analysis used in candidate models to explain variation in fruit set (Table 4.3) across field sites in Prince Edward Island and Nova Scotia. Results are restricted to the first four principal components and plant tissue parameters marked with white dots indicate co-variation among parameters. In order to reduce multicollinearity, only the white-dotted micronutrient parameters marked with asterisks were included in subsequent candidate models.

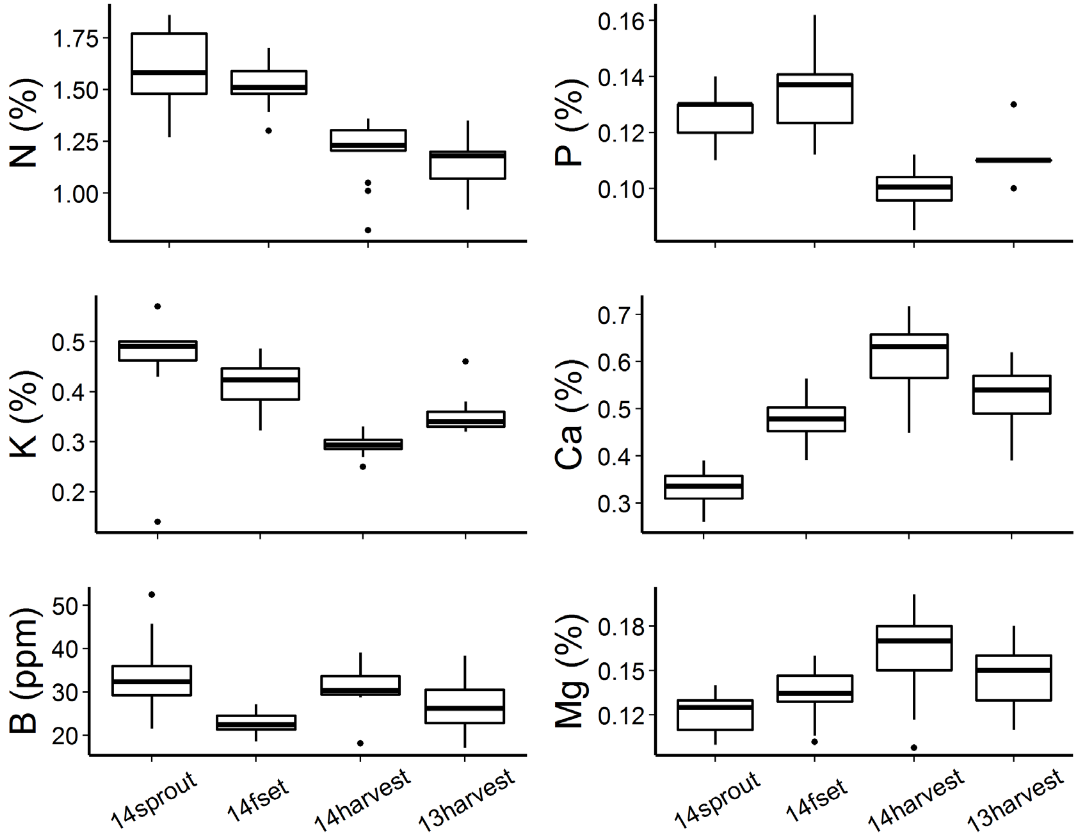


Figure 4.8 Box and whisker plots of transect-level macro and micronutrient concentrations in stem and leaf tissue collected from lowbush blueberry fields in Prince Edward Island and Nova Scotia in 2013 and 2014. Tissue samples in 2013 were only collected immediately prior to harvest (‘13harvest’; *n*=16). In 2014, samples were collected the July the year previous to harvest (‘14 sprout’; *n*=14), in late June-early July after fruit set (‘14fset’; *n*=34), or immediately prior to harvest (‘14harvest’; *n*=34). The horizontal line signifies the median field yield, lower and upper box boundaries the 25 and 75th percentiles, respectively, and whiskers the most extreme data points that are no more than 1.5 times the length of the box.

Table 4.1 Results of general linear models (GLM – logit link) explaining the proportion of stigmas with no (<4), low (4-10), medium (10-50), or high (>50) amounts of pollen tetrad deposition during peak bloom of lowbush blueberry fields in Prince Edward Island and Nova Scotia in 2014 (n=34 fields). The overall significance of the model is represented by a chi-square (X^2) test and model fit was estimated using McFadden's pseudo-r^2. Parameter estimates (± SE) and t-test statistics for slopes (bold where P<0.05) are reported.

Response variable	Predictor variable[a]	Estimate	± SE	*t*	*P*
<4 tetrads	(intercept)	-0.234	0.299	-0.78	0.4390
$X^2_{30} = 0.9999$	*Apis* (bees/h)	-0.004	0.003	-1.46	0.1547
$\mathbf{P < 0.0001}$	non-*Apis* (bees/h)	-0.011	0.005	-2.11	**0.0438**
$r^2 = 0.29$	flowers open on transect	1.08×10^{-6}	3.33×10^{-7}	3.26	**0.0028**
4-10 tetrads	(intercept)	-1.752	0.311	-5.63	**0.0001**
$X^2_{31} = 0.8539$	*Apis* (bees/h)	7.63×10^{-5}	0.002	0.03	0.976
$P = 0.1461$ $r^2 = 0.08$	non-*Apis* (bees/h)	-0.008	0.005	-1.41	0.169
10-50 tetrads	(intercept)	-0.42	0.274	-1.53	0.1358
$X^2_{30} = 0.9928$	*Apis* (bees/h)	0.002	0.002	0.89	0.3810
$\mathbf{P = 0.0072}$	non-*Apis* (bees/h)	8.85×10^{-4}	0.004	0.20	0.8444
$r^2 = 0.21$	flowers open on transect	8.61×10^{-7}	3.13×10^{-7}	-2.75	**0.0099**
>50 tetrads	(intercept)	-2.460	0.436	-5.65	**0.0001**
$X^2_{30} = 0.9986$	*Apis* (bees/h)	0.003	0.004	0.78	0.4446
$\mathbf{P < 0.0014}$	non-*Apis* (bees/h)	0.023	0.006	4.10	**0.0003**
$r^2 = 0.41$	flowers open on transect	-11.4×10^{-7}	5.4×10^{-7}	-2.13	**0.0417**

a - Models represent the best fitting model of three parameters and their two-way interactions: *Apis* (the number of honey bees visiting blueberry flowers on transects), non-*Apis* (the number of other bees, including wild bees and managed *Bombus impatiens* and leafcutter bees), and the number of flowers open along the transect (flowers). During model selection, *Apis* and non-*Apis* were retained, but interactions and flowers were deleted if they did not significantly contribute to fit.

Table 4.2 A comparison of fit and estimate of predictor variables for models explaining the number of lowbush blueberry fruit per stem in late June (initial) or ripe berries per stem at harvest in early August 2014 (n=34 fields) across field sites in Prince Edward Island and Nova Scotia. All models included the number of initial flowers per stem ('flowers' – counted in early June), and either: *pollen* (the total number of pollen tetrads per stigma), *none* (the proportion of stigmas with <4 tetrads per stigma), *low* (the proportion of stigmas with ≥ 4 or <10 tetrads), *med* (the proportion of stigmas with ≥10 or <50 tetrads) or *high* (the proportion of stigmas with ≥ 50 tetrads). Fit is expressed in terms of the Akaike's information criterion (AIC) and the Δ column depicts the difference between a model's AIC and that of the best-fitting model. SE represents the standard error of the parameter estimates. Significance of *t*-test for each parameter are as follows: ns, $P > 0.05$; *, $P \leq 0.05$; **, $P \leq 0.01$; ***, $P \leq 0.001$; ****, $P \leq 0.0001$.

Model	**AIC**	**Δ**	**Pollen parameter estimate**	**± SE**	**Flower parameter estimate**	**±SE**	**Intercept**	**±SE**
				FRUIT SET PER STEM				
flowers	212.31	20.13	-	-	0.37^{***}	0.10	2.33	2.19
pollen + flowers	208.12	15.94	0.24^{**}	0.07	0.41^{****}	0.08	-3.31^{ns}	2.58
none + flowers	199.37	7.19	-17.83^{**}	5.36	0.48^{**}	0.09	7.74^{ns}	2.51
low + flowers	204.12	11.94	-24.14^{ns}	13.43	0.33^{***}	0.10	6.04^{ns}	2.96
med + flowers	206.14	13.96	11.51^{ns}	7.56	0.41^{***}	0.10	-2.39^{ns}	3.77
high + flowers	192.18	0	31.85^{****}	6.99	0.39^{****}	0.08	-1.89^{ns}	1.96
				RIPE BERRIES PER STEM				
flowers	162.09	7.42	-	-	0.09^{*}	0.04	3.68^{***}	1.14
pollen + flowers	165.96	11.29	0.07^{ns}	0.04	0.11^{*}	0.04	2.04^{ns}	1.48
none + flowers	158.66	3.99	-3.56^{ns}	3.04	0.11^{*}	0.05	4.86^{*}	1.52
low + flowers	154.93	0.26	-12.98^{ns}	6.87	0.08^{ns}	0.04	5.33^{**}	1.41
med + flowers	159.29	4.62	2.06^{ns}	3.81	0.10^{*}	0.04	2.90^{ns}	1.85
high + flowers	154.67	0	9.08^{*}	4.06	0.11^{*}	0.04	2.20^{ns}	1.27

Table 4.3 Biological interpretation of predictors used in candidate models to predict fruit set (late June) and ripe berries per stem at harvest (early August) along study transects in Prince Edward Island and Nova Scotia. Parameters used in the full models to explain fruit set in 2013 (13f) and 2014 (14f) and ripe berries at harvest in 2013 (13h) and 2014 (14h) are indicated by checkmarks. Mean and standard deviation (SD) for all parameters used in models for 2013 (*n*=16) and 2014 (*n*=34) are included.

Predictor variables	**13f**	**13h**	**14f**	**14h**	**Biological interpretation (hypothesis)**	**2013 (mean ± SD)**	**2014 (mean ± SD)**
Apis (bees/h)	✓	✓	✓	✓	Increase fruit set because of pollination activity.	51.8/h ± 14.0	55.3/h ± 7.4
non-*Apis* (bees/h)	✓	✓	✓	✓	Increase fruit set because of pollination activity.	58.3/h ± 7.0	37.7/h ± 3.9
Apis × non-*Apis*	✓	✓	✓	✓	A positive interaction indicates a synergetic influence between these groups. A negative interaction suggests that the presence of additional pollinators decreases the effect of those already present (i.e., pollinator saturation).	na	na
flowers per stem	✓	✓	✓	✓	Increase fruit set because low initial flower number preclude possibility of high yields.	29.4/stem ± 9.1	24.0/stem ± 9.1
diseased flowers	✓	✓	✓	✓	Decrease in potential fruit at pollination or shortly after because of monilinia and botrytis blights.	0.2/stem ± 0.2	0.3/stem ± 0.5
bloom phenology	✓	✓	✓	✓	Increase fruit set at harvest for fields that bloom earlier.	73.4% ± 4.8	70.7% ± 2.4
nitrogen (June)			✓	✓	Macronutrient – optimal levels required for fruit set.	-	1.5% ± 0.1
nitrogen (August)		✓		✓	Macronutrient – optimal levels required for fruit set.	1.1% ± 0.1	1.2% ± 0.2
phosphorous (June)					Macronutrient – optimal levels required for fruit set.	-	0.12% ± 0.02

Table 4.3 *continued*

Predictor variables	**13f**	**13h**	**14f**	**14h**	**Biological interpretation (hypothesis)**	**2013 (mean ± SD)**	**2014 (mean ± SD)**
phosphorous (August)		✓			Macronutrient – optimal levels required for fruit set.	0.11% ± 0.01	0.09% ± 0.01
potassium (June)					Macronutrient – optimal levels required for fruit set.	-	0.40% ± 0.05
potassium (August)				✓	Macronutrient – optimal levels required for fruit set.	0.37% ± 0.05	0.29% ± 0.01
boron (June)			✓	✓	Micronutrient – increased levels associated with higher pollination efficacy.	-	24.1ppm ± 6.3
boron (August)		✓		✓	Micronutrient – increased levels associated with higher pollination efficacy.	26.6ppm ± 5.8	32.6ppm ± 8.0
magnesium (June)			✓	✓	Micronutrient – optimal levels required for fruit set.	-	0.14% ± 0.01
magnesium (August)		✓			Micronutrient – optimal levels required for fruit set.	0.14% ± 0.02	0.17% ± 0.03
insect defoliation	✓	✓		✓	Decrease fruit set because of reduced photosynthetic capacity.	0.12/stem ± 0.2	0.13/stem ± 0.2
weed cover	✓	✓		✓	Decrease fruit set because competition for moisture.	0.84 ± 0.40	0.68 ± 0.54
leaf diseases	✓	✓		✓	Decrease fruit set because of reduced photosynthetic capacity.	0.51/stem ± 0.4	0.64/stem ± 0.4
V. angustifolium (%)	✓	✓		✓	Increase fruit set since it indicates lower levels of the incompatible species *V. myrtilloides*.	96.0% ± 4.2	97.3% ± 4.3

Table 4.4 Results of a linear model explaining lowbush blueberry fruit set per stem (late June), and ripe berries per stem (early August) in 2013 (*n*=16 fields) and 2014 (*n*=34 fields) across field sites in Prince Edward Island and Nova Scotia. A model explaining the weight of ripe berries was also included in 2014. Parameters were selected from full models using predictor variables listed in Table 4.3. Multi-Model Inference techniques were used for model selection using the small-sample bias-corrected Akaike information criterion (AICc). The average result for predictor variables for all models with AICc scores within 2 of the best fitted model ($\Delta i < 2$), as well as their weighted importance (importance), estimate, and adjusted standard error (SE) are listed. *z*-test statistics and *P*-values are reported. *P*-values <0.05 are bolded.

Predictor variables	**Importance**	**Estimate (model averaged)**	**± SE**	***z***	***P***
		2013			
		FRUIT SET PER STEM			
(Intercept)	1.00	-1.08	6.41	0.17	0.8660
flowers per stem	1.00	0.52	0.13	4.01	**<0.0001**
insect defoliation (intensity score)	0.68	-13.31	6.62	2.01	**0.0445**
non-*Apis* density (bees/h)	0.42	0.05	0.03	1.79	0.0741
bloom phenology (% bloom)	0.23	-10.37	4.38	2.37	**0.0180**
weed cover (average score)	0.23	-20.19	8.50	2.38	**0.0175**
		RIPE BERRIES PER STEM			
(Intercept)	1.00	0.79	1.96	0.40	0.6882
flowers per stem	1.00	0.18	0.06	2.87	**0.0041**
Apis density (bees/h)	0.56	0.02	0.01	1.69	0.0904
leaf disease (intensity score/stem)	0.24	-1.61	1.37	1.18	0.2388
non-*Apis* density (bees/h)	0.13	0.03	0.02	1.29	0.1981

Table 4.4 *continued*

Predictor variables	**Importance**	**Estimate (model averaged)**	**± SE**	***z***	***P***
		2014			
		FRUIT SET PER STEM			
(Intercept)	1.00	-19.07	21.78	0.86	0.3907
flowers per stem	1.00	0.32	0.10	3.02	**0.0025**
non-*Apis* visitation (bees/h)	1.00	0.10	0.04	2.51	**0.0119**
bloom phenology (% bloom)	0.52	9.64	5.67	1.63	0.1028
Ca/B (% tissue at fruit set)	0.45	14.44	8.69	1.59	0.1113
V. angustifolium (%)	0.39	32.78	22.16	1.42	0.1561
Mg (% tissue at fruit set)	0.06	60.65	50.30	1.16	0.2479
Apis visitation (bees/h)	0.05	0.03	0.03	1.07	0.2838
non-*Apis* × *Apis* visitation (bees/h)	0.05	0.0001	0.0001	1.82	0.0682
N/P/K (% tissue at fruit set)	0.05	-7.52	5.85	1.23	0.2182
weed cover (average score)	0.05	1.80	1.78	0.96	0.3347
		RIPE BERRIES PER STEM			
(Intercept)	1.00	5.99	4.63	1.27	0.2057
non-*Apis* density (bees/h)	0.94	0.04	0.02	2.45	**0.0141**
Mg (at fruit set) / N/Ca/Mg (at harvest) (% tissue)	0.75	-56.97	24.67	2.22	**0.0262**
B/P (% tissue at harvest)	0.63	0.10	0.05	1.96	0.0500
flowers per stem	0.43	0.08	0.05	1.68	0.0936
leaf disease (intensity score/stem)	0.31	1.45	0.84	1.66	0.0962
bloom phenology (% bloom)	0.30	4.34	2.95	1.42	0.1564
weed cover (average score)	0.09	-1.56	0.83	1.79	0.0732
diseased flowers (flowers/stem)	0.06	0.95	0.81	1.13	0.2602

Table 4.4 *continued*

Predictor variables	**Importance**	**Estimate (model averaged)**	**± SE**	***z***	***P***
		2014			
		WEIGHT PER RIPE BERRY			
(Intercept)	1.00	1.31	0.59	2.16	**0.0307**
B/P (% tissue at harvest)	1.00	0.01	0.00	2.28	**0.0224**
leaf disease (intensity score/stem)	0.72	-0.10	0.04	2.20	**0.0275**
V. angustifolium (%)	0.30	-0.96	0.50	1.83	0.0673
N/P/K (% tissue at fruit set)	0.71	-0.35	0.16	2.18	**0.0295**
diseased flowers (flowers/stem)	0.72	0.11	0.04	2.62	**0.0089**
Mg (at fruit set) / N/Ca/Mg (at harvest) (% tissue)	0.75	2.29	1.36	1.62	0.1052
flowers per stem	0.62	0.01	0.00	2.53	0.0115
weed cover (average score)	0.09	-0.09	0.04	2.08	0.0377
insect defoliation (intensity score)	0.38	0.10	0.08	1.10	0.2694

Table 4.5 Results of a mixed effects model (field as a random effect) explaining final yield of lowbush blueberry across five 1m^2 quadrats located along transects in 2013 (*n*=16 fields) and 2014 (*n*=16 fields) across field sites in Prince Edward Island and Nova Scotia. Models listed were arrived through a backward step model selection procedure. The full model is listed in the text. Parameter estimates (± SE) for repeated measure fixed effects, *t*-test statistic, and *P*-vlaues are reported. *P*-values <0.05 are bolded.

Predictor variables	**Estimate**	**± SE**	***t***	***P***
	2013			
(Intercept)	0.93	0.11	8.16	**<0.0001**
leaf disease (stems/plot)	-5.3×10^{-5}	2.4×10^{-5}	-2.21	**0.0304**
	2014			
(Intercept)	2.14	0.62	3.48	**0.0014**
flowers per plot	16.5×10^{-6}	5.8×10^{-6}	2.83	**0.0238**
Mg (at fruit set) / N/Ca/Mg (at harvest) (% tissue)	-1.27	0.50	-2.55	**0.0403**

4.5 DISCUSSION

Lowbush blueberry is completely dependent on visitation by insect pollinators to set fruit (Cutler et al., 2012a; Lee, 1958) and yet my findings show that variation in bee visitation rate could not entirely explain patterns of fruit set and did not explain final plot yield. This finding challenges the general assumption behind current valuation of pollinator services (e.g., Calderone, 2012; Gallai et al., 2009; Lautenbach et al., 2012), in which the entire value of crops like lowbush blueberry (i.e., crops in which pollination is entirely mediated through insects) is attributed to pollinator visitation. Although there was no evidence to support the hypothesis that variation in pollinator activity is the primary determinant of fruit set and final yield, the failure to reject this hypothesis was not because of the dominance of honey bees, which is associated with high pollen deposition but low fruit set in other crops (Garibaldi et al., 2013). My findings show that in spite of high rates of visitation by managed honey bees, it was largely matched by visitation by wild pollinating bees that are adapted to pollinating this crop. Moreover, while species other than honey bees (non-*Apis*) were more strongly associated with lowbush blueberry fruit set, this association was explained by a quantitative increase in the proportion of stigmas with high rates of pollen deposition (>50 tetrads/flower), rather than any qualitative difference between pollinator species. Combined, these results confirm the findings from Chapter 3 and suggest the benefit of insect pollinators to lowbush blueberry yields should be contextualized within a commercial agronomic setting.

The problem of pollen quality in lowbush blueberry is largely associated with the peculiar characteristics of the plant's growth habit and mating systems. Lowbush blueberry spreads laterally through underground rhizomes resulting in 0.5-10 m wide areas of stems from largely one plant (syn. clone) (Vander Kloet 1988). Moreover, most clones are self-incompatible such that if pollinators restrict their visitations within clones much of the pollen will not set a fruit (Bell et al., 2009b). Significantly, another study was also unable to demonstrate a consistent relationship between different combinations of managed and wild pollinators and harvested lowbush blueberry fruit per stem, even though yields increased when outcrosses were performed by hand pollination (Fulton, 2012). The ability to increase berry numbers by experimental outcrossing but not by the abundance of pollinator taxa implies the importance of pollen quality (i.e., outcrossed pollen) in limiting yield, but also that the existing pollinator fauna is unable to adequately optimize outcrossing during commercial production. This hypothesis is supported by the observations of Fulton (2012), in which managed honey bees and bumble bees were shown to be equally poor at outcrossing, largely restricting their foraging trips within fewer than two distinct clones. These findings suggest that poor pollen quality, owing to inadequate outcrossing, is why pollinator visitation in my study did not predict fruit set (2013) and yield (2013 and 2014). Yet, while I observed considerable variation in clonal diversity within a radius of 5 m from the center of my yield plots, ranging from 5-17 morphologically distinct *V. angustifolium* clones, this factor was not retained during model selection for plot yield. The lack of significance of clonal density factors in my model, consequently, suggest that variation in clonal diversity cannot fully account why pollinator visitation did not explain variation in plot yield.

My results from 2014 suggest that under certain conditions pollinator visitation rate can increase the numbers of berries per stem. This effect appears associated with large amounts of pollen deposited per stigma. Such a result may not have been detected by Fulton (2012) and Garibaldi et al. (2013) as both these studies restricted their analyses to the proportion of stigmas in lowbush blueberry with at least 4 or 5 tetrads per stigma. Although I was also unable to relate non-*Apis* abundance to the proportion of stigmas with <4 tetrads (i.e., the inverse of Garibaldi et al. and Fulton, >4 or 5 tetrads), non-*Apis* pollinator visitation rate better explained initial fruit set (but not harvestable fruit) when the threshold was raised to 50 tetrads. Since *V. angustifolium* has an average of 64.2 ovules per flower (Bell, 1957), pollen stigma loads with >50 tetrads exceed by more than 3-fold the number of pollen grains needed to fully-fertilize an average flower. Such high pollen loads may function to overcome pollen incompatibility. For example, maximal fruit set in highbush blueberry (*Vaccinium corymbosum*) was shown to occur when there was 5-fold more pollen grains on stigmas relative to ovules, which was found to diminish the benefits of outcrossing (Dogterom et al., 2000). Since most wild bees, particularly *Bombus* spp. and *Andrena* spp., deposit more pollen per visit than honey bees (Javorek et al., 2002), the significance of non-*Apis* pollinators in 2014 was probably from their ability to deliver more pollen per visit rather than behaviors that increase outcrossing. If this were the case, it would constitute a deviation from the more general pattern described by Garibaldi et al. (2013) in their meta-analysis of pollinator efficacy in agricultural systems. Their study demonstrated that while variation in honey bee visitation is more strongly associated with pollen counts on stigmas compared to visitation by non-*Apis* species, variation in non-*Apis* pollinators better explains patterns of fruit set, suggesting that fruit set is determined by qualitative (e.g., pollen compatibility) rather than quantitative factors (e.g., number of pollen grains on stigmas). My findings suggest the opposite may be true in lowbush blueberries and that apparent problems of pollen quality (e.g., pollen incompatibility) and fruit set may sometimes be better explained as a failure to attain a threshold amount of pollen.

While my findings suggest that in 2014 only non-*Apis* species were associated with an increased proportion of stigmas with high pollen deposition, it is unclear to what extent this could be mitigated using managed non-*Apis* species. Very few commercial fields in our study, as well as few fields in general, use alfalfa leafcutter bees, making it difficult to draw inferences on the effectiveness of this species. Although managed *B. impatiens* colonies are commonly used, their visitation rate in all fields in my study were far lower than that for wild *Bombus* queens. My work suggests more research is needed to develop strategies for the effective implementation of managed *B. impatiens* colonies in Atlantic Canada, as has been done for Maine (Stubbs and Drummond, 2001) and Quebec (Desjardins and De Oliveira, 2006). Significantly, one producer (Field Id 2, 28 and 29, Fig. 4.4) had consistently higher rates of *B. impatiens* visitation, suggesting the possibility of identifying better practices for utilizing this species.

A key finding of this study is that while variation in pollinators explained fruit set in 2014, this effect was not observed in 2013 (*P*=0.07 fruit set for non-*Apis*, Table 4.4) nor to final yield of ripe berries in quadrat plots in either year. My findings agree with other recent work demonstrating pollination benefits to be conditioned by other agronomic factors including levels of fertilizer application (Klein et al., 2015; Marini et al., 2015), the availability of soil moisture (Boreux et al., 2013; Klein et al., 2015) and the efficacy of pest management (Chapter 3, but also Bos et al., 2007a; Lundin et al., 2013). Moreover, my findings from 2013 suggest in some contexts pollen limitation may not even be a predominant factor in explaining observed variation in yield. Although the lack of pollen limitation may seem counter-intuitive—given that lowbush blueberry is entirely pollinator dependent—my findings may simply reflect that in any given year variation in pollinator abundance is not always the predominant driver of variation in yield. Even in a year like 2014 where variation in non-*Apis* pollinator visitation explained variation in fruit set, these benefits were conditioned by the number of flowers per plant and the negative effect of excessive magnesium at fruit set. Moreover, the effect of non-*Apis* pollinator abundance on fruit per stem in 2014 did not ultimately explain variation in yield at the level of plots (quadrats). My findings support the valuation framework recently proposed by Hanley et al. (2015), which discriminates between the capacity of the plant to bear fruit and pollinator effects, but also generates significant challenges for such a framework, since it indicates considerable flux not only between pollen and plant limitations within the same crop and within the same region, but between two different years.

One unexpected finding was that the predictor variables explaining the final ripe fruit per stem were not the same as those identified for quadrat plot yields in both years.

For example, while the significant variables explaining ripe fruit per stem in 2014 included both non-*Apis* pollinator visitation and a leaf tissue micronutrient (magnesium at fruit set, which was multicollinear with nitrogen, calcium and magnesium levels in tissue at harvest) (Table 4.4), the harvested yield was best explained by a model in which the effect of the same micronutrient remained, but non-*Apis* visitation was excluded and the number of initial flowers in each yield plot was included (Table 4.5). One key explanation for this seeming discrepancy is that the weight of ripe berries in 2014, unlike fruit set, was explained by a number of agronomic factors and was independent of non-*Apis* pollinator visitation rate. Consequently, healthy plants may be better able to compensate for low fruit set by increasing fruit weight, as described in Chapter 3. Another possible explanation is that while transect fruit set measured the number of *potentially harvestable* fruit, quadrat yield plots measured the number of fruit that were *actually* harvested. It is possible that pollen visitation rate and agronomic factors may work independently or interact with one another to affect the rate of fruit ripening, which in turn, may contribute to fruit drop onto the ground prior to or during harvest (i.e., shatter). Transect fruit set counts would likely be more insensitive to such losses than yield measurements from harvested plots because there is less risk of dislodging berries from stems during harvest. Other explanations of this discrepancy include the fact that: a) plot data were only available for a subset of the 2014 fields from which stem data were collected; b) many agronomic parameters were measured differently for the stem data as compared to quadrat plots (e.g., weed pressure for the stem data was in terms of the average transect weed score versus the percentage weed cover for quadrat plots); and c) multiple quadrats per field enabled me to estimate the random field error term in a way that was not possible for the stem data. Consequently, I caution against interpreting these results in terms of the importance of any single factor in determining yield, but rather, in terms of demonstrating the contextual basis of pollination benefits as a whole.

My findings do, however, suggest some characteristics of the agronomic context through which pollination benefits were realized in this study. Significantly, a factor explaining fruit set in all the models (with the exception of 2014 harvestable berries per stem) was the starting number of flowers per stem in early June. In general, more flowers per stem resulted in more fruit. Significantly, flower numbers are strongly correlated with the number of floral buds set by plants the previous year (sprout phase), which can be managed by the combined activity of nitrogen fertilization and weed control (Penney and McRae, 2000). Consequently, an important line of future research would be to examine the yield response to low and high pollinator visitation rates in plots with few or abundant flowers per stem. Plots with varying flower density could be established by using or withholding sprout-year herbicide and fertilizer applications and then low and high pollinator visitation treatments on those plots could be conducted using the methods I developed in Chapter 3.

Micro and macronutrients did appear to play a role in fruit set and yield, although the effects were inconsistent between years. While little is known about the effect of macro and micronutrients on lowbush blueberry yield, preliminary research examining the experimental supplementation of fields with foliar sprays of boron and calcium demonstrated that both micronutrients facilitate pollen germination, resulting in higher fruit set and berry weight (Chen et al., 1998). In light of this study, my results suggest that the effect of optimal boron and calcium on pollen germination might have been a key mechanism limiting fruit set in 2014. Yet growers only add NP fertilizers (either monoammonium or diammonoim phosphate) and do not manage other macronutrients or micronutrients (Bell et al., 2009a), so it is unclear what the source of this variation in tissue micronutrients could be.

Identifying the key agronomic determinants of pollination benefits to lowbush blueberry will require more focused factorial experiments in which different predictor variables are manipulated. In fact, a limitation of my study is the inability to discern interactive effects between pollinators and agronomic predictor variables owing to the large number of variables under consideration. Recent factorial experiments have manipulated one or two variables related to pest management (Chapter 3, Lundin et al. (2013), irrigation (Klein et al. (2015), Groeneveld et al. (2010), or nitrogen fertilizer (Klein et al. (2015) and Marini et al. (2015)) to precisely study their relationship to pollination. The strength of my study is that it provides insight into variation among multiple indicators of plant limitation across a spectrum of commercial production. In Chapter 3, for example, I was able to demonstrate the importance of pest management factors but only by contrasting recommended insecticide and fungicide levels to plots that were left completely untreated. Such work provides limited information as to the extent of plant and pollen limitations in typical agricultural settings. The closest parallel study is that of Boreux et al. (2013) in coffee production in India. The work outlined in this Chapter, combined with the experiment in Chapter 3, suggests the importance of both factorial and multivariate exploration of actual farm practices working in tandem in order to better resolve how pollination benefits accrue under present management conditions.

4.6 SUMMARY TO CHAPTER 4 AND TRANSITION TO CHAPTER 5

The findings from Chapter 4 build on those of my factorial experiment in Chapter 3, demonstrating that biophysical dimensions of pollinator contributions to agricultural output are contextualized within current farm practices, problematizing the assumption of strict pollen limitation in calculations of pollinator ESV outlined in Chapter 2. One conclusion that might be drawn from these findings is the need to modify current methods of calculating pollinator ESV so that they account for their agronomic context. Given the complexity and changing character of the agronomic context I found in commercial blueberry production in just my two year study, it is clear that any project to specify the agronomic context across agriculture would be an immense undertaking. But this focus on the compounding technical problem of valuation seems to sidestep a crucial question: why do we think that more and better valuations will translate into better conservation outcomes? This question should be familiar to the reader as I raised it first in Section 2.7. But to properly take this question up requires shifting from pollination ecology to the realm of social theory. Consequently, Chapter 5 introduces the second methodology of my dissertation, critical theory. Using critical theory I explore how focusing narrowly on the technical questions, such as the ones raised by my research in Chapter 3 and 4, tends to overlook the problem of how conservation might possibly advance given our present social context. Critical theory begins from the premise that we are unable to fully separate our thoughts about society from our social context, even though this might *appear* to be the case. Using this methodology I advance the argument that the key problem of ecosystem service value approaches are that they tend to consider themselves as standing outside of their social context, which renders the notion that technical innovation will bring about conservation plausible. Yet it is this plausibility, I suggest, that enables ecological degradation to advance, paradoxically, in parallel with increasing technical sophistication of valuation. I argue that this assumed distance from social context is also present in all major criticisms of ecosystem service approaches, which means that while these criticisms are insightful, they remain entirely descriptive of the shortcomings associated with these approaches. Consequently, contemporary criticism of ecosystem service valuation are likewise unable to shed insight into how the present opposition of ecological degradation and social necessity might be overcome.

My concluding chapter (Chapter 6) will attempt to draw my insights from pollination ecology and critical social theory together in order to better specify my original question, why conservation of wild pollinators appears to proceed so slowly, in spite of the large value they purport to deliver to agriculture and the broad public awareness around pollinator decline.

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CHAPTER 5 CRITIQUE AND TRANSFORMATION: ON THE HYPOTHETICAL NATURE OF ECOSYSTEM SERVICE VALUE AND ITS CRITICISMS

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5.1 ABSTRACT

Ecosystem services valuation (ESVn) attempts to transform the opposition of human economic necessity and ecological conservation by valuing the latter in terms of the services rendered to the former. However, despite a number of ESVn-inspired sustainability initiatives since the 1990s, global ecological degradation continues to accelerate. This suggests that ESVn has fallen far short of its goals of sustainable social transformation—a failure which has generated considerable criticism. This paper reviews three prominent lines of ESVn criticism: 1) the neo-Marxist criticism, which emphasizes the “fictitious” character of ecosystem commodities; 2) the liberal criticism through Friedrich Hayek’s concept “scientistic objectivism”; and 3) the pragmatist criticism of “value monism”. Although each form of criticism provides insight into the limitations of ESVn, all share ESVn’s inability to discern what kind of social transformation is *possible*. Unable to provide an account of their own immersion in social and historical context, these approaches operate in the *hypothetical*. In light of these shortcomings, this

paper advances a critical theory approach, which I contend provides conceptual tools uniquely well-suited to more adequately address the question of social transformation.

5.2 THE HYPOTHETICAL CHARACTER OF ECOSYSTEM SERVICE VALUATION (ESVN)

Increasingly the conservation of ecosystems is justified on the basis of the economic value of the human welfare these ecosystems support. Since many of the "services" supplied by ecosystems (e.g. carbon sequestration, water purification, habitat for insects pollinating nearby crops) are not currently captured in markets, advocates of ecosystem service valuation (ESVn hereafter) hope to revitalize conservation efforts by calculating and revealing the associated and hidden welfare benefits (e.g. Armsworth et al., 2007; Costanza, 1996; Daily, 1997; Liu et al., 2010; MEA, 2005). ESVn has come a long way. From humble speculative beginnings in the 1970s by figures such as Walter Westman, EF Schumacher and Paul Ehrlich (Braat, 2012; Gómez-Baggethun et al., 2010), to the ambitious estimation of global service provision in the late 1990s by Robert Costanza and colleagues (1997), ESVn is now the central force justifying contemporary conservation efforts, the most prominent of which is the newly launched Intergovernmental Platform on Biodiversity and Ecosystem Services (IPBES) (Cardinale et al., 2012).

At the same time, ESVn has been unable to address the intricate interrelationship between social structure and ecology. While the majority of ecosystems that contribute to human well-being are currently being degraded, much of this degradation has accelerated throughout the latter half of the twentieth century (MEA, 2005)—*precisely the period in which ESVn developed.* This includes the 1980-1990s, a period that Gómez-Baggethun and Ruiz-Pérez (2011) suggest gave rise to a myriad of market-based environmental protection initiatives; the immediate precursors of ESVn. Yet, many of the initiatives from this period (e.g. managing externalities of pollution through tradable allowances (Newell et al., 2013; Stavins and Schmalensee, 2012), wetland mitigation banking (Robertson, 2006) or promoting local economic development as a means to slow biodiversity loss (Ghazoul, 2007; Muradian et al., 2013)) have fallen far short of their anticipated goals.

ESV initiatives developed in the last decade have fared no better. The first international wave of these initiatives, the Millennium Ecosystem Assessment (MEA) (2005) and The Economics of Ecosystems and Biodiversity (TEEB) (2007), coincided with the failure to meet the Millennium global targets on biodiversity (2000-2010) (Butchart et al., 2010). Expectations are high that the accumulation of research, theoretical approaches and practical experience with ESVn will finally coalesce under the IPBES (Cardinale et al., 2012; Perrings et al., 2011; Perrings et al., 2010), enabling it to play a decisive role in meeting the Aichi Targets of the Strategic Plan for Biodiversity 2011-2020 (Convention on Biological Diversity, 2011). This assessment, however, presupposes that the prior limitations of ESVn were technical in nature and that future limitations can be resolved by simply *more* research, theory and operationalization (e.g.

Daily, 2000; Daily et al., 2009; Kinzig et al., 2011; Kremen, 2005; Kremen and Ostfeld, 2005).

Such a narrow focus on technical and operational shortcomings is indicative of what I will refer to as ESVn's *hypothetical* character. The focus on technical progress, I contend, is not incidental, but emerges from an assumed distance from the social-historical context that generates ecological deterioration. Ecosystem service value, then, operates in the hypothetical insofar as it presupposes a separation between itself and the social-historical context within which its research activity takes place. Indeed, ESVn advocates are able to think of themselves as making progress, despite their own prognosis that ecological systems are being degraded, because they assume an Archimedean standpoint outside of the ecologically destructive dimensions of society. It is by virtue of this decontextualization that ESVn advocates are able to perpetuate the assertion that more and better market valuations of "ecosystem services" will overcome past failures, in spite of any evidence of efficacy (Laurans et al., 2013). However, ESVn fails to provide an adequate account of ecological degradation (much less a feasible strategy towards sustainability) because its practitioners are unable to grasp how their research activity is *mediated* by a social-historical context deeply connected to patterns of ecological degradation. The ESVn approach is *hypothetical* because it lacks the means of discerning how the constitutional logic of modern capitalist society might inhibit: 1) efforts to illuminate this structure and 2) collective efforts to deal with pressing social problems, such as global climate change, in an effective manner (i.e. in a manner that does not regenerate the problem itself in a different form) (Dahms, 2008: 14-15).

Likewise, scholars examining the continual degradation of global ecological systems have not yet fully recognized the social and historical context through which such degradation takes place. Amid historically unprecedented levels of political-economic global interconnectivity following the end of the Second World War (see, e.g., McNeill, 2000), the acceleration of ecological degradation throughout the latter half of the twentieth century appears paradoxical: *In the post-WWII era, degradation is compounded in proportion to our awareness of these problems* (although see, Blühdorn, 2013; Stoner, 2014; Stoner and Melathopoulos, 2015). Following Stoner (2014), I refer to the paradox of increasing ecological degradation amid growing environmental attention and concern as the *environment-society problematic*. Unable to discern this paradoxical historical development, the normative aim of ESV (i.e., sustainability) is not borne out in practice and remains *hypothetical*. The rising tide of ESVn—far from indicating an increasing capacity to shape our future towards less ecologically-destructive ends—actually signals a growing inability to shape (let alone understand) the social-historical context that is generative of such runaway ecological degradation.

This is not to suggest that the limitations of ESVn have gone unnoticed. The growing recognition that ESVn coincides with the deterioration of key biophysical indicators, rather than their improvement, has led to what Fischer et al. (2007) refer to as a widening "sustainability gap". For example, ESVn has not been able to address how human welfare has increased in the face of the degradation of ecosystems (Raudsepp-Hearne et al., 2010). According to critics, ESVn obscures, rather than clarifies, the social basis for ecosystem conservation. Indeed, as shortcomings of ESVn have become increasingly visible in recent years, criticisms have mounted. Chief among these are 1) neo-Marxist; 2) liberal; and 3) pragmatist lines of ESVn criticism.[4] Although these three lines of criticism represent the most significant attempts to understand the limitations of ESVn to date, none are able to make historical sense out of the growing popularity of the ecosystem service approach itself. Consider a familiar example, the claim that €153 billion of agricultural production globally in 2005 rested on the activity of a single ecosystem service: the pollination of crops by insects (Gallai et al., 2009). As I will demonstrate, these criticisms are unable to account for key social, institutional and historic dimensions of this activity, such as: a) why pollination ecologists presently rely on ESVn to motivate pollinator conservation (i.e., in ways they did not in the past), b) why they would be so willing to justify conservation from such a thin layer of empirical data interpreted through a troublesome set of assumptions (Chapter 2) and c) how the value of pollination services were so readily taken up and reproduced within civil society (e.g., in mass media, among environmental lobby groups and activists, beekeeping associations, even in the brochures of agrochemical companies) without affecting a commensurate improvement in pollinator conservation (Chapter 6). As I endeavor to demonstrate, this is in large part because the theories underlying each criticism are also

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[4] For the sake of brevity and clarity, I will select authors whose works we believe offer exemplars of each approach, though I certainly recognize that these scholars may or may not self-identify with the labels (neo-Marxist, liberal, and/or pragmatist) I ascribe to them.

unable to grasp their own immersion in society and history. In this way, critics are only able to consider ESVn as either “wrong thinking” or determined by agents that somehow stand outside or above society (e.g., market environmentalists, ecological technocrats, hardened ideologues). Consequently, like ESVn, the criticisms are restricted to reacting to social transformations, passively describing these changes, but never being able to regard them reflexively – never attaining the level of a theory about how society could potentially change. In other words, both ESVn and its criticisms fail to recognize the *potential* for society *to change* because neither can grasp the deeper causes of social discontents (e.g., discontents to which a pollination ecologist who calculates ESVn or those who promote their estimates are ultimately responding) or the ways in which such discontents are integrated back into social structures, thereby allowing long-standing socio-ecological problems to be perpetuated.

This paper engages in an immanent critique of the neo-Marxist, liberal, and pragmatist attempts to understand the limitations of ESVn in order to illuminate the historical specificity of our current inability to locate a social basis for ecosystem conservation. This immanent critique reveals how, in opposing ESVn, these criticisms reproduce its most problematic feature: environmental degradation is decontextualized and, as a result, sustainability remains hypothetical. Against this background, I outline two key methodological motifs of a critical theory approach, which I contend provides conceptual tools that are uniquely well-suited to more fully comprehend the links between economic progress and ecological deterioration and the discontents this generates.

5.2 THE CRITICISM OF ECOSYSTEM SERVICE VALUATION (ESVN)

5.2.1 The Neo-Marxist Criticism: Ecosystem Services as Commodities

The neo-Marxist line of criticism (exemplified by Kallis et al., 2013; Kosoy and Corbera, 2010; Peterson et al., 2010; Robertson, 2012; Robertson, 2000) argues that ESVn fails because the process of abstracting commodities (i.e., services) from ecosystem functions obscures the complicated interconnections within ecosystems and between society and ecosystems, leaving us to mistakenly "think that capital grapples directly with material nature" (Robertson, 2012: 396). Moreover, the abstraction of value from ecosystems undermines the potential awareness of these interconnections because consumers, land managers, ecologists and others become fixated on ecosystem values in the market, a process Kosoy and Corbera (2010) liken to the worship of a fetish object in pre-modern societies. Kosoy and Corbera's reference to this fetish-like reverence of ecosystem value, like all neo-Marxist criticisms of ESV, draw on Marx's analysis of commodities in *Capital Vol. 1*, specifically the final section of Chapter 1 titled "The Fetish of the Commodity and Its Secret" (Marx, (1976 [1867]): 163-177).

While this line of criticism purports to develop from Marx's analysis it asserts the necessity of adopting neo-Marxist theory (post-1960s) in order to broaden its scope beyond Marx's seeming narrow focus on the centrality of wage-labor to commodity-formation (e.g. Appadurai, 1986, for Kosoy & Corbera 2010). Significantly, this also involves taking up the key concerns of the sub-discipline of neo-Marxist geography, specifically the question of how, under capitalism, "technologies of measurement and abstraction are used specifically to define adequate bearers of value" (Robertson, 2012: 388). The criticism, in turn, rests on a careful analysis of the social and ecological implications associated with each of the stages in transforming ecosystems into a commodity that can "bear value"—that is, the itemization, characterization and spatial mapping of ecosystem functions, their re-categorization as ecosystem services and their packaging or bundling into tradable commodities. To generate value for pollination services, for example, the activity of wild pollinating insects must first be: 1) separated from their broader ecological context and recontextualized in terms of their spatial location relative to agricultural crops (Lautenbach et al., 2012; Schulp et al., 2014), 2) their economic benefits to agricultural production calculated from their new context (Chaplin-Kramer et al., 2011), and then, 3) rebundled with other ecosystem services associated with crop yields (Bommarco et al., 2013).

According to the neo-Marxist criticism, the significance of this technical process in creating "a more and more differentiated realm for the circulation of capital" (Kosoy and Corbera, 2010: 1231) is in stark contrast to ESVn advocates, who view the process of technical innovation as key to resolving ESVn shortcomings. Indeed, the neo-Marxist criticism provides a compelling account for why technical solutions *appear* efficacious to ESVn practitioners. By structuring the way society understands its relationship to ecosystems – in terms of "services" that "bear economic value" – the definition and specification of those services only appears to be a path leading to sustainability; while in actuality, this path leads to the destruction of ecosystems:

> *Bracketing nature as 'material' or as an unmediated force in capitalist accumulation prevents us from discerning the struggle over the creation of value bearing abstractions from its materiality (...) In dealing with nature, we are always attended by the invitation to mistake the ordering of appearances for order itself, to mistake the difficulties of classifying and categorising nature for the intransigence of nature itself. (Robertson, 2012: 397)*

On this basis, the neo-Marxist criticism purports to explain a number of problematic aspects of contemporary ESVn practices, including: a) how the overwhelming demand for some services (e.g. carbon sequestration) leads to the degradation of other associated ecosystem functions (e.g. reforestation with fast growing trees reduces biodiversity and soil retention) (Kosoy and Corbera, 2010; Peterson et al., 2010), b) why diverse social and cultural values associated with ecosystems have become less relevant to the assessment of ecosystem value, paving the way for highly asymmetrical power structures for rendering management decisions (Kallis et al., 2013; Kosoy and Corbera, 2010). Moreover, by grounding the emergence of ESV in the rise (1980s-present) of "market environmentalism" (Kosoy and Corbera, 2010) or "neoliberalism" (Kallis et al., 2013; Peterson et al., 2010; Robertson, 2012; Robertson, 2006) the criticism c) provides an explanation of why ESVn has been accompanied by an upward redistribution of wealth.

The neo-Marxist criticism views the fetish character of putting monetary values on ecosystems as synonymous with the idea that valuation constitutes "commodity fiction". The idea of a "commodity fiction", however, does not arise from Marx, but from the thought of Karl Polanyi (2001 [1944]), whose historical sociology sought to ground the underlying dynamic of modern society in the commodification of three things that supposedly cannot be commodities, namely land, labor and money. According to the neo-Marxist criticism, the first of Polanyi's fictitious commodities (land) can be broadly reinterpreted to encompass ecosystems. Polanyi maintained that any attempt to render these dimension of life as commodities results in a resisting social counter-movement that sets up an oscillating historical dynamic between advancing markets and restricting these markets through public control. Consequently, rather than any form of social transformation, these authors look to a broadly-constituted *resistance* against commodification.

The recourse to resistance, however, misses a key dimension through which social discontents of the 1970s not only failed to reduce ecosystem degradation, but also reproduced capitalism in a new form. While neo-Marxist authors acknowledge that market-oriented utilitarian approaches to environmental and resource management gained popularity just as global neoliberal capitalism was emerging in the 1970s, they attribute this development to forces outside of society. According to the neo-Marxist criticism, the synchronicity of neoliberalism and the growing popularity of utilitarian approaches to the environment during the 1970s involved expanding the scope of markets to incorporate dimensions of life considered public goods or common pool resources. This, in turn, rolled back the scale and scope of national environmental legislation and regulatory agencies, which had been tasked with protecting these resources beginning in early decades of the twentieth century (Peterson et al., 2010). Neo-Marxists typically interpret the declining role of the state in regulating society's relationship to ecosystems as paving the way for a new round of capital accumulation following the global political and economic crises of the 1970s (Kallis et al., 2013; Kosoy and Corbera, 2010; Peterson et al., 2010; Robertson, 2012; Robertson, 2000).

The advance of "market environmentalism" and the subsequent appeal of ESVn, however, cannot be fully explained with reference to the rolling back of the state-centered conservation framework "from above" (Heynen et al., 2007; McCarthy and Prudham, 2004). Unlike the 1960s, the 1980s witnessed growing public fatigue with environmental issues, which was combined with a sense of the insufficiency of environmental regulation (the "implementation deficit", Røpke, 2005: 268). That is to say, popular discontent was an important part of the socio-historical context that gave rise to "market environmentalism"—the seeming opposite of Polanyi's prediction that social counter-movements push in the direction of restricting markets and placing them under public control. Consequently, advocates of so-called "market environmentalism" may not have driven the rolling back of state-regulation of the ecosystems as much as some neo-Marxists imply (Kallis et al., 2013; Kosoy and Corbera, 2010; Robertson, 2000). A more likely scenario is that market environmentalists opportunistically adapted to pressures arising from society itself, such as fatigue with environmental issues. Rather than attempt to grasp the basis of this regulatory roll-back in terms of social mediation (an approach I develop below in our discussion of critical theory—Section 5.4), the neo-Marxist criticism relies on the unwarranted assumption that popular discontents (e.g. contemporary environmentalism) stand outside the social structure of capitalism. Such a perspective fails to adequately account for how capitalism itself is transformed (e.g., from its regulatory state form to one in which regulation is rolled back) in and through popular discontents (e.g., 1960s discontents with industrial pollution results in the expansion of state regulation in the 1970s and the disappointment with these regulations, coupled with unease from a severe economic downturn, contributes to their undoing in the 1980s).

What is entirely overlooked by these critics is that Marx's "economic" categories—value, commodity, capital, labour, surplus value—are not solely economic, but rather *forms of social being* specific to the capitalist mode of production (Postone, 1993). As Postone (2007: 16) observes, "Polanyi's insistence on the fictitious character of labor, land, and money as commodities obscures Marx's analysis of the commodity as a form of social relations". In other words, in contrast to Polanyi, commodity for Marx is not a matter of an underlying social, cultural or ecological foundation made obscure to itself by the "fiction" of exchange. Instead, the peculiar form of social objectivity and subjectivity characteristic of capitalist societies is mediated through the commodity form and must be understood as such. Furthermore, for Marx, the contradictory character of commodities (i.e., how they appear concrete and natural as well as abstract and "fictional") cannot be separated from the historically unprecedented *social* dynamic (i.e. capital) in which his theory takes root. From this perspective, history is not driven by forces outside society – there is no "natural" ground for discerning a "real" from a "fictitious" commodity, only the changing form of modern society itself.

Although Robertson (2012: 396) comes close to recognizing the distinctly social character of commodities when he states that the necessity for resistance "does not arise from the point at which intransigent nature expresses its material character, but rather at the point at which we (…) retract our consent to the adequacy of social abstractions as bearers of value", this insight is confined to his Polanyian approach. For Robertson grounds agency in the reaction against the commodity form rather than attempting to recognize the *possibility* for further transformation lying within it. For Marx, on the other hand, the conscious transformation of society—e.g., towards less ecologically destructive ends—requires critically recognizing social context as that which renders possible new (and more adequate) forms of thought and activity.

5.2.2 The Liberal Criticism: Ecosystem Services as "Scientistic Objectivism"

Unlike the neo-Marxist line of criticism, the liberal criticism does not contend that markets obscure social and ecological reality. According to the liberal criticism, exemplified by the prominent environmental philosopher Mark Sagoff, the "fiction" of ESV exists precisely to the extent that ecosystem service values are taken to be as an objective measure of social welfare when in fact they represent the narrow interests of environmentalists. From this perspective, the 1980s are not, as the neo-Marxists would have it, a period marked by the wholesale commodification of ecosystems by "market environmentalists". Rather, the 1980s signals the weakening strength of environmentalist politics (1970s) in light of economic stagnation and the inability of ecological economists to reverse this trend by asserting that economic activity objectively resides within ecosystems (Sagoff, 2012). Although taken from this perspective neo-Marxist and liberal criticisms of ESV appear to have little in common, I contend that they express two related features of the 1980s (albeit in a fragmented and one-sided manner). Both neo-Marxists and liberals regard ESVn as being imposed on society from “above” (i.e., by “market environmentalists” for neo-Marxists and by ecological economists for liberals). They differ only in how they conceptualize society; neo-Marxists (following Polanyi) understand society in collectivist terms (i.e., society is constituted by shared social values, particularly among poor and marginalized strata of society) and liberals in terms of the individual (i.e., self-interested individuals generate a social whole that is more than the sum of its parts).

In this section I demonstrate that the collectivist/regulatory state that emerged in the 1930s was not eliminated in the 1980s, as liberals had hoped, but *transformed* such that collectivist and individualist features of societies became deeply integrated. I develop this proposition through the concrete example of a well-studied ecosystem service, the insect pollination of agricultural crops, and show how valuations enabled conservation to be integrated into state-regulation of agricultural markets through agri-environmental initiatives. Integration, however, progressed in-step with the “liberalization” of agriculture in the 1990s, which demonstrates how the liberal criticism cannot be readily separated from its social context. I conclude that the liberal criticism of ESVn is hypothetical because, like ESVn, it merely reflects (and perpetuates) the social context that is generative not only of ESVn and ecological degradation, but the growing illiberal character of society more generally.

The liberal criticism is indebted to Friedrich Hayek (1899-1992). Hayek's key works were written at the height of WWII when a weakened liberalism appeared eclipsed by state-centered planning and intervention in the economy (i.e., in the wake of the 1930s with fascism and communism in Europe and Japan and the New Deal reforms in the U.S.) (Amadae, 2003: 15-23). As Castree (2007) points out, Hayek's most famous work, *The Road to Serfdom* (2006 [1944]), was published the same year as Polanyi's *The Great Transformation*, but drew opposite conclusions on the character of the 1930s. In contrast to Polanyi, Hayek's thought (along with Joseph Schumpeter and Karl Popper), anticipated what Amadae (2003) broadly characterizes as *rational choice liberalism* which, "differs from classical liberalism from its single-minded dedication to the principle of rational self-interest, and in proposing that questions of constitutional design can be settled by recourse to precisely formulated mathematical models reflecting individuals' self-interested calculations" (154). Rational choice liberalism had a profound effect in the U.S. during the Cold War through thinkers as diverse as Kenneth J. Arrow, James M. Buchanan and William H. Riker. It articulated "a philosophy of markets and democracy that was developed in part to anchor the foundations of American society during the Cold War" such that "from the closing days of WWII to the mid-1980s, rational choice theory rebuilt the conceptual cornerstones of Western ideals" (Amadae, 2003: 2-3). Although the implications are beyond the scope of this paper, it is important to remark that rational choice liberalism broke significantly from the traditions of both classical liberalism and neoclassical marginal utility (Amadae, 2003: 193-250) and was more squarely situated within the same Cold War context that was generative of contemporary environmentalism and ecological economics.

The liberal criticism of ESV draw on Hayek's formulation of "scientistic objectivism", which he developed in a series of essays (Hayek, 1942, 1943, 1944) immediately prior to the publication of *The Road to Serfdom* (Hayek, 2006 [1944]), a work that attacked the authoritarianism of the regulatory state of the 1930s. Hayek develops the term "scientistic" (and "scientism") to describe a form of rational planning in the 1930s that was informed by "objective" laws derived by "impartial" specialists who assume a perspective above society. Such planning purported to act in the name of an objectivity previously accorded only to the physical sciences. But the parallel between Science and "scientism", Hayek asserts, is illusory. "Scientism", for him, is but the "slavish imitation of the method and language of science" (Hayek, 1942: 269) and its "objectivity" contributes "scarcely anything to our understanding of social phenomena" (Hayek, 1942: 268).

Take, for example, the case of pollinator ESVn. Liberal critics claim that while pollination ecologists employ Science to classify patterns of wild bee pollinator biodiversity, they cross over into "scientism" when they extend their "objectivity" into claims of the social welfare benefits for conserving wild bee habitat. It is one thing, the critics suggest, to maintain that bee biodiversity is linked to patterns of uncultivated land-use around agricultural crops, or that the yield of these crops increases in relation to the dynamic and interacting character of this biodiversity (Luck et al., 2009: 228), but it is another to assert that preserving bee habitat is the only avenue available for farmers to increase their profits (Sagoff, 2011). In other words, pollination ecologists engage "scientism" when they extend their objective ecological findings to questions of social necessity.

What makes social and physical phenomena so different? Hayek claims that social categories have no underlying (i.e., ontological) ground. In contrast to the neo-Marxist Polanyian framework discussed above, liberals eschew the notion that there is a predetermined basis for discerning a "fictitious" from an "authentic" commodity. Rather categories of society are constituted by the thoughts of people within society (Hayek, 1942: 285). Hayek notes "neither a "commodity" or an "economic good ", nor "food" or "money" can be defined in physical terms but only in terms of views people hold about things" (Hayek, 1942: 281). But, according to Hayek, what individual people *think* about the character of society as a "whole" is a vastly different proposition from how the "whole" is *constituted* by the thoughts of all the people in society. "Scientism" confuses the former (i.e., our opinions of social processes) with the latter (i.e., how society functions). According to the liberal critics, what the pollination ecologist assume, but never develop, is an individualist or "compositive theory of social phenomena" that can "grasp how the independent action of many men can produce coherent wholes, persistent structures of relationships which serve important human purposes without having been designed for that end" (Hayek, 1943: 27).

"Scientistic objectivism" enables liberal critics to account for a key shortcoming of ESV approaches—the "environmentalist's paradox" whereby human wellbeing appears unconnected to the degradation of ecosystems (Raudsepp-Hearne et al., 2010).

Extending the pollination example, the paradox can be expressed as follows: In the face of large and growing estimates of ESVn by pollination ecologists (Calderone, 2012; Gallai et al., 2009; Lautenbach et al., 2012), the explosive growth of pollinator-dependent crop output since the 1990s (Aizen et al., 2009) and the relatively slow growth of insect pollinator supply (Aizen and Harder, 2009; Breeze et al., 2011; Breeze et al., 2014b; Schulp et al., 2014), why do farmers growing pollinator-dependent crops fail to make investments in conserving pollinator habitat (Ghazoul, 2007; Hanes et al., 2013; Melathopoulos et al., 2015)? The liberal critic dissolves this paradox by asserting that production increases with declining pollinator service flow because growers manage pollen limitation dynamically. Growers do more than adjust inputs and adopt new technologies (see Chapter 3 and 4, but also (Boreux et al., 2013; Lundin et al., 2013), they also seek to control pollen limitation in relation to society *as a whole* by managing the constant interaction of economic variables: “the elements, components, or units of ecosystems relevant to valuation are determined by and through the economic activity that surrounds them” (Sagoff, 2011: 501). According to the liberal criticism, the “environmentalist’s paradox” only *appears* to ecologists because in bracketing the activity of many independent actors working within a socially dynamic system, pollinator conservation is equated with static “ecological constants”, derived from seemingly “objective” biophysical laws within which growers are expected to conform (e.g., patterns of bee biodiversity across landscapes and their effect on crop yield (Luck et al., 2009)). Growers do not spend money on pollinator habitat restoration, according to liberal critics, because they are not sound investments.

Following rational choice liberalism and Hayek in particular, growers who deal with pollination services are regarded as self-interested, competitive firms that are fluidly engaged in local and informal negotiations to resolve unpaid externalities: "ecosystem services as a general rule already receive more or less appropriate quantification and pricing either explicitly in market exchange or implicitly in the Coasian bargaining that arises in the penumbra of markets and in the shadow of common law" (Sagoff, 2011: 500). Yet, if I take the history of pollination services into account, the liberal criticism does not withstand scrutiny.

The first valuations of pollination services did not come from ecologists, but from rational and independent beekeepers in full sight of "market exchange" (Chapter 2). In fact, all recent efforts by ecologists to calculate pollinator value employ a general methodology that originates from a U.S. National Honey Board-commissioned study (Robinson et al., 1989). The explicit focus of this study was to lobby the federal government to save the Honey Price Support Program on the grounds that beekeepers generated $9.3 billion in largely hidden social welfare benefits (e.g., positive externalities in the form of unpaid pollination of agricultural crops)(Muth and Thurman, 1995; Robinson et al., 1989). The Program had been in place since the 1950s, but like price support programs for all other agriculture sectors, it came under scrutiny after a steep decline in commodity prices and crippling double-digit interest rates in the early 1980s. By the end of the 1980s, over 70% of large farms in the U.S. were at risk of foreclosure and expenditures on commodity support programs soared to $26 billion (Potter, 1998), with $81 million (1985) being paid out specifically to beekeepers (Muth and Thurman, 1995).

Unlike other sectors of the economy, gains in agricultural productivity are rapidly lost to low prices (Timmer, 2009). The chronic tendency of agriculture towards oversupply has plagued the farm sector through most of the twentieth century and only episodically resulted in short periods of high profits (e.g., 1973-1980 and 2006-2011) (Timmer, 2010), resulting in uneven income distribution between agricultural and non-agricultural sectors (Timmer, 2009: 30-34) and social pressures that emerge from the growing disparity between rural and urban incomes. The pattern for resolving these pressures—managing agricultural prices through various forms of subsidization (e.g., import tariffs, supply management programs, payments for removing land from production, and direct payments)—emerged in the 1930s following the first major collapse of the newly constituted global market for grain in 1921 (Friedmann, 1982, 1993). Although Hayek would correctly identify the rise of state-planning in the 1930s with the erosion of cosmopolitan liberal democracy and retrenchment of the nation state, the example of agriculture shows this not to be the straightforward product of technocratic planners (i.e., as Hayek supposed) but also from broadly constituted discontents (e.g., farmers) that lay squarely in "the penumbra of markets and in the shadow of common law" (Sagoff, 2011: 500).

Farm income support programs were themselves drawn into a deep crisis in the mid-1970s along with many other features of the post-WWII political and economic order. As governments in industrialized countries attempted to restructure their economies by reducing government expenditures and promoting global trade liberalization, farm income programs became increasingly difficult to justify to a primarily urbanized tax base already facing deep cuts to their own state welfare benefits (Potter, 1998; 2009).

Yet, attempts to liberalize agriculture in the 1990s failed and the agricultural sectors in the U.S. and E.U. remain highly regulated and subsidized (Timmer, 2009). As Friedmann (1993: 29) correctly anticipated, “the choice is not between ‘regulation’ or ‘free trade’ (…) but between new forms of implicit or explicit regulation”. Consequently, while liberal criticisms of ESVn presuppose liberalization—that agricultural firms directly and immediately interact with one another (and with surrounding ecosystems)—they ultimately miss how the relationship between agricultural firms and ecosystems are *actually* mediated through social structures (e.g., through state-regulated price control programs) designed to cope with quasi-objective laws of modern society (e.g., the chronic tendency of agriculture towards oversupply). In doing so, liberals not only overlook how farm sector lobbyists joined ecologists in advancing their interests using valuations, but also the way in which their interests became increasingly integrated through state-led agri-environmental schemes starting in the 1990s (Potter, 1998). Today, a major source of agricultural price support comes in the form of state-legislated biofuel blending standards in the U.S. (beginning in 2007) and by E.U. members states (2003) that are justified politically in terms of increasing carbon sequestration ecosystem services (Gerasimchuk and Koh, 2013; Tyner, 2008). Significantly, these changes have implications for wild bee biodiversity and pollinator

ESVn, since a key feedstock for bio-diesel production in these counties is oilseed rape (Breeze et al., 2014b), a moderately pollinator-dependent crop that many wild pollinators utilize (Holzschuh et al., 2011; Holzschuh et al., 2012).

Since the sectional interests of farm groups appear better able to control the terms of agri-environmental policies, they are typically accompanied by weak provisions for evaluating conservation outcomes (Kleijn et al., 2011). Moreover, in the face of deep cuts in state expenditures after the 2008 global economic crisis, ecologists and conservationists correctly fear that even the meager conservation provisions within existing agri-environmental programs will be disproportionately weakened relative to farm income support (Pe'er et al., 2014; Potter, 2009). In this sense ESVn approaches may not so much be “scientistic” (i.e., advancing an abstract image of how society and ecosystems ought to interact *from above*) as imitating what they correctly deem as the successful rent-seeking strategies coming *from below* (e.g., those used by farm lobbyists competitors) in light of an illiberal social context.

Liberal critics are certainly correct in associating the growth of ESVn with the diminishing role of politics in socially manifesting and working through disagreements over ecosystem management. They point out that ESVn is the antithesis of the sharp and clearly-posed political differences that shaped the framework of U.S. federal environmental laws and enforcement in the early twentieth century (Sagoff, 2002). What liberal critics are wholly unable to explain is *why* political arguments for ecosystems invariably assume an apolitical and illiberal character today (i.e., why arguments for ecosystem preservation proliferate everywhere in society except in clearly-articulated legislation (see Kennedy, 1998)) (Sagoff, 2012). There is a clear tautology in the liberal criticism that claims ESV undermines political discourse over ecosystems. If ESVn is simply “scientism” (i.e., it is not constitutive of society) where, then, does it find the power to undermine politics, unless, in fact, ESVn is deeply rooted in society (i.e., it is not “scientism”)?

The liberal criticism operates in the hypothetical because it is unable to locate the illiberal features of society *in society*, but instead attributes them to a largely exogenous force (i.e., ESV) and asserts society is otherwise liberal (i.e., governed by common law and constituted by markets). But as I have argued, the growth of ESVn reflects a growing integration between conservation and new forms of “implicit” regulation that have advanced alongside “liberalization” (i.e., the dismantling of “explicitly” regulated markets) since the 1990s. Although this enables liberal critics to describe many of the problems associated with ESV it is unable to grasp ESVn as symptomatic of a deeper social dynamic. This dynamic—typified by the chronic tendency of agriculture towards oversupply and the perpetual return of regulation—assumes a quasi-objective form, to which farmers, conservationists and liberal critics are all subject. Unable to recognize itself as reflecting this dynamic, the liberal criticism of ESVn becomes a way of perpetuating this dynamic along with all its contradictions.

5.2.3 The Pragmatist Criticism: Ecosystem Services as “Value Monism”

The pragmatist line of criticism is exemplified by the work of the environmental philosopher Bryan Norton, who draws together American pragmatist philosophy with the thought of the early twentieth century conservationist Aldo Leopold. Whereas neo-Marxists reject the objectivity of ESVn on ontological grounds ("fictitious commodities"), and liberals reject ESVn as the attempt to undermine the legitimacy of liberal society, pragmatists reject ESVn to the extent that it precludes "a more profound reexamination of how one might create a rational process of policy evaluation that truly takes into account both economic *and ecological* impacts of our decisions" (Norton and Noonan, 2007: 665).

Pragmatists attempt a deep reconsideration of society's relation to ecosystems through a process of rational deliberation (Norton, 2005: 51-56). In this sense, the pragmatist and liberal criticisms bear some resemblance to one another. The difference between the two approaches revolves around their respective assessments of the bitter debates over ecosystem management that characterized the twentieth century. While liberals look to the "moral fervor" of these debates as a progressive social force (Sagoff, 2012), pragmatists view the opposing poles of these debates as being hardened and unresponsive to changes in ecosystems, communities and values. For this reason, pragmatist critics consider contemporary environmentalism as a force for polarizing society, diminishing the "room for flexibility and for learning from experience" (Norton, 2005: 56). The pragmatist criticism of ESVn is distinct in its anticipation of the possibility for a more constructive and open-ended approach to relating ecosystems to society.

Pragmatists criticize ESVn for assuming that social welfare can be captured through a single dimension—namely, the quantitative and objective measure of individual consumer preferences (Norton and Noonan, 2007). Yet, in practice, the attempt to render the functional complexity of ecosystems, as well as the public and future interests people ascribe to them, into monetary units demands considerable theoretical abstraction. Paradoxically, such abstraction means relaxing the very neoclassical economic principles of utility ESVn looks to employ. Consequently, ecologists who attempt to motivate social demands for conservation by emphasizing the economic benefits of protecting ecosystems face what Norton and Noonan (2007: 668-669) term "the ecologists' dilemma"**;** they either must restrict the scope of benefits generated by ecosystems in order to fit within the marginal utility framework, or stretch the definition of utility to a point where it is no longer is recognizable as economic value.

While the "ecologists' dilemma" poses a seemingly insoluble trade-off to ecological economists, Norton and Noonan (2007) suggest this is only the case to the extent that they insist that welfare benefits be expressed in terms of a single "objective" dimension of value. Moreover, they assert that breaking with this reductionist approach, which they term "value monism", is a matter of choice:

> *(...) if we recognize that the decision to model ecological values in the economic framework is a choice among multiple possible metaphors and models, then the decision as to what is important to measure rests on a value judgment (Norton and Noonan, 2007: 670).*

The choice against valuation is a process of communicative action known as *adaptive managemen*t. Drawing on Pickett and Candenasso (2002), who emphasize the role of metaphors in generating scientific models of ecosystems and in mediating societal values towards ecological systems, Norton and Noonan envision adaptive management as foregrounding these value-laden metaphors, using them consciously in an iterative, experimental and deliberative process. Here ecosystem service value is re-conceptualized not as an impartial form of objectivity but as one metaphor among many—Norton and Noonan characterize ESVn as a metaphor for "ecosystems as a welfare producing machine" (2007: 655)—whose adequacy can be judged not on its own basis (monism) but on its capacity, in relation to other metaphors, to match a constellation of social values with the dynamic and complex processes of ecological systems (pluralism).

The pragmatist criticism is potentially significant for locating the constraints to conscious social transformation in a form of instrumental rationality that gives rise to value monism, and the possibility that intersubjective deliberation might dissolve the hardened ideologies that perpetuate monistic approaches. However, pragmatists cut short these insights by not reflecting on the grounds of their own thought. The unwavering commitment to communicative action, for example, assumes subjective values (e.g., the different ways people value wild pollinators) exist independently of objective constraints (e.g., the chronic tendency of agriculture towards oversupply—Section 5.2.2). As such, the pragmatist criticism is ill-equipped to meet the concerns raised by neo-Marxists, whereby the experience of an ecosystem—and by extension one's value of it—is mediated through being considered as a commodity (i.e., "a welfare producing machine"). By extension, the pragmatist emphasis on the autonomy of social values fails to address a key characteristic of the present ecological crisis; if social values offer a path to sustainability how can this be reconciled with the fact of that proliferation of environmental awareness (i.e., environmental values) failings to bring about sustainable practices (Blühdorn, 2013; Stoner, 2014)?

These shortcomings highlight the limitations of defining the environment-society problematic (Section 1) in communicative terms alone. Pragmatism's emphasis on deliberative action falls short insofar as it recasts the classical epistemological subject-object relation in intersubjective and linguistic terms (Nelson, 2011). Significantly, the pragmatist criticism has paid less attention to the elusive processes of *social mediation*, which I return to elaborate below. The separation of subjective values from objective social structure therefore risks sinking into a "self-imposed abstractness" whenever pragmatists attempt to relate their theory to actual practices (Norton, 2005: 389). Not surprisingly, the promise of social transformation, which attended the communicative shift (see Habermas, 1983 [1981]) during the late 1970s and early 1980s, has failed. Indeed, there is an elective affinity between this failure, on the one hand, and the demand for politics, which has yet to be met. In fact, the increasing demand for social transformation (i.e., the desire for an ecologically "sustainable" society), expressed through the pragmatist proposals for adaptive management—far from facilitating effective communication—may even function to "disperse political responsibility and obscure chains of accountability" (Blühdorn, 2013: 31). Indeed, this pattern is illustrated by the failed initiatives to shape British health policy in the late 1990s through a process of public deliberation (Parkinson, 2004). Although these deliberative initiatives involved large numbers of people across multiple levels of health policy (regional to national), they ultimately functioned to legitimate transformation already underway by severely restricting the scope of discussion. In this way, adaptive management may conceal underlying powerlessness through a process that appears as its opposite (i.e., conscious and rational deliberation over ecosystem management). This vulnerability arises from the fact that most adaptive management initiatives claim successful outcomes without adequate implementation studies, leading to unsubstantiated claims that the approach can be readily scaled up to highly complex and large-scale socio-ecological problems (e.g. climate change) (Rist et al., 2013). Because of uncertainty over the capacity to deliver management outcomes, adaptive management assumes a similar hypothetical approach to transformation that characterizes ESVn. The interplay between concern for ecological degradation and the uncertain operational parameter of adaptive managements results in what Sagoff (2008: 86) characterizes as an "academic blessing" for entrenched positions that "institutionalize paralysis by analysis and (…) guarantee indecision over the long run". The growing concern around ecosystem degradation that characterizes the environmental-society problematic finds its full expression in adaptive management. That is to say, adaptive management meets the demand for transformation without, however, having to deliver outcomes. Ironically, this feature of adaptive management does not differentiate it from ESVn, but parallels it (Laurans et al., 2013).

5.3 METHODOLOGY OF CRITIQUE

In this paper I have shown how, in criticizing the shortcomings of ESVn, neo-Marxist, liberal and pragmatist approaches take a standpoint outside their socio-historical context. As a result, these approaches are unable to address the central question underlying the issue of sustainability—namely, what kind of context might generate the type of transformation that could overcome the opposition of social necessity and ecosystem integrity? Both ESVn and its critics proceed without asking what kind of transformation is historically possible; and in this sense, they operate in the *hypothetical*. Notwithstanding their differences, the neo-Marxist, pragmatist, and liberal critics of ecosystem valuation remain a *reflection of* socio-historical context rather than a critical *reflection on* socio-historical context (Dahms, 2008). Unable to grasp their immersion within the socio-historical context that generates ESVn, these critics perpetuate "the sustainability gap" (Fischer et al., 2007) by recourse to nostalgia for past and failed political mediations (liberal), schemes for idealized deliberative spaces (pragmatists), and/or the frustrated call for "resistance" (neo-Marxist).

In light of these shortcomings, this section outlines key aspects of a critical theory approach, which I contend provides conceptual tools uniquely well-suited to more adequately address the question of social transformation. Because an adequate elaboration of critical theory is well beyond the scope of the thesis, the following focuses on two methodological motifs—*immanent critique* and *mediation*.

5.3.1 Immanent critique

As Kuhn's (1996[1962]) famous scientific revolution thesis makes clear, understanding scientific change and development is impossible without an acute comprehension of the significance of historical events and situations in shaping the activity of science. This basic insight is particularly relevant to critical theory, which must be situated in relation to its context—that is, modern capitalist society. Accordingly, one of the central aims of critical theory is to recognize and make explicit how, in an integrated capitalist society, it is impossible to think outside of our societal context. Even thoughts and phenomena which appear transhistorical, such as land, human labor, political legitimacy and human communication, always take place within a given context. Critical theory seeks to go beyond mere historicity (i.e., the past as an accumulation of facts and events) in order take into account how thought itself is historically constituted. Such a critical and reflexive approach can be traced back to G.W.F. Hegel (1770-1831), whose philosophical system was premised on the recognition that ideas about reality are historically situated. Later critical theorist such as Herbert Marcuse, Max Horkheimer, Theodor W. Adorno, and others associated with the Frankfurt Institute for Social Research in the 1930s, 40s, and 50s questioned the notion that researchers can separate themselves from the phenomena and societies they study. The Frankfurt School critical theorists resuscitated the critical impetus of Hegelian-Marxist social theory—an approach they saw as being commensurate with their commitment to social emancipation, particularly in the wake of the German Weimer Republic (1919-1933). In doing so, the Frankfurt School critical theorists sought to specify how ideas such as progress were legitimized through unquestioned authority in a repressive and administered world.

Max Horkheimer laid the groundwork for a methodological approach capable of taking into account the idea that knowledge claims are constrained by the reality perceived in his distinction between traditional theory, on the one hand, and critical theory, on the other (Horkheimer, 1972 [1937]). According to Horkheimer, the social-historical "object" of analysis—namely, modern capitalist society—is in no way separate from the social-historical reality of the researcher's milieu, which in turn defines the reality perceived (Horkheimer, 1972 [1937]). Traditional theory, by contrast, ignores the dualisms between thought and being, on the one hand, and understanding and perception, on the other. Severed from its social-historical context, traditional theory views scientific activity taking place alongside all other activities in society, "*but in no immediately clear connection with them*" (Horkheimer, 1975 [1937]: 197 [emphases added]). Theory must, then, be "critical" enough to account for its immersion in history. Theory, rather than being considered exogenous, must be recognized as an integral part of capitalist society. Critical theory is thus confronted with the twofold task of critique and transformation. Such an approach must provide a critique of its own social-historical context—and it must do so in a radically immanent manner, so as to specify the nature of historical development which must be confronted and overcome (Leiss, 2011) in any attempt to effect sustainable transformation. I contend that a critical theory geared toward sustainability must, at the very least, provide a critical and reflexive account of:

- The conditions of its own possibility (i.e. social context); and
- The immanent possibility of the fundamental transformation of this context (i.e., the notion that social context itself is generative of its own supersession).

Critical theory meets this twofold challenge of critique and transformation by engaging a methodology known as *immanent critique*. To begin, the critical theorist must explain how it is possible to critique capitalism while being a part of capitalism. That the structures and underlying social relations of modern society are contradictory is what generates the possibility of a critical stance toward this context (Postone 1993: 88). Take, for example, the chronic tendency of agriculture towards oversupply discussed in Section 5.2.2. The independent activity of freely contracting agricultural firms rapidly increases agricultural output, which in turn, undermines prices leading (in the twentieth century) to decades of low profitability. The discontents associated with this social dynamic are manifold and are not simply associated with the income gap that opens between agricultural and non-agricultural sectors of society. Attempts to regulate this dynamic in industrial countries (e.g., by subsidizing farmers) frequently resulted in increased productivity (i.e., through machinery and agro-chemicals) (Friedmann, 1993; Potter, 1998) that generated new discontents (e.g., environmentalist and consumer responses to the "industrialization" of agriculture, increased costs of farm programs and the backlash by the urban tax base, etc.).

As explained previously, according to Marx's theory, capital, as the structuring principle of underlying social relations, both generates and prevents the possibility of its own supersession. Critical theory's referent of critique, although generated within capitalist society, points outside existing social conditions. In other words, critique's conditions of possibility are socially constituted by the dynamic and contradictory nature of capital itself. For example, the environmentalist ideal of having enough, as opposed to having more, is a real possibility generated by the enormous wealth-producing capacity of industrial capital. But capitalism, as a system of social organization premised on producing more and more *ad infinitum*, simultaneously undermines the possibility that such an ideal will become actual. This contradiction between immanence and transcendence is what normatively compels and analytically enables critical theory to develop tools capable of elucidating critical recognition of the problematic features of modern capitalist society and the related consequences that result from how our lives are created (Postone, 1993; Strydom, 2011).

Critical theory, then, is not a general theory but rather a method of analysis whose core is immanent critique (Antonio, 1981). To reiterate, critical theory confronts the twofold task of critique and transformation via immanent critique, which begins by accounting for its immersion in history. But critical theory is not merely descriptive; it also seeks to specify the possibility of qualitative social transformation, which is necessary if society is to relate to ecosystems in a less ecologically destructive manner.

5.3.2 Mediation

A key shortcoming of the ESVn criticisms discussed above is their inability to deal with *social mediation*. By social mediation I am referring to processes whereby social structure constitutes and is constituted by human actors (see Postone, 1993: 216-225). The emphasis on social mediation, which is integral to the methodology of critical theory, is motivated by the recognition that, in modern capitalist society, social relations are constituted in alienated form.[5] Although ecological degradation is a product of human activity, in modern capitalist society, such activity is conditioned by abstract social forces, which appear to be beyond human control (Biro, 2005; Vogel, 2011). Agricultural production, as I have argued, is not simply the immediate interaction among farmers and ecosystems but is strongly mediated through state agricultural policies (e.g., biofuel blending mandates) that are themselves constituted in response to a social dynamic (e.g., the tendency of agriculture to oversupply) that farmers participate in by do not themselves direct.

For Marx, alienation is the foundation of the entire complex of social relations under capitalism, and as such it comprises a set of mediating processes between subject and object. Marx developed his (unfinished) theory of alienation through a critique of G.W.F. Hegel and Adam Smith, and in the process he identified and analyzed the intrinsic contradictions constitutive of modern, bourgeois society (Dahms, 2006: 11).

Marx's category of alienation refers to an inherently dynamic set of social processes, constituted by the capitalist mode of production that, in addition to estranging humans from nature, self, consciousness, and others, simultaneously rewrites reality so as

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[5] The mediation between humans and ecosystems is a basic property of human labor, whereby both humans and environment are transformed in the process of meeting a given end. In modern capitalist society, however, human labor takes a particular form, which Marx (1988 [1844]) termed alienation (Stoner and Melathopolous, 2015).

to inhibit these very same humans from consciously recognizing that this estrangement is indeed the case

One of the advantages of Marxian critical theory over ESVn and its critics lies in its explicit recognition and critique of alienation. Neo-Marxists, for example, identify ESVn with alienation by pointing out that while technical development appears to be leading to sustainability, these technical developments ultimately give rise to entirely unsustainable outcomes (see Section 5.2.2, but also Robertson, 2012). However, I do not regard this contradiction as proof that ESVn is “wrong”. Rather, this shortcoming is a crucial symptomatic fragment of how ecology and society *actually* relate in the present and in this moment of reflection, how they could relate differently in the future. The task, therefore, is not one of flatly opposing ESV. Rather, thought and action must draw on its immediate or superficial appearance so as to press beyond it – to grasp how seemingly unrelated dimensions of social experience are *actually* connected and how *apparently* related dimensions (e.g. ecosystems function and ecosystem service value) are deeply mediated through an alienated social structure.

I suggest Marx’s critical theory as a starting point from which interdisciplinary research efforts might begin to address elusive processes of social mediation. Indeed, the focus on social mediation is an insight shared by successive generations of critical theorists (e.g., Lukács, Adorno, Postone) whose work represents the continued relevance of Marx’s theory for analyzing the contemporary world.

Marx’s critical theory is also a theory of praxis in that it aims to specify exactly how, through concrete forms of social practice, both subject and object are produced.

"Praxis," in this sense, refers to the process whereby forms of social objectivity and social subjectivity are socially constituted *simultaneously*. In other words, praxis, as such, can be analyzed and understood only in terms of structures of social mediation (Postone, 1993: 218, 220). Indeed, it is on this basis that Marx elucidates the link between epistemology and normative action as being rooted in the structure of social relations (Postone, 1993: 219). Since the criterion of validity is social rather than absolute (Postone, 1993: 219), Marx is able to ground his critique without collapsing into relativism—the standards of critique are a function of existing social reality. Again, the dynamism of Marx's theory is grounded in the dynamic and contradictory nature of capital—a historically specific motion generated from within the social context of which the theory itself is an integral part.

Marx's theory of mediation therefore offers an important corrective to the criticisms of ESVn discussed above. In contrast to neo-Marxists who ground their criticism of ESVn in an underlying subjectivity that is veiled or undermined by objectifying the value of ecosystems, the focus on mediation illuminates such market valuation practices as socially constituting activity. Market valuation of ecosystems is real, to be sure. Yet, Marx's focus on mediation, as discussed above, is also different from those of liberals. Liberal critics are unable to square their assertion that markets reflect society with their attempt to separate a sphere of politics and ethics within which environmental politics should take place. Such a duality of social life need to be recognized as constituted by people through concrete social practices, which are grasped by the categories (value, etc.) of Marx's critique (Postone, 1993: 220). The pragmatist attempt to generate new mediating practices by foregrounding intersubjective communication (e.g., adaptive management) ignores the constraints imposed by these categories and, as such, unwittingly becomes a means of accommodating transformations necessitated by capital.

5.4 CRITIQUE AND TRANSFORMATION

Critique must go beyond simply describing how the shortcomings of ESVn (e.g., by describing how ESVn redistributes power and wealth (neo-Marxists), undermines political activity (liberal) or reinforces rigid ideologies (pragmatist)). Rather it must press forward to understand the meaning of ESVn as an expression of the changing structure of society itself. This is because for critical theory transformation is not a matter of contingency (i.e., the idea that social change comes about by unearthing 'wrong' thinking), but a property of a deeper dynamic within society itself. Critique would need to understand ESVn not as an accidental or irrational feature on an otherwise unproblematic social whole, but as an expression of what society *is* and what it *could become*. A critique of ESVn, in other words, would need to go beyond describing its effects on society and ecosystems and be able to understand it as the product of tensions and pressures *emerging from this social dynamic*. Insight about the character of this dynamic could then be used to theoretically illuminate the kind of social transformation that is in fact *possible*.

The ESVn criticisms outlined in this paper have been considered in relation to the question of transformation. Using the methodological motifs of critical theory discussed above—immanent critique and mediation—I can discern three features, which are conspicuously absent in recent attempts to ameliorate societally-induced environmental degradation: 1) a theory of historical dynamics that grasps how ecologically destructive forms of society emerge, transform and reproduce (*against* the neo-Marxist claim of the necessity of "resistance"), 2) a political practice that can render this structure increasingly comprehensible from the starting point of current discontents (*against* the liberal claim of the identity of politics and markets), and 3) a way of mediating the relations between thought and these political forms that does not simply become a hardened ideology (*against* the pragmatist assertion that the overcoming of ideology is merely a matter of correct communicative procedure). The expression of these features—absent in the ESVn criticisms discussed above—allows me to identify, albeit very coarsely, key dimensions of our ecologically destructive form of society as a necessary precondition for the *possibility* of transformation beyond the present form of society. Rather than providing a set of broad prescriptions for reconciling the antagonism between modern capitalist society and ecological well-being, these insights need to be understood as a starting point from which political actors might pursue effective socio-ecological transformation. Although the specifics cannot be predetermined at the outset, such transformation must allow free rational human control over the ceaseless process of capitalist production that characterizes modern society, and by extension, the ecosystems we depend upon.

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CHAPTER 6 CONCLUSION: LOCATING THE SOCIAL BASIS FOR CONSERVATION IN AND THROUGH ECOSYSTEM SERVICE VALUE (ESV)

6.1 ABSTRACT

The magnitude of ecosystem service value (ESV) is frequently taken as expressing the social necessity for conservation. Yet ESV-based approaches have failed to mobilize significant investment in conservation and are theoretically unable to explain how social welfare can expand as ecological systems are degraded. Using the example of insect pollinator ESV of agricultural production I identify key methodological problems which partially explain these shortcomings of ESV-based approaches. However, a focus on methodological problems can lead to the conclusion that ESV fails to motivate conservation because of largely technical shortcomings, notably that it does not adequately model pollinator benefits relative to the dynamic character of agro-ecological systems. Yet a focus on the technical dimensions of ecosystem service valuations assumes the increased popularity of ESV is largely independent of its social context. Pollinator ESV, however, has been broadly reproduced within the media and popular culture and this response has shaped pollinator protection initiatives. I argue that our capacity to understand the current grounds for conservation cannot advance without simultaneously recognizing how the dynamic nature of agronomic practices are

generative of social tensions, such as low farm income, that find their expression in society through popularized forms of ESV.

6.2 INTRODUCTION

> *A bustling city at dawn. Industrious workers set out from their homes. Coming and going in a perfect and productive ballet. But by evening the workers vanish. No trace of foul play. No bodies left behind. Mass disappearances like this have recently occurred across the globe, not of humans, but of millions of honey bees*[6].

The ominously titled 2007 Public Broadcasting Service documentary *Silence of the Bees* begins with a montage of the streets of a major U.S. city that had grown silent because its inhabitants vanished. The city, we are told, is a metaphor to describe Colony Collapse Disorder, a syndrome of commercial honey bees that has resulted in massive colony losses beginning in 2006.

A few minutes into the documentary we are informed that the metaphor should be considered more literally, as "the bee's disappearance could have colossal repercussions for humans". As the documentary continues, a chorus of honey bee experts proclaim the apocalyptic scale of the unfolding crisis, as bees "account for one third of the food

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[6] Doug Schultz, *Silence of the Bees* (Educational Broadcasting Corporation, 2007), 50 min., 40 sec.

produced in America". One suggests that "unless we only want to eat corn, wheat, and rice we need bees". Another supposes that "without bees, life as we know it won't exist."

As I have explained previously, the claim that bees "account for one third of the food produced in America" exemplifies an increasingly prominent approach to motivating the conservation of ecosystems. This ecosystem service value (ESV) approach seeks to locate the aspects of ecosystems (e.g., the habitat adjacent to pollinator-dependent crops) that are associated with human welfare (e.g., food production), so as to transform the long backlog of conservation into a social necessity. As Paul Armsworth and his colleagues (2007: 1383) point out in their defense of ESV, "nature for nature's sake resonates only with the already converted," whereas "business interests, farmers, and the billion humans living in rural poverty remain unwilling or unable to move." Along with other advocates, they envision ecosystem service valuations as helping us move beyond the pitted debates of hardened conservationists and the rest of society by specifying the grounds for making the latter "partners in conservation (…) providing a means of motivating and enabling them" (1383). They foreground research linking wild pollinators and crop yields across the breadth of agricultural production—from northern California watermelons (Kremen et al., 2004) to Costa Rican highland coffee (Ricketts et al., 2004), to oilseed rape (canola) on the Canadian prairies (Morandin and Winston, 2006)—suggesting such work in particular holds the promise of "restoring and reemphasizing the fundamental links between nature and human well-being" (1384).

Certainly, as the documentary demonstrates, pollinator service valuation has been successful in advancing the cause of pollinators far beyond the "already converted." The phrase that bees "account for one third of the food produced in America"—originating from a recent effort to determine the importance of pollinating insects to food production in light of declining pollinator stocks (Klein et al., 2007) (Chapter 2)—has been reproduced across an impressive range of cultural forms (e.g., news segments, documentaries, movies, novels, pop music) in a sustained manner for almost a decade. Moreover, media expression of the necessity to conserve pollinators is far more strident than that of scientists, frequently assuming the kind of apocalyptic tone exemplified in the opening sequence of *Silence of the Bees*. Yet in spite of the success of impressing the urgent necessity of pollinator conservation—far beyond the "already converted" (Armsworth et al., 2007: 1338) —ecosystem service valuation has proven to be a less than adequate "means of […] enabling" (Armsworth et al., 2007: 1338) the conservation of wild pollinator habitat. Although far fewer people would question the necessity of pollinators today than a decade ago, it has failed to translate into substantive investments in either privately- or public-funded conservation, let alone into a commitment to long-term monitoring of wild pollinator populations. Paradoxically, this lack of investment is most visible among farmers growing pollinator-dependent crops who appear to expand their output in spite of a purported lag in the growth of pollinator stocks (Aizen and Harder, 2009; Breeze et al., 2011; Breeze et al., 2014b; Garibaldi et al., 2011a; Lautenbach et al., 2012).

In this concluding chapter I address the question of why the successful and broad proliferation of research and initiatives—under the rubric of ESV—has resulted in such tepid conservation outcomes. I begin by revisiting my review of pollinator benefits to agriculture in Chapter 2, where I demonstrate that the underlying social necessity implied by pollinator ESV calculations rest on assumptions that are not empirically supported using the example of oilseed rape. Next I regard how ESV fails to account for the agronomic context that individual growers operate within, exemplified by my research in Chapters 3 and 4 on pollinator benefits to lowbush blueberries. The findings from these initial chapters could lead to the conclusion that efforts to price the ecosystem services of pollinators fail to translate into conservation because they do not adequately model pollinator benefits relative to the dynamic character of agro-ecological systems. Yet criticism at this level cannot explain the mass cultural response to the message that bees "account for one third of the food produced in America." Any attempt to merely expose ecosystem service valuation efforts, as well as their public reception, as false, unscientific or irrational falls prey to what I characterize in Chapter 5 as a *hypothetical* approach. Such criticism assumes that transformation in society is merely a process of changing ideas and, as such, fails to consider how ideas themselves are grounded within a socio-historical context. By failing to consider the interplay between the intention to change society and socio-historical context, critics are liable to reproduce the unintended outcomes that have plagued ESV approaches to date. Instead I argue for the need to consider the idea of ESV critically by regarding the resonance of the message that bees "account for one third of the food produced in America." I determine that the broad appeal of pollinator ESV does not signal an approaching consensus on the need to protect wild pollinator populations, but rather the opposite: a deeply integrated social process whereby ecological degradation paradoxically accelerates with increasing awareness of this degradation. I conclude by drawing out implications for future research in pollination ecology and social theory.

6.3 FAILING TO ADD UP: POLLINATOR DECLINE AND SOCIAL NECESSITY

My review of pollination ESV in Chapter 2 examined four assumptions of all national and global valuations that fail to capture the role of pollination in crop production. In spite of the massive value attributed to the activity of pollinators, I demonstrate that we actually know very little about how current agricultural value would change if pollinator populations declined. Oilseed crops (e.g., soybean and oilseed rape) likely represent the most striking and significant example of this uncertainty, because they constitute the bulk of the pollinator ESV estimates in many countries. All extant pollinator ESV calculations assume, for example, that current levels of oilseed rape yield would drop by 20% (Gallai et al., 2009) to 50% (Calderone, 2012) if pollinators were not present. But the dependency of both soybean and oilseed rape yield on pollinators is assumed to be static in spite of tremendous variation across the vast areas over which these crops are grown, not only with respect to pollinator abundance, but also crop management practices, crop varieties and environmental conditions. Moreover this static quality is completely at odds with the historically dynamic character of these crops.

6.3.1. Pollination in flux: the case of oilseed rape

While twenty five years ago Canadian oilseed rape (canola) was largely a mixture of two species (the self-incompatible *Brassica rapa* and self-compatible *B. napus*) that were generated using traditional line-breeding methods, today only *B. napus* remains and is largely composed of hybrid varieties which rapidly obsolesce within a matter of years of their release (Brewin and Malla, 2013; Veeman and Gray, 2010; Wood et al., 2013). Moreover, there has been a dramatic shift in cropping systems over this time, particularly with regards to the adoption of zero-till seeding technology and the incorporation of herbicide-tolerant traits (Brewin and Malla, 2013; Veeman and Gray, 2010). Although researchers have pointed to historical trends suggesting a startling increase in insect pollinator dependency on global food output since the 1990s (Aizen et al., 2008, 2009; Garibaldi et al., 2011a), these apparent trends assume that all crops, including oilseed rape, are equally dependent on pollinators across space and time. But such an extrapolation fails to regard that, for all intents and purposes, the oilseed rape grown in the early 1990s is very different crop from that growing in fields today.

The inability to regard the changing character of a key crop such as oilseed rape may be one reason why researchers are unable to explain how the aggregate per hectare yield among ostensibly pollinator-dependent crops has experienced such an impressive and steady increase since the 1990s in spite of pollinator decline. In fact, in the specific case of oilseed rape, its per hectare yield has dramatically outpaced pollinator-independent crops that are grown in direct rotation with it (e.g., cereal crops) (Brewin and Malla, 2013; Veeman and Gray, 2010). Although the causes of this relative yield increase are complex, an undoubtedly important dimension is the growing reliance on high-yielding hybrid seed (Brewin and Malla, 2013; Gray, 2014). Hybridity in oilseed rape provided insect pollinators with a new job, since the production of this seed (F_1 seed) requires insect pollinators to transfer pollen from spatially separated intercropped male-fertile plants to male-sterile plants (Westcott and Nelson, 2001). This new role enables pollinators to enhance yield but not in ways captured in any ESV estimate, which focus only on the role of insects when pollinating commodity oilseeds for oil and seed meal (F_2 seed); hybridity boosts yield relative to traditional open-pollinated propagation methods. But in contrast to the insensitivity of managed pollinator stocks to the growing output of the commodity crop (F_2 seed), the need for insect pollinators to produce the hybrid seed crop (F_1 seed), starting in the 1990s, translated into a well-defined market for two species of managed pollinators (honey bees and alfalfa leafcutter bees).

While the purported lag between the density of managed honey bees and the area planted to oilseed rape appears to signal a “mismatch” that generates the necessity for “green infrastructure” (Breeze et al., 2014b), the expansion of yield through heterosis shows this “necessity” to be far from necessary. The assumed constraints on commodity oilseed expansion, as a function of managed pollinators implied by a “mismatch” hypothesis, is directly challenged by the fact that in regions where the production of hybrid oilseed rape seed has expanded, there has been a dramatic increases in managed pollinator stocks. For example, in the province of Alberta, Canada, where the production of hybrid oilseed rape seed is focused, the number of managed honey bee colonies has doubled (148,000 colonies in 1993 vs. 280,000 colonies in 2013; Statistics Canada) while the prediction of the profitability of pollinator conservation adjacent to oilseed rape fields has made no practical inroads (Morandin and Winston, 2006). Consequently, the necessity implied by pollinator ESV methodologies appear hampered by their inability to conceptualize the dynamic character of agricultural production, particularly for a crop such as oilseed rape that has witnessed a massive social investment in biotechnology (Bonny, 2014; Brewin and Malla, 2013; Gray, 2014).

6.3.2. Managing pollen limitation dynamically: the case of lowbush blueberry

My research in lowbush blueberry demonstrates that the dynamic manner by which agricultural production responds to pollinator shortages is not restricted to the most technologically-advanced sectors of agriculture but also applies in crops in which opportunities related to pollinator service management are severely constrained. As a wild plant, commercial lowbush blueberry lacks the benefits of crop improvement that come from selective breeding (let alone hybridity). Moreover, field establishment is only minimally managed through use of selective herbicides that remove competing vegetation to help perpetuate growth of blueberry plants. But even in this crop I was able to challenge the central assumption of current approaches to pollinator valuation: that pollen limitation is the overriding factor determining yield. My factorial experiment in Chapter 3 revealed that pollinator abundance and pesticide use are both required for high fruit set. Moreover, I demonstrated the potential of blueberry plants to compensate for low pollinator visitation by increasing berry size, but this compensatory response was only realized in the presence of pesticide use.

Although analogous findings have been demonstrated in other cropping systems using a similar factorial design (e.g., Groeneveld et al., 2010; Klein et al., 2015; Lundin et al., 2013; Marini et al., 2015) the limitation of such studies are that they only demonstrate the *potential* for an interaction between pollinator activity and various other inputs, not the actual interaction that exists across a crop in given year. Consequently, such factorial designs are of limited use in being able to explain actual yield losses, and by extension agricultural value, attributable to given levels of pollinator decline. In Chapter 4, I address this through a field study that captures the actual interaction between pollinator densities and other agronomic factors in commercial fields. While this study confirmed that pollinators play an essential role in fruit set, final harvested yield was not strictly pollen limited, challenging the central assumption that underpins all current pollinator ecosystem service valuation efforts. In fact, in the first year of the study, variation in the abundance of pollinators failed to explain the levels of fruit set across commercial fields. Moreover, while wild pollinators were more strongly associated with higher fruit set, this was largely correlated with the availability of a very restricted set of pollinator taxa. The lack of pollinator diversity associated with yield benefits in lowbush blueberry does not lend itself to the necessity for diverse or expansive wild pollinator conservation, but rather to efforts focused on a small subset of species (primarily *Andrena* spp.) whose life histories are strongly linked to the phenology of lowbush blueberry. Such findings support conclusions that the goal of enhancing wild pollinator contributions to agriculture may be largely disconnected from the more generalized aim of wild pollinator conservation (Kleijn et al., 2015).

My analysis of the theoretical link between pollinators and oilseed production in Chapter 2 and my empirical analyses of the role of pollinators in lowbush blueberry in

Chapters 3 and 4 reveals that the supposed link between pollinator conservation and social welfare is much more complex and dynamic than assumed in the current pollinator dependency construct and allied pollinator ESV calculations. These examples are consistent with an emerging recognition that efforts to price ecosystem services cannot adequately account for how welfare expands in parallel to increased ecological degradation: what Raudsepp-Hearne et al. (2010) characterize as the “environmentalist paradox.” Yet this recognition alone is insufficient, because, as I argue in Section 2.7, the valuation of pollinator ecosystem services may not merely represent a methodology for accounting how pollinators and society interact, but a methodology whose proliferation largely reflects a change in how ecological degradation is recognized (and rationalized) within society itself. In this way, I argue that despite decades of effort on pollinator service valuation, the resulting enormous sums quantified, and the broad-based cultural awareness induced, the failure to achieve meaningful gains in wild pollinator conservation cannot be resolved by simply confronting technical deficiencies in ESV calculations. A focus on debunking pollinator ESV based on the findings from Chapters 2-4, might only redirect research to better specify the interaction between plant and pollen limitations on yield (e.g., Hanley et al., 2015). This only characterizes the agronomic/biophysical context at the expense of understanding the broader, and at least as dynamic, social context through which conservation currently operates. I argue that the failure of ESV to bring about substantial investment in conservation is due to our poor understanding of the social basis for wild pollinator conservation, and that this hinders our ability to answer the agronomic/biophysical question, "how much variation in yield is explained by variation in pollinator abundance and diversity?".

6.4 WHAT NECESSITY DOES ECOSYSTEM SERVICE VALUE (ESV) INDICATE?

Given that ESV does not translate directly into claims for the social necessity of wild pollinator biodiversity, it has nonetheless resulted in an expanding circle of those concerned with the plight of pollinators. This concern has certainly not gone unnoticed by policy makers. Significantly, the message that bees "account for one third of the food produced in America" figured prominently in the opening lines of all the pollinator protection bills introduced in the U.S. House of Representatives and Senate (e.g., *The Pollinator Protection Act* (H.R. 1709 (2007), *The Save America's Pollinators Act* (H.R. 2692 (2013)) in advance of the omnibus 2008, and 2014 Farm Bills. Also, legislators drew heavily on recent results of pollinator service valuation work to explain the social need for such legislation. As co-sponsor of the 2013 *Pollinator-Beneficial Farm Bill Amendment,* Congressman Alcee Hasting explains, "How I'm fond of putting it is: If you don't have no bees, we ain't going to have no food" (Peterka, 2014). Yet legislation such as the one advanced by Hastings, which would have mandated long-term monitoring of wild pollinator populations through coordinated efforts of federal agencies, have largely not survived intact in the final Farm Bill (Coenen-Davis, 2009; Peterka, 2014). Certainly the final tally of public support for pollinator initiatives in the U.S. comes nowhere close to the $100 million paid by the U.S. government to beekeepers in 1988 at the height of the Honey Price Support Program when awareness of the role of pollination in agriculture and the absolute scale of pollinator contribution to total farm output was arguably far lower (Muth and Thurman, 1995; Robinson et al., 1989) (see Section 5.2.2). There appears to be a considerable problem translating increasing awareness into effective political action on pollinator conservation.

Moreover, there remains little evidence that pollinator conservation is progressing at any significant rate without state-funded agri-environmental programs, in spite of reported benefits to farm output (Breeze et al., 2014a; Hanley et al., 2015). In fact, the opposite may be happening. The "business interests (and) farmers" (Armsworth et al., 2007: 1383) have proven to be quite reluctant "partners in conservation" since even sectors focused on highly pollinator-dependent crops invest virtually nothing in habitat conservation (Ghazoul, 2007; Hanes et al., 2013; Munyuli, 2011). Disproportionately, it is not in the area of public discourse, but at the farm level where ESV has failed to make serious inroads. Perceptive observers have sensed this problem. As Ghazoul (2007: 220) warns, "overemphasizing the economic benefits of ecosystem services without due recognition of trade-offs and opportunity costs or of other forms of valuation may risk undermining conservation efforts, as well as the credibility of conservationists, if economic outcomes turn out to be less than favourable." But the manner in which this credibility has been undermined has taken on a specific form that was clearly not anticipated by Armsworth (2007: 1383) and his colleagues. The farm sector has watched as public support for pollinator protection failed to make gains on existing state subsidies for agricultural conservation programs (Pe'er et al., 2014; Potter, 2009), but considerable new restrictions on the use of insecticides, resulting in significant conflicts between agrochemical companies and major farming associations on one side, and environmental and conservation groups on the other (Copping, 2013; Goulson, 2013; Gross, 2013; House of Commons Environmental Audit Committee (U.K.), 2013; House of Representatives Subcommittee on Horticulture, 2014; Standing Senate Committee on Agriculture and Forestry (Canada), 2015). Paradoxically the attempt by advocates of ESV to move beyond the confrontational character of 1970s and 1980s environmental politics—the impulse to locate "partners in conservation" (Armsworth et al., 2007: 1383) within the farm sector—seem to have simply reproduced these confrontations anew. Environmental activists, just as much as farm-sector advocates, seem more likely to engage in what Koh et al. (2010) term a "wash and spin cycle" in which industry "greenwashing" around the benefits of farming practices to conservation is increasingly countered by an equally spurious "blackwashing" by activists. Contrary to the goals of ESV advocates, clarification of the social basis upon which conservation could be accomplished—advanced under the pretext of ESV—readily sinks into the apolitical "wash and spin" of interest groups that liberal and pragmatist critics of ESV specifically warn against (Section 5.2).

One response is to identify the "wash and spin cycle" that arises in response to ESV efforts directly, by taking up the banner of "écrasez l'infâme"[7] and advancing a

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[7] "Crush the infamy" – the response of the eighteenth century French philosopher Voltaire to the persistence of superstitious thought in the wake of the Enlightenment expressed in a letter to Jean le Rond d'Alembert (28 November 1762).

well-crafted argument that targets the uncertainty of current pollinator ESV estimates. ESV advocates might conceivably counter by stating that the "wash and spin cycle" is an irrational response to an otherwise rational endeavor to calculate the social benefits of wild pollinators. Yet it is precisely because the technical dimensions of the ESV calculation are disconnected from society's understanding of ESV estimates that enables ecologists to focus on methodological questions at the expense of questions of efficacy (Laurans et al., 2013). These responses indicate that in spite of many differences, both ESV critics and advocates share a *hypothetical* approach in that they assume that transformation in society is merely a process of changing ideas and, as such, they fail to consider how their ideas are grounded within a socio-historical context (Chapter 5). It is telling that neither ESV critics nor advocates appear particularly interested in asking what the broader response to the value being placed on pollinator conservation means about the character of society itself.

Ideas (or theories) and social context are not simply opposed, but would need to be linked for social change to be actively directed. Failing to understand the social context that propelled pollinator valuation efforts far beyond the "already converted," practitioners discover that their attempts to locate "partners in conservation" (Armsworth et al., 2007: 1383), has the unintended consequence of creating more social friction than unity over conservation priorities. But such failures may provide insight if the meaning of the failure could be critically regarded. Dahms (2008: 41) remarks that a hypothetical approach is based on the assumption "that modern society is an internally consistent form of social, political, cultural, and economic life and organization", whereas in fact, "actually existing modern societies are *constitutionally irreconcilable*".

This claim to society being "constitutionally irreconcilable" is evident immediately below the surface of pollinator ESV methodology. As I describe in Section 2.3.2, the most significant development in pollinator ESV over the past twenty-five years was the advent of the Insect Pollination Economic Value (IPEV) methodology. IPEV was first employed to build public and legislative opposition to proposed cuts to the Honey Price Support Program (Muth and Thurman, 1995; Robinson et al., 1989). The context for the proposed cuts, however, extended far beyond the particular interests of U.S. beekeepers and was a part of a far-reaching social transformation in which the restructuring of agricultural policy was merely one facet[8]. But in an important sense

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[8] The long period of social stability in industrial countries following the Second World War came to an abrupt end at the beginning of the 1970s. This crisis had multiple dimensions and was felt internationally, but a key feature took the form of a severe economic crisis in the U.S., the key economic engine of the post-WWII reconstruction. A mixture of high inflation and persistent unemployment (stagflation) took hold at the end of the 1960s prompting the Nixon government to implement wage and price controls in 1971. These economic problems were immediately compounded by a steep rise in oil prices following an energy embargo by the Organization of Petroleum Exporting Countries (OPEC) in 1973. The economic slowdown strongly conditioned the restructuring of farm income support programs, particularly in the 1985 U.S. Farm Bill and changes to the E.U. Common Agricultural Policy in 1986 (McGranahan et al., 2013; Potter, 1998). A key dimension of these changes was the implementation of agri-environmental programs with an explicit commitment to conservation rather than farm income support.

IPEV constituted one form in which the sweeping changes in society appeared to people in that moment. As I will argue below, the extent to which results of the application of the IPEV method resonated in society, it did so as an expressed form of consciousness. By consciousness I mean that the idea that the public should support pollinator populations, rather than beekeeper incomes, is one of the ways in which the deeper changes taking place within society in the 1980s were recognized.

Yet the idea that pollinator conservation might be uncoupled from farm income support was readily integrated back into the longstanding problem of chronically depressed farm income through the development, in the 1990s, of agri-environmental programs (Cain and Lovejoy, 2004; Finegold, 1982; McGranahan et al., 2013; Potter, 1998). Although such programs have continually promised to tie subsidies to the production of environmental public goods and services (and not food), the stubborn problem of farm income has continually undermined these ideals resulting in underwhelming conservation outcomes relative to overall program expenditures (Kleijn et al., 2011; Pe'er et al., 2014). The consciousness attending IPEV, rather than developing into an opportunity for society to bring the longstanding problems with agricultural production to the surface, readily became a means of avoiding these problems. By assuming that the problem of farm income and conservation is "internally consistent," at the expense of recognizing the extent to which they are "irreconcilable," the social dimensions of these contradictions can never be worked through. In this manner, pollinator valuations simply served to provide cover and a rationalization for changes that were seemingly already in motion. The ultimate expression of how such consciousness both perpetuated ecological degradation and obscured the problem of low farm income was the rise of biofuel blending standards in the 1990s (Chapter 5). A key contention of my work is that consciousness that merely reflects its social context, rather than critically regarding it, invariably results in unintended consequences.

6.5 LOCATING THE WHOLE FROM THE PARTS

If the social necessity for ecological conservation cannot be expressed in terms of the ESV calculations themselves, the critical approach I outline in Chapter 5 suggests it can only be located through understanding what generates the appeal of the message that bees "account for one third of the food produced in America". Not only has there been sustained (2006-2015) mainstream news reports on the poor state of pollinator populations, but as the opening scene of *Silence of the Bees* makes clear, ESV has fused with broad and general anxieties of contemporary life. Consequently, the plight of pollinators and their purported significance to welfare has not only been prefaced in all the acclaimed feature length documentaries that have emerged over this time (e.g., *More than Honey* (2012), *Queen of the Sun* (2010), *Who Killed the Honey Bee?* (2009), *Vanishing of the Bees* (2009)), but also more diffusely through the realm of the "everyday", appearing everywhere from globally syndicated television programs (e.g., *The Simpsons* episode "The Burns and The Bees" (2008)) to dystopic novels (e.g., Douglas Coupland's *Generation A* (2009), and Margaret Atwood's *The Year of the Flood* (2009)). In the domain of pollination service provisioning, ESV has successfully transcended the rarefied world of ecologists and activists and now assumes an independent quotidian existence in mass culture.

To be sure, the mass appeal of efforts to price ecosystem service represents a theoretical opportunity to resolve the problem of the ecologically destructive character of society, but not one that immediately leads to a solution. The popularity of ecosystem service valuation efforts is bound up with diffuse anxiety over *actual* pressures and tensions that have festered within society and which valuation efforts attempt to connect together, but ultimately are unable to adequately clarify. Borrowing a phrase from the critical theorist Theordor Adorno (2002: 162), the problem with valuation efforts is “the fallacy not of the material thus interconnected, but of the spuriousness of the link”. A critique of ESV must draw attention to the “spuriousness of the link” in such a way that can practically advance consciousness toward the underlying social tensions that ESV is giving expression to (“the material thus interconnected”). In this sense I suggest that pricing ecosystem services is not simply a rational straightforward calculation of how humans value ecosystems, but a surface-level expression of deeper social tensions that needs to be brought to light. These tensions are exemplified in the origins of pollinator ESV calculations, which, in the 1980s, attempted to mediate at least two seeming irreconcilable issues: a) maintaining farm income support benefits for beekeepers in light of a broader economic downturn (e.g., high state debt and unemployment, double-digit interest rates, low farm prices); and b) a general discontent with ecological degradation in agricultural landscapes (e.g., the persistence of the 1970s environmental movement). In this sense the continued growth of the valuation phenomenon must be understood as the means by which many interlocking social tensions *appear* connected to the role of pollinators in generating agricultural surpluses. That so many people in society view the problem of pollinators in this way does not represent an illusion or artifice, but is the necessary form of the appearance of the problem of pollinators and society given our current social context. This context is not only characterized by persisting problems, such as the chronic tendency of agricultural productivity to undermine prices (first expressed on a global scale with the collapse of grain prices in 1921) and concerns over ecological degradation (marked by the first Earth Day in 1970), but new issues such as the lingering dissatisfaction around the public regulation of resources, populist discontents around the increasing influence of corporations, and the broad failure of environmental politics in the 1980s.

A critical approach would start from the premise that underlying contemporary tensions within society can only be practically understood in, and through, a reflection of existing forms of mass consciousness, in this case the consciousness of society embodied in the broad appeal of ESV. Such a critical approach is fundamentally different from liberal criticisms that identify the problem of pollination valuation efforts in how they distort the actual economic activity that a bee generates with each visit she makes to a flower. It also does not call for "resisting" ESV initiatives on the grounds that they constitute deviation, or "fiction", from the "natural" state of affairs as neo-Marxists do. Nor does it follow the pragmatist criticism of circumventing the problems that arise from valuation efforts by deferring to social values, which ultimately leads directly back to ESV in mass culture. Instead, the task of critical theory is to connect the consciousness of society embodied in the phrase that bees "account for one third of the food produced in America" to the deeper social tensions that ultimately renders pricing of ecosystem services a plausible account for how society and ecosystems interact.

6.6 FUTURE RESEARCH IN POLLINATION ECOLOGICAL AND CRITICAL THEORY

Moving forward, my work sets important tasks for pollination ecology and critical theory in engaging the question of the role of ecosystem service valuation if they are to go beyond the numerous one-sided criticisms of ESV approaches. Ultimately, the focus of this work ought to be to take existing consciousness and determine how it might lead to a more adequate social and political expression of the deeper structural constraints that not only block the possibility for conservation, but that are interconnected with other dimensions of social life that similarly appear blocked. While such a goal is well beyond the scope of the present work, I conclude with a sketch of some research directions that I feel could provide important clues on how to proceed.

I envision future pollination ecology research to go beyond the work presented in Chapters 3 and 4, where I demonstrate how pollen limitation cannot be understood independently of its agronomic context. I believe it is possible to further contextualize pollinator contributions to agricultural output, particularly in relation to the broad socio-historical transformations that ultimately condition how society and ecosystems have come to interact. For instance, while the total and relative output of crops that are dependent on the activity of pollinators has been observed to rise steeply since the 1990s

(Aizen and Harder, 2009), there has been no research designed to explicitly describe the characteristics of this expanded output. My work suggests that given the problems associated with how we estimate the dependency of agricultural yield on pollinators, it remains entirely unclear what exactly these trends reflect. Although the 1990s are associated with far-reaching changes in both the global movement of agricultural commodities as well as consumptive patterns (Friedmann, 1993), this period was also accompanied by a massive investment in biotechnology that likely had an effect on the dependency of key crops (e.g., oilseeds) on pollinators. Pulling apart these factors may be facilitated by a better understanding of what is driving these trends. A very straightforward experiment could characterize how the changing socio-historical context has affected the pollinator-dependency of specific crops that are central to this jump in estimated value. For example, plot experiments could be conducted to observe whether the yield of plant varieties developed in the 1960s, 1990s and the present are similarly dependent on pollen transfer by insects. An excellent model for this type of work would be the apparently pollinator-dependent crops that were early beneficiaries of the massive private investment in biotechnology, for example oilseed rape (Abbadi and Leckband, 2011; Brewin and Malla, 2013; Gray, 2014; Wood et al., 2013). Potentially, breeders of oilseed rape may have intentionally or simply accidentally adapted newer varieties to declining pollinator abundance, resulting in increasing pollinator-independence. Preliminary results, in fact, support this hypothesis with the observation that yield among newer hybrid varieties of oilseed rape (i.e., the plant whose seed is the product of bee-mediated outcrossing) are far less dependent on pollinators than varieties developed by using traditional line-breeding (Marini et al., 2015). Such findings would support the idea that the dynamic transformation of society since the 1970s—exemplified by the transition from plant breeding that was primarily focused on public agricultural research institutions to a small number of well-capitalized private agro-chemical corporations—conditions how society and ecosystems effectively interact. Conversely, a lack of evidence for changes in pollinator dependency over time might allow for research to focus on other facets of the transformations which characterize this period, such as changes in global consumptive and trade patterns.

Along this vein, similar hypotheses could be tested with respect to the apparent historical decline of wild pollinators. The work of Carvalheiro et al. (2013) is exemplary in connecting declines in wild bee biodiversity to phases of industrial development after World War II and its stabilization with the introduction of E.U. agri-environmental schemes in the 1990s. More focused research could be conducted here as well. For example, it would be interesting to consider how the chronic tendency for agriculture towards oversupply (described in Section 5.2.2) may explain historic patterns of wild pollinator abundance and diversity. Certainly, periods of low farm profits, beginning in the 1930s, have resulted in the expansion of state-subsidized programs to remove land from cultivation in order to restrict farm output (Potter, 1998), and such uncultivated land has been associated with increased pollinator diversity and abundance (Kennedy et al., 2013). Conversely, the area of certain mass-flowering commodity crops, notably oilseed rape, has been shown to have a positive effect on pollinator abundance (Holzschuh et al., 2012; Riedinger et al., 2015; Westphal et al., 2003), and the sustained expansion of such crops (e.g., during the peculiar spike in agricultural prices beginning in 2008, which followed almost thirty years of relatively depressed farm income (Timmer, 2010)), followed by their invariable retraction, may have profound regional and global effects on the composition of wild pollinator biodiversity. Consequently, an important direction for research would be to test whether the price of mass-flowering commodities might explain shifts in pollinator abundance and biodiversity in some landscapes. Ideally, an understanding of such shifts would not only include the expansion phase of these crops, but the turn back to agri-environmental programs and state-intervention during periods of low prices, which results in the temporary expansion of cultivation onto more marginal land (Potter, 1998; Potter and Tilzey, 2005). Such work might demonstrate that broad-scale patterns in the abundance and diversity of pollinator taxa over time may, in part, result from broad socio-historical transformations, rendering patterns of global social transformation in the twentieth century pertinent to understanding patterns of natural history.

But describing connections between wild pollinator abundance, biodiversity and socio-historical transformation only approaches the problem of pollinator conservation from its "objective" dimension. There is a continued need for critical theory to press forward and understand the emergence of ESV—and what it indicates about the possibility for understanding the "subjective" and "objective" dimensions of ecological degradation—as being symptomatic of the kind of helplessness portrayed in the opening sequence of *Silence of the Bees*. My work proposes that the emergence of ESV has been conditioned by obstacles encountered by the 1970s environmental movement, which remain poorly understood in theory. Moreover, its popularity has meant that it has also shaped how these obstacles have been subsequently mediated. Clearly, the antagonism in the 1980s between environmentalists and workers whose employment was threatened by environmental protection points to the *need* for theory and the failure to generate forms of environmental politics that could relate such a theory to the prevailing forms of discontents. What is missing, in other words, is not political activity or critical theory *per se*, but recognition that there is no social practice, at present, which could adequately mediate the two in the direction of conscious social transformation.

Although pollination ecosystem service valuation efforts have elevated awareness of wild pollinator declines, it appears entirely unable to facilitate the types of societal changes that could resolve the *anxieties* surrounding this decline, let alone the *actual* decline of wild pollinator biodiversity. This phenomenon of increased awareness coupled to the inability to generate adequate social transformation is something Biro (2011b: 6) observes to be endemic of environmental issues in the present: “while environmentalists’ assertions are increasingly accepted as fact, environmentalism as a movement seems incapable of mobilizing more than anaemic, and often individualized, responses.” It is what I refer to in Chapter 5 as the *environment-society problematic* (from Stoner (2014)). In this account of our social context, social life plausibly *appears* as a swarm of activity to the extent to which we are unable to take hold of the underlying social tensions that repeatedly block meaningful and sustained activity. Biro (2011a: 224) underscores this by pointing out that the problem with contemporary activity around ecological degradation is not that it fails to lead to social transformation, but that this change is “experienced as a kind of 'second nature'" in which "the possibility of consciously reflecting on how [...] transformation might be managed [is] lost".

The problem with ecosystem service valuation and its criticisms is that it asserts a unity between society and ecosystem conservation in the context of competition among farmers, advances in farming technology, and public reception of the pollinator crisis, without regarding how these practices fall outside their theory. This attempt at unification at the expense of working through the interconnected contradictions between theory and social practices is a key problem that actively undermines conservation initiatives and blocks further questioning into the nature of contemporary society and its capacity to change. In the 1960s the critical theorist Theodor Adorno observed that the persistent problem of relating a theory of the total structure of society to social practices was giving rise to what he termed "pseudo-activity" (2005 [1969]: 269), where the separation of theory and practice was falsely resolved by emphasizing activity over thought. For Adorno, this "impatience with theory" further disintegrates the awareness of the need to relate theory and social practices anew, rendering the problem of conscious transformation of society increasingly intractable (Adorno, 1998 [1969]: 292).

Significantly, the modern environmental movement was born in the confusion over the failure of politics that intensified in the 1960s. The "impatience with theory," however, has only grown more acute given the scale of the threat posed by ecological degradation. In fact, what characterizes the response to efforts to price ecosystem services is not substantive transformation or understanding, but "pseudo-activity" in the form of either urgent activism or apathy, or a combination of both[9]. Consequently, future work in the area of critical theory should focus on explaining how the mass cultural phenomenon expressed in the phrase“bees account for one third of the food produced in America” is passively constitutive of current approaches to environmental policy. One focus of this work might be to trace patterns of mass culture and activism around pollinators with respect to changes in agri-environmental policy and pesticide regulation. Current scholarship on biofuels, for example, has demonstrated that while discontents over ecological degradation were integrated into agri-environmental programs to generate support for farm incomes in the 1990s this compact has begun to fracture following the combined surge in food prices between 2006-2011 (Headey and Fan, 2008) and the discovery that the aggressive expansion of biofuel crops has hastened carbon emissions (Gerasimchuk and Koh, 2013; Koh and Ghazoul, 2008; Koh et al., 2009)[10]. This work on

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[9] This sentiment is expressed well in a pithy restatement of Karl Marx’s famous eleventh thesis on Feuerbach (i.e., “The philosophers have only interpreted the world, in various ways; the point is to change it”): “In the past it may have seemed as if philosophers had hitherto only interpreted the world, but today it seems that people seeking to change the world have stopped interpreting it” (Cutrone et al., 2015: 535).

[10] There are attempts underway in the E.U. to renegotiate the renewable fuel directive to better address the issue of GHG emissions by adding so-called Indirect Land Use Change (ILUC) provisions that mandate the amount of second generation non-crop sources of biofuel do not change agricultural land use patterns (e.g., from algae, waste and organic cellulosic residues). Predictably, these new provisions are being vigorously opposed by farm groups since it significantly reduces farm price support dimensions of the initial strategy. Sensing an opportunity, as well as a crumbling coalition, farm groups have responded by arguing that

biofuels points towards a growing gulf between farm policy and conservation that is being rendered even more acute by increasing budgetary pressures following the 2008 financial crisis. Future research should focus on the manner in which the renewed antagonism between farm groups and environmentalists over pollinator protection corresponds to changes in ESV methodologies and the manner in which resulting 'prices' are used in policy debates to address this antagonism.

To be sure, advocates of ecosystem service valuation formulate that the practice provides an antidote to pitted and endless debates that characterized 1980s environmentalism (Armsworth et al., 2007). But the attempt to escape the past, without understanding it, only ends up reproducing it, as evinced by the fact that results of valuation efforts have readily been integrated into the types of protest that ESV advocates explicitly attempt to avoid (e.g., against neonicotinoid insecticides). Rather than avoid the past and risk repeating it, a key role for critical theory is to show how ignoring past failures at transforming society obscures our capacity to even conceive of society as changeable. The emergence of ecosystem service valuation efforts out of the failure of environmental politics in the 1980s and the broad appeal of pollinator and similar ecosystem service valuations needs to be developed as part of larger trend in which social transformation is experienced as something external to society ("second nature") rather

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recent high prices prove the need for national self-sufficiency in food production and have called for a return to direct support of E.U. farmers (Potter, 2009).

than something we consciously bring about. Thus, a key focus of future work should be to better situate the historical emergence of the valuation enterprise in the 1990s as the product of failing to critically regard the deeply contradictory features of contemporary society. If these valuation efforts constitute a “wrong turn” as pragmatist critics point out (Norton and Noonan, 2007: 665), then the temptation must not be to oppose it with a theory that similarly glosses over this socio-historical context. Rather than collapse tentative and disjointed insights into an airtight synthesis, a commitment to critique must anticipate the possibility that the best outcome at this juncture may be to, “remind us that we haven’t gotten anywhere” (Pippin, 2004: 428).

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APPENDIX A SUPPLEMENTARY MATERIAL

Table A. Global positioning coordinates for of commercial lowbush blueberry fields in Prince Edward Island (PEI) and Nova Scotia (NS) used to study pollen limitation in 2013 (*n*=16 fields) and 2014 (*n*=34 fields) (Chapter 4). In all fields in 2013 both transect fruit set (indicated in the 'measurement' column as 'transect') and quadrat yield (indicated in the 'measurement' column as 'yield quadrat') were measured. In 2014 all locations had transect fruit set measurements but only a subset of the fields had quadrat yield.

Year	Province	ID	Longitude	Latitude	Measurements
2013	PEI	1	-62.8941	46.35348	transect + yield quadrat
2013	PEI	2	-62.4594	46.33698	transect + yield quadrat
2013	PEI	3	-62.8575	46.38509	transect + yield quadrat
2013	PEI	4	-62.7616	46.39681	transect + yield quadrat
2013	PEI	5	-62.8069	46.29942	transect + yield quadrat
2013	PEI	6	-62.7551	46.40872	transect + yield quadrat
2013	PEI	7	-62.475	46.32405	transect + yield quadrat
2013	PEI	8	-62.1913	46.39501	transect + yield quadrat
2013	PEI	9	-62.4998	46.23456	transect + yield quadrat
2013	PEI	10	-62.2829	46.46725	transect + yield quadrat
2013	PEI	11	-62.8779	46.35292	transect + yield quadrat
2013	PEI	13	-62.7532	46.05678	transect + yield quadrat
2013	PEI	13	-62.7418	46.01589	transect + yield quadrat
2013	PEI	14	-62.339	46.36619	transect + yield quadrat
2013	PEI	14	-62.7999	45.99151	transect + yield quadrat
2013	PEI	16	-62.4446	46.33401	transect + yield quadrat
2014	PEI	17	-62.9123	46.36382	transect + yield quadrat
2014	PEI	18	-62.4787	46.33608	transect
2014	PEI	20	-62.8835	46.35834	transect + yield quadrat
2014	PEI	21	-62.8389	45.9968	transect + yield quadrat
2014	PEI	22	-62.517	46.32099	transect + yield quadrat
2014	PEI	23	-62.5768	46.44903	transect
2014	PEI	24	-62.7571	46.40303	transect + yield quadrat
2014	PEI	25	-62.8239	46.31077	transect + yield quadrat
2014	PEI	26	-62.7577	46.41001	transect + yield quadrat
2014	PEI	27	-62.7519	46.40446	transect
2014	PEI	28	-62.408	46.34678	transect
2014	PEI	29	-62.408	46.34678	transect

Year	Province	ID	Longitude	Latitude	Measurements
2014	PEI	30	-62.5431	46.45599	transect
2014	PEI	31	-62.2055	46.38781	transect + yield quadrat
2014	PEI	32	-62.2181	46.39155	transect
2014	PEI	33	-62.2861	46.47172	transect + yield quadrat
2014	PEI	34	-62.6831	46.01849	transect + yield quadrat
2014	PEI	35	-63.0916	45.49965	transect
2014	PEI	36	-62.7625	46.05576	transect + yield quadrat
2014	PEI	37	-62.7712	46.05016	transect
2014	PEI	38	-62.8524	46.38616	transect
2014	PEI	39	-62.7438	46.01458	transect + yield quadrat
2014	NS	40	-62.7902	46.29204	transect
2014	NS	41	-63.4945	45.41398	transect
2014	PEI	42	-62.527	46.27163	transect + yield quadrat
2014	PEI	44	-62.3283	46.36723	transect + yield quadrat
2014	PEI	45	-62.4682	46.34734	transect + yield quadrat
2014	PEI	46	-62.4629	46.34215	transect + yield quadrat
2014	NS	47	-63.4811	45.4277	transect
2014	NS	48	-62.8895	45.46275	transect
2014	NS	49	-62.9923	45.49035	transect
2014	PEI	50	-62.5216	46.23543	transect

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