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Theoretical analysis and preference modelling for the valuation of ecosystem services from native pollinators in selected Thai rural communities

Cumulative dissertation

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SCIENTIFIC PAPERS

This doctoral thesis is a *cumulative dissertation* and its body of research consists of the following three scientific articles[1]:

PAPER 1 (Chapter 2)	Narjes, M. E., & Lippert, C. (2019). The optimal supply of crop pollination and honey from wild and managed bees: an analytical framework for diverse socio-economic and ecological settings. *Ecological Economics 157: 278-290*. DOI: 10.1016/j.ecolecon.2018.11.018
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PAPER 3 (Chapter 4)	Narjes, M. E., & Lippert, C. Regional differences in farmers' preferences for a native bee conservation policy: the case of farming communities in Northern and Eastern Thailand. *Manuscript submitted for publication*.

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[1] Cross-references have been added to the scientific manuscripts constituting this thesis and the articles' original numbering format has been modified to fit the span of this cumulative dissertation. Chapters 3 and 4 are otherwise verbatim copies of the respectively published and submitted manuscripts. On the other hand, a revised version of Chapter 2 was published in compliance with the journal's peer review process.

To my parents

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CONTENTS

LIST OF TABLES

LIST OF FIGURES

LIST OF ABBREVIATIONS, ACRONYMS AND SYMBOLS

AIC	Akaike information criterion
ASC	Alternative specific constant
BIC	Bayesian information criterion
DCE	Discrete choice experiment
d.f.	Degrees of freedom
DoAE	Department of Agricultural Economics (Thailand)
EUR / €	Euro
EV1	Extreme value type 1 (Gumbel) distribution
F	F-test
FAO	Food and Agriculture Organization of the United Nations
GMXL	Generalized mixed logit
H_0	Null hypothesis
H_a	Alternative hypothesis $(a = 1,\ldots, A)$
ha	Hectare
I	Number of surveyed individuals (alternatively n or N)
IIA	Independent of irrelevant alternatives
IID	Independently and identically distributed
IPI-POA	International Pollinator Initiative’s Plan of Action
K	Number of estimated parameters
kg	Kilogram
LL	Log-likelihood function
LRT	Likelihood ratio test
ML	Mixed logit (also known as random parameter logit)
MLE	Maximum likelihood estimation
MSL	Maximum simulated likelihood
N	Number of choice observations (alternatively number of surveyed individuals)
OLS	Ordinary least squares
R^2	Coefficient of determination R squared
RPL	Random parameter logit (also known as mixed logit)
RUT	Random utility theory
TEV	Total economic value
THB / ฿	Thai baht
TVMP	Total value of marginal products
USA	United States of America
USD	United States dollar
VMP^{H}	Value of marginal product of fruit (from bee pollination)
VMP^{F}	Value of marginal product of honey
WTP	Willingness to pay
$\sim \chi^2$	Chi-squared distributed
$\sim N$	Normally distributed

Chapter I

General introduction

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In this chapter, I review the research approach and basic concepts that were applied throughout this dissertation, as follows. Section 1.1 gives a brief overview of the principles of pollination biology, of the importance of this ecosystem function to the reproduction and genetic improvement of plants, and of its contribution to food security. Section 1.2 explains the role of bees as pollinators of wild plants and crops, placing emphasis on the importance of pollinator diversity to agricultural production. I introduce the current global pollinator crisis and the international initiatives that aim at conserving bees and other pollinators in Section 1.3. Section 1.4 presents the state of the art in the economic valuation of pollinating bees and gives a supplementary review on the principles of choice modelling, which I propose for addressing the challenge of capturing non-use values of bees. The motivation and objectives of this thesis are presented in Section 1.5, followed by Section 1.6 with the research questions I pursue to answer. An overview of the sites that I chose for my research and of the data and software that were used is given in Sections 1.7 and 1.8, respectively. Finally, Section 1.9 outlines this dissertation's body of research.

1.1 Pollination: basic principles and ecological significance

Pollination is the process by which pollen grains, containing male gametes, are transferred from the male to the female reproductive organs (i.e., to the stigma in angiosperms or directly to the naked ovule in gymnosperms[2]) of spermatophytes (i.e., seed-producing plants) from the same species, and thus precedes their sexual reproduction (Delaplane and Mayer 2000; Flamini 2012). In angiosperms (i.e., flowering plants) for instance, when a pollen grain gets from the anthers of one flower to the stigmatic surface of the same or another flower, the germination of a pollen tube is induced, after which it grows to penetrate the ovary where it delivers two male gametes, one of which will effectuate ovule fertilization giving rise to a zygote (i.e. a diploid embryonic seed). The other gamete fuses with a female diploid nucleus, the resulting cell of which divides repeatedly forming the triploid endosperm tissue that will provide nutrition to the developing embryo (Willmer 2011). Fertilization is a fundamental stage in the development of seeds (in

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[2] Gymnosperm is a taxonomically loose term for all spermatophytes other than angiosperms. It is etymologically indicative of the "exposed seeds", a defining characteristic of gymnosperms that contrasts with the "hidden seeds" of flowering plants (Willmer 2011).

all seed plants), which store food reserves for the early seedling growth, mainly in the form of carbohydrates, oils and fats, and proteins; thus the importance of seeds as major sources of food for humans and animals (Bewley and Black 1994). Moreover, a unique aspect of angiosperms is the development of fruits as their seeds mature. Botanically, a fruit is the structure containing the seeds, i.e., the pericarp that is formed from the wall of a mature ovary after the flowering stage. However, the flesh in some fruits may partly develop from various floral components, such as the receptacle, sepal and inflorescence. Fruit formation in plants has evolved as a seed dispersal strategy, insofar the energy and nutrition contents of fleshy fruits may attract surrounding animals that may disseminate the seeds after ingesting and defecating them, or during the process of eating the fruit pulp (Nath et al. 2014). Moreover, in many plant species the growth of fruit flesh is associated with the activity of the developing seeds and in some cases (e.g., in melon and other Cucurbitaceae) fruit size is positively correlated with seed number (Bewley and Black 1994).

Technically, pollination does not include pollen germination and ovule fertilization (Willmer 2011), yet describing these processes contextualizes pollen transfer, thus highlighting its importance as a regulating ecosystem function.

Plants, lacking motility themselves, also rely on moving agents to increase the chances of immotile male gametes reaching equally immotile female gametes. Such agents may be abiotic, i.e., wind or water currents, and/or biotic i.e., mobile animals (in particular flying ones), commonly referred to as pollinators (Willmer 2011). Most flowering plants are pollinated by animals[3] (i.e., zoophilous), especially by insects (i.e., entomophilous), with a minority of species relying on abiotic pollen vectors, mainly wind (anemophilous) (Ollerton et al. 2011). The opposite is true for gymnosperms, which are mostly anemophilous, and among which only Gnetales (the closest living relatives of angiosperms) and Cycadales present entomophily (Crepet 1974; Labandeiraet al. 2007; Nepi et al. 2009; Willmer 2011).

Pollinators are especially important to ensure cross-pollination, which refers to the transfer of pollen between different plants from the same species (i.e., conspecific pollen transfer). Cross-pollination is an evolutionary strategy of angiosperms to promote outbreeding that is more commonly found among perennials than among short-lived annual plants. Self-fertilizing[4] plants are more vulnerable to inbreeding depression, which often manifests in smaller seed size, lower seed count, slower germination and slower growth. Therefore, almost all habitually self-pollinating plants have also the ability to cross-pollinate.

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[3] Ollerton et al. (2011) estimated the proportion of animal-pollinated species of flowering plants (i.e., angiosperms) to increase from a mean of 78% in temperate-zone communities to 94% in tropical communities. Correcting for the latitudinal diversity, this proportion results in 88% of the estimated global species-level diversity of flowering plants.

[4] Darwin (1876) proposed that plants adapt to self-pollination as a (backup) reproductive assurance when outcrossing fails. Kalisz et al. (2004) corroborate that hypothesis providing empirical evidence for an increase of self-fertilization rates in *Collinsia verna* in periods of infrequent pollinator visits.

Furthermore, the visit of pollinators to facultative cross-pollinating plants often results in improved seed or fruit quantity and/or quality (or uniformity) (Willmer 2011). For instance coffee (*Coffea arabica*), which is capable of self-pollination, will produce significantly higher yields with lower peaberry[5] incidence in the presence of abundant native or introduced bees (Roubik 2002; Ricketts et al. 2004).

Pollinators can be important contributors to the provision of calories in the human diet, are critical to the production of a considerable portion of its vitamins and minerals and support the production of many plant-derived medicines. Thus, pollinators are essential to food security and sound nutrition, and to the biodiversity that enriches our diet and medicine (El-Berry et al. 1974; FAO 1995; Canter et al. 2005; Ellis et al. 2015; Eilers et al. 2011). Animal-mediated pollination, in particular by bees (i.e., melittophily), is important for the sexual reproduction of 87 of the 124 leading global crops[6] used directly by humans, and of the majority of wild plants (Klein et al. 2007).

1.2 Pollination by bees

To take advantage of the behavioral flexibility of pollinators, plants reward flower visitors (mainly) with pollen and nectar. Bees (Hymenoptera: Apoidea: Anthophila), which (with the exception of a few species) rely exclusively on floral resources for both larval and adult nutrition, have a higher flower visitation rate, and vary considerably more in their interaction with flowers, than any other taxon. Furthermore, their sizes, hairy bodies, variably long tongues, learning capabilities and floral constancy (among other adaptations that may be species-specific), equip bees especially well to visit flowers efficiently and make them effective pollinators (Willmer 2011). Bees are thus usually the most effective pollinators of the various crops and wild plants that are not wind-pollinated (Free 1993; Batra 1995; Delaplane and Mayer 2000).

The benefits humans draw from pollination make this ecosystem function an essential ecosystem service (de Groot et al. 2002; Pascual et al. 2010). Crop pollination is an ecosystem service in that wild pollinators, particularly wild bees, supply a valuable input to agricultural production. It may also be considered a farm management tool when domesticated bees are kept, purchased or rented by farmers in many countries to either supplement the local pollinator fauna or to restore the decline of its services (Richards 1993; Heard 1999; Ricketts et al. 2008; Gallai et al. 2009). In this regard, it should be noted that the contribution of wild bees to crop pollination has generally been underestimated: the European honeybee (*Apis mellifera* L.) has often been credited with pollination services actually provided by non-Apis bee species and other wild pollinators (Parker et al. 1987). In fact, *A. mellifera* is currently the single

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[5] In coffee, peaberries are small misshapen seeds resulting from one of the two ovules failing to develop (Ricketts et al 2004).

[6] Crop plants constitute < 0.1% of all flowering plant species (Ollerton et al. 2011), which represent about 80% of all known living plant species on earth (Cronquist, et al., 2016).

most commonly used species in managed pollination services worldwide and is particularly deployed within highly productive crops that flower intensively for brief periods (Slaa et al. 2006; Potts et al. 2010; Magrach et al. 2017; see also Section 2.1.1).

On the other hand, there are an estimated over 40,000 bee species globally, of which more than 25,000 have so far been fully described. The majority of all bee species[7] are so-called solitary bees (in the sense that each female, being fertile, makes and stocks her own nest), which encompass many genera (e.g., *Osmia* and *Megachile*) in most of the bee families. Solitary bees belong to one of six groups of bees that Willmer (2011) classifies as melittophily subsyndromes[8], the other five being carpenter bees (i.e., mainly Xylocopini), euglossine bees (Euglossini), bumblebees (Bombini), stingless bees (Meliponini) and honeybees (Apini). Within the Apidae family, all species in the latter three groups (tribes) are eusocial bees[9], a condition only present in a minority of bee species, while only degrees of sociality are displayed by some members of the Xylocopini and Euglossini tribes (Willmer 2011).

Solitary bees tend to form specialized mutualisms with particular plant species and are therefore said to be more effective[10] pollinators (per visit) than social bees (Welsford and Johnson 2012). The latter have more generalized flower-visiting habits as they cover a longer sequence of forage plants to feed their colonies, which compared to solitary bees, are active for much longer periods over the year. Social bees may thus also offer flower-visiting behaviours that favour pollination: in addition to being generalists, they tend to cover substantially larger foraging ranges than solitary bees, communicate foraging site locations among individuals (thus saving individual efforts of searching for forage) and can visit flowers even in unpropitious weather conditions, among other features (Willmer 2011). Klein et al. (2003) showed that, for 24 agroforestry systems in Indonesia, bee diversity rather than abundance explains the variability in fruit set of highland coffee (*Coffea arabica*). In additional experiments, they showed that the (per visit) pollination effectiveness of solitary bee species was indeed significantly higher (leading to higher fruit set) than that of social bee species. Nevertheless, the collective role of a species-rich bee community was ultimately important for the successful pollination of highland coffee, i.e., social bees matter, due to their

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[7] According to Batra (1984) more than 85% of all bee species are solitary.

[8] A pollination syndrome is a grouping of flower species (often from very different taxonomic groups) that are classified according to traits they have developed in convergent evolution in order to attract a particular pollinator (Willmer 2011). In this regard, a subsyndrome of melittophily refers here to a group of bees, for the visits of which flowers have specialized.

[9] Eusociality is the highest form of social organization in animals, involving different morphological types (castes) or age groups that assume different social functions. A eusocial bee colony usually consists of one fertile queen that engages in egg production, one or more fertile males that fertilize eggs, and one or more classes of sterile workers that exclusively engage in food gathering, nest maintenance, colony defence and cooperative brood care (Strickberger 2005).

[10] For consistency purposes, I use the term “effective” throughout this section to refer to the “per visit effectiveness” of a pollinator species, as defined by Willmer (2011). Welsford and Johnson (2012) for instance use “effectiveness” to describe what Klein et al. (2003) call “efficiency”, when referring to the “fruit set after a single visit of a specific bee species to a single virginal previously bagged coffee flower.”

high visit frequency, and solitary bees due to their pollination effectiveness (ibid). Brittain et al. (2013) arrived to a similar conclusion after conducting field experiments in 25 Californian almond (*Prunus dulcis*) orchards that were visited by both *Osmia lignaria* and *A. mellifera* bees. One may thus not categorically say that one type of bee is a better pollinator than the other. After all, a visitor's pollination effectiveness involves both measures: *i)* per visit effectiveness (as in e.g., seed set, or preferably the amount of conspecific pollen deposited on the stigma); and *ii)* visit frequency. At any rate, the concept of pollination syndromes represents an oversimplification: predicting a plant's pollinator by just cataloguing its key floral traits is constrained by external ecological factors. In reality, in a diurnally and seasonally changing environment, most flowers are generalists rather than specialists and can be potentially visited by a range of pollinators, which in turn may vary in their pollination effectiveness through time and space (Willmer 2011).

The last several decades have seen an increasing interest in the development of management practices for applied crop pollination with non-honeybee pollinators (Slaa et al. 2006), such as bumblebees [e.g., *Bombus impatiens* for greenhouse tomatoes (*Lycopersicon esculentum*) (as described by Morandin et al. 2001 for Ontario, Canada)], solitary bees [e.g., *O. lignaria* in almond (*Prunus amygdalus*), apple (*Malus domestica*) and pear (*Pyrus communis*) orchards in Utah, USA (Bosch and Kemp 2000) and *Nomia melanderi* for alfalfa (*Medicago sativa*) in the Touchet Valley, USA (Cane 2008)] and stingless bees[11] (Heard 1999; Slaa et al. 2006). The importance of finding alternatives to *A. mellifera* crop pollination has increased as, in recent years, Europe and the USA have experienced severe honeybee stock declines, which have drawn attention to the vulnerability of this species to diseases (e.g., viruses), pathogens (especially the ectoparasitic mite *Varroa destructor*) and other environmental stressors that may act synergistically. The increasing dependence of agricultural crops (and possibly of wild plants) on the pollination services from *A. mellifera* alone is thus worrisome (Potts et al. 2010).

1.3 Global pollination crisis: drivers and pollinator conservation initiatives

In their study, Aizen and Harder (2009) claim that despite increasing evidence of regional and local declines in pollinators, the global population of managed European honeybees has increased by ~45% during the last half century. Nevertheless, the same study reveals that the share of agricultural output that requires animal pollination has increased by more than 300% (in response to increasing demand per capita) during the same period, which is likely to exceed the global honeybee pollination capacity. As a consequence of such an increase of pollinator dependency in global agriculture, the relative reduction in

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[11] For instance, the Sumatran *Trigona minangkaba* for greenhouse strawberries (*Fragaria chiloensis* × *ananassa*) in Shimane, Japan (Kakutani et al. 1993), *Melipona subnitida* for greenhouse sweet peppers (*Capsicum annuum*) in Fortaleza, Brazil (de Oliveira Cruz et al. 2005) and *Tetragonula laeviceps* in Chanthaburi, Thailand (Chuttong et al. 2015; see also Sections 2.1.1 and 2.1.2).

pollination services may offset the expected benefits from agricultural intensification (Deguines et al. 2014). Moreover, Aizen et al. (2009) propose that pollination shortages will intensify demand for agricultural land, this trend currently being more pronounced in the two-thirds of global cultivated land corresponding to the so called developing world.

Landscape alteration is one of five main global change pressures to which pollinator declines have been attributed, the other four being agricultural intensification, species invasions, the spread of pathogens and climate change (González-Varó et al. 2013; Goulson et al. 2015). Habitat loss (or more commonly habitat fragmentation) is thought to be the most important contributor to reductions in bee diversity and abundance (Potts et al. 2010). A quantitative meta-analysis that Winfree et al. (2009) conducted over 54 studies on the effects of anthropogenic disturbances to bee abundance and/or richness significantly supports such hypothesis. Lately, the decline in habitat availability has been sharpest in the tropics, where species richness is greater and, according to a meta-analysis by Vamosi et al. (2006), the risk of pollinator and plant diversity losses is higher. Nevertheless, like other drivers that affect biodiversity and pollinator populations, land-use change rarely acts in isolation (Schweiger et al. 2010; Potts et al. 2010). Agricultural intensification, thought to be a major driver of pollinator declines, generally occurs in the most altered landscapes, which can make it difficult to disentangle its effect on pollinator communities from that of habitat degradation. Intensive agriculture is characterized by monocultures with increasing farm sizes, which in combination with an intensive application of herbicides, reduce the diversity and spatiotemporal availability of foraging resources and nesting sites for bees (Willmer 2011; Potts et al. 2010; González-Varó et al. 2013). Additionally, intensive pesticide use has a well-documented effect on pollinators, especially on wild ones. Poisoning of honeybees can for instance result in direct mortality, abnormal communication dances, inability to fly and displacement of queens (Johansen et al. 1983; Kearns et al. 1998; Potts et al. 2010; Willmer 2011). Recently, a relatively new class of globally used insecticides called neonicotinoids (chemically similar to nicotine), have become an important subject for public debate as they have been directly implicated in the declines of bee populations including honeybees, bumble bees and solitary bees. Neonicotinoids compromise the central nervous system of pest-insects, paralyzing and killing them. At sub-lethal levels[12] however, molecules of these neuro-toxic insecticides influence the cognitive abilities of non-target insects such as bees, weakening their performance and ultimately impacting the viability of their colonies (Blacquière et al. 2012; Sánchez-Bayo et al. 2016). Furthermore, scientific evidence points at synergistic effects between neonicotinoids and pathogen infection (Alaux et al. 2010; González-Varó et al. 2013) and at a significantly higher propensity of colony failure when bees

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[12] Neonicotinoid insecticides can for instance be translocated into the pollen and nectar (e.g., from seeds that have been coated to systemically protect the plant) that are collected by foraging bees (Girolami et al. 2009; Cresswell 2011).

are exposed to neonicotinoids in combination with another commonly applied pesticide (i.e. λ-cyhalothrin) (Gill et al. 2012).

Habitat fragmentation and agricultural intensification are also likely to partly explain the recently reported substantial losses of flying insect biomass (i.e., more than 75%) in protected nature areas of Germany (Hallmann, et al. 2017). The total insect biomass was measured with standardized protocols over 27 years (between 1989 and 2016) in natural reserves of limited size, most of which (94%) are enclosed by agricultural fields and thus characterize a typical West-European fragmented landscape. These declines are especially alarming considering that they were recorded in areas that have been designated for the preservation of ecosystem functions and biodiversity (ibid).

For nearly three decades now, declines of wild and managed pollinators have been reported in every continent (except for Antarctica) in at least one region and/or country (FAO 2008; Willmer 2011). The prospect of a global pollination crisis (Kluser and Peduzzi 2007), although controversial in the past (Ghazoul 2005a, b; Steffan-Dewenter et al. 2005), has raised concerns among the scientific community and national and international policy-makers (Dias et al. 1999; Kremen et al. 2002; Millennium Ecosystem Assesment 2005; FAO 2008; Byrne and Fitzpatrick 2009; Potts et al. 2010; Vanbergen et al. 2013; Goulson et al. 2015; IPBES 2016). The São Paulo Declaration on Pollinators (Dias et al. 1999) made such concerns explicit and laid the conceptual foundations to develop the International Initiative for the Conservation and Sustainable Use of Pollinators (also known as the International Pollinator Initiative - IPI) that was officially established in May 2000 at the 5th Conference of the Parties of the Convention on Biological Diversity in Nairobi, Kenya (Byrne and Fitzpatrick 2009; Willmer 2011). The initiative's plan of action (IPI-POA) offers guidelines for the improvement and/or development of policies and practices that enhance the conservation of pollinators and that restore their habitats, which can be summarized in four elements: *i)* assesment of bees and their services; *ii)* adaptive management; *iii)* building capacity; and *iv)* getting bees into policy-mainstreaming. The globally coordinated implementation of such guidelenes aims at reaching four main objectives, namely: *a)* monitoring pollinator declines, their causes and impacts on pollination services; *b)* addressing the lack of taxonomic information on pollinators; *c)* assessing the economic value of pollination and the economic impact of pollinator declines; and *d)* promoting the conservation, restoration and sustainable use of pollinator diversity in agriculture and related ecosystems (Byrne and Fitzpatrick 2009; CBD Secretariat 2017). Since its formation, several other initiatives have joined IPI both at the regional level (e.g., the African Pollinator Initiative, the North American Pollinator Protection Campaign and the Oceania Pollinator Initiative) and at the national level (e.g., the Brazilian and Colombian Pollinator Initiatives and the UK Insect Pollinators Initiative). Other organizations that have the common goal of conserving pollinators include the Xerces Society, the STEP (i.e., Status and Trends in European Pollinators) project and COLOSS (i.e., Prevention of honey bee Colony LOSSes) (FAO 2017; COLOSS 2017; STEP 2017; Xerces Society 2017).

Neonicotinoids have also entered the policy arena in the European Union, where on the Commission's (EC) request, the European Food Safety Authority (EFSA) assessed the risks associated with the application of three neonicotinoid varieties, namely imidacloprid (i.e., the most used insecticide worldwide), clothianidin and thiamethoxam. After the EFSA released its conclusions confirming their toxicity (European Food Safety Authority 2013a, b, c), the EC issued in 2013, on precautionary principle, a two-year moratorium on these chemicals. Agrochemical companies have nevertheless disputed the laboratory studies from which such conclusions were drawn, arguing that they are not realistic as they did not replicate the results of field trials, where a negative effect could neither be established on honeybees [see reviews by Blacquière et al. (2012) and by Carreck and Ratnieks (2014), but cf. meta-analysis by Cresswell (2011)] nor on wild bee species (Carreck and Ratnieks 2014; Schmuck and Lewis 2016). Since the moratorium expired in December 2015, the EFSA has updated its assessments based on new evaluations, including a recently released 2-year-long study (Woodcock et al. 2017) that was funded by Syngenta Ltd. and Bayer CropScience, which provides a country-specific verdict on the effect of two neonicotinoids (clothianidin and thiamethoxam) on honeybees and wild bees. According to Woodcock et al. (2017), domesticated honeybees were harmed by these substances in the United Kingdom and Hungary, while in Germany they showed no treatment effect. Wild bees (i.e., *Bombus terrestris* and the solitary bee *Osmia bicornis*) nevertheless had a less differentiated response between countries and showed a depressed reproductive potential that was associated with the neonicotinoid residues found in their nests (ibid). These results corroborate previous findings regarding the harmful effect of neonicotinoids to wild bees [e.g., from a field study by Rundlöf et al. (2015) and from field-realistic laboratory trials by Laycock et al. (2012), by Whitehorn et al. (2012) and by Sandrock et al. (2014)] and to honeybees [e.g., from field realistic experiments by Henry et al. (2012)]. Following two separate consultations with EU Member State pesticide experts, the EFSA has concluded in a series of recent reports (European Food Safety Authority 2018a, b, c) that the risk posed by neonicotinoids to bees (i.e., managed: *A. mellifera* and bumblebees, and wild: bumblebees and *Osmia bicornis* as a representative species for wild solitary bees) is confirmed. According to the EFSA (2018d), their conclusions are to be assessed by risk managers from the EC and Member States for potential revisions of the current restrictions on neonicotinoids, which in spite of Bayer's maintained objections, are likely to become tighter (Stokstad 2018).

1.4 Economic valuation of pollinating bees

According to the TEV classification by Pascual et al. 2010, the economic value of (conserving or losing) pollinating bees can be classified into the direct and indirect *use value* (actual or future), and the *non-use value* categories of their total economic value (TEV). Direct use values of pollinating bees are derived from their hive products (i.e., honey, pollen, royal jelly, wax and/or propolis) and less commonly (depending on the bee species) from their venom. The ecosystem service of pollinating crops and wild plants is classified under the indirect use values of bees and is likely to contribute the most to their TEV. Among the non-use values of pollinating bees, one can classify the value perceived for their mere existence and the cultural and religious significance that human societies assign to them. Other non-use values include the philantropic satisfaction of having other humans derive benefits from bees in the present or the future (ibid).

Moreover, the ecosystem services provided to agriculture by wild bees have the characteristic of public goods. Given the case of becoming insufficient nevertheless, crop pollination can be supplemented or reinstated with managed bees and thus acquire characteristics of a private good and/or of a positive externality.

1.4.1 State of the art

The appeal for methods to estimate the value of ecosystem services has increased since, in the 1970s, this concept started developing around the usefulness of ecosystem functions to humans, giving rise to the adoption of utilitarian arguments for biodiversity conservation (Westman 1977; Ehrlich and Ehrlich 1981; Daily 1997; de Groot et al. 2002; Gómez-Baggethun et al. 2010).

After the urgency of assessing the economic impact of pollinator declines was expressed with the establishment of the IPI (see Section 1.3), the FAO published a review of methods for the economic valuation of ecosystem services (Mburu et al. 2006). Among the approaches listed by Mburu et al. (2006), they deem three ways suitable for the valuation of pollination services, namely *1)* the market price method, *2)* the production function approaches and *3)* the cost-based methods (preventive expenditure, damage costs avoided and replacement costs).

1.4.1.1 Market prices based valuation

The first approach consists in observing market prices for commercial pollination services (i.e., managed bees) at different provision levels (i.e., beehive quantities and strengths), and then obtaining the aggregated farmers' willingness to hire such services (i.e., the demand function) and the aggragated costs of commercial beekeepers to supply them, in order to calculate the consumer surplus (CS) and the producer surplus (PS) resulting from commercial pollination. This method could for instance be applied to the case of pollination services hired by Californian almond orchardists (see Sections 2.1.1 and 2.1.2), provided that the transaction costs would be negligible and that the relevant agricultural markets would be undistorted by subsidies.

1.4.1.2 Production function approaches

In the literature, one of the earliest attempts to estimate the value of pollination services concerns the second approach and dates back to the first half of the 20th century, when Butler (1943) assumed that honeybees were responsible for half of Britain's commercial fruit crop production and as a result valued their pollination at £4 million per annum. He juxtaposed this value with the £1.287 million worth of honey yearly produced by the 429 thousand honeybee colonies available to Britain's agriculture at the time, and thus estimated their total contribution to the nation's economy at £5.287 million per annum (i.e., £12.32 per colony and year). These estimates suggest that, at the time of Butler's publication, the contribution of honeybees to Britain's agriculture would have been worth 3.1 times the value of the honey they produced. Carreck and Williams (1998) re-estimated the 1943 value of honeybee pollination at £22.06 million, by weighting that year's crop values[13] with factors that represent three categories of insect pollination dependency, namely 0.1 (low), 0.5 (medium) and 0.9 (high), and further attributing 80% of insect pollination to domesticated honeybees. Accordingly, they estimated the total value of honeybees in 1996 at £153.6 million (£137.8 million thereof corresponding to crop pollination). The proportion to which the value of honeybee pollination stands relative to the value of honey in the UK would thus be 8.7 in 1996 and (if corrected ex post) 17.1 for 1943.[14] These estimates can give a first impression of the importance of bee pollination services to agriculture by comparing their *indirect* use value with the *direct* use value of beehive products, thus offering a straightforward economic argument for their conservation.

Butler (1943) and Carreck and Williams (1998), like other studies [e.g., Morse and Calderone (2000) and Losey and Vaughan (2006)], also convey in their valuation approach the idea that bee pollination is a crop production input, the value of which can be equated with that of the production reductions they ascribe to a complete loss of pollinators, as assumed with the application of pollination dependence ratios. In other words, if $F = f(B)$ expresses the quantity of fruit produced as a function of the number of pollinating bee colonies B, it follows (according to this approach) that their pollination related value equals their contribution to crop yields multiplied by the crop price P, as given by $V(B) = P * \int_0^B (\partial F/\partial B)\, dB$. Other studies have in contrast implicitly applied a 100 percent dependence ratio by assigning the total value of entomophilous crops to pollinators [e.g., Levin (1983) and (Costanza et al. 1997)]. At any rate, simply multiplying the changes in crop output with farm-gate prices (i.e., calculating changes in revenues) gives an approximation to the economic value of pollinators that can be problematic in view of following caveats: *i)* it is based on revenue considerations that neglect the costs that farmers may save through short term adaptations (e.g., by hiring less labour for the reduced harvest), thus potentially leading to an overestimation of pollination-deficit related production losses; *ii)* it assumes that changes in the

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[13] Carreck and Williams (1998) re-estimated the 1943 crop values using that year's relevant agricultural statistic reports from the UK Ministry of Agriculture, the Department of Agriculture for Scotland and the Government of Northern Ireland.

[14] The corresponding proportion was estimated at 135 (i.e., USD 18.9 billion worth of pollination vs. USD 140 million worth of honey and beeswax) for the USA in 1981 (Levin 1983), while a more recent bulletin published by FAO reported this factor to range between 30 and 50 for Western Europe and to be 100 for Africa (Bradbear 2009).

availability of pollination services do not affect the market, which would only be sensible if such changes were only relevant at the local level, or if a crop yield reduction (increase) would not lead to substantial increases (decreases) in crop prices; *iii)* it does not give a complete picture of the changes in social welfare (ΔW) as it neglects the demand function and thus the consumer surplus (see area III in Figure 1.1); *iv)* the dependence ratios are applied under the assumption that the reference yields (i.e., from which changes in crop-pollination are considered) were realized at the optimum crop pollination level (i.e., where the production function's first-order condition with respect to pollination equals zero): this is evidently often not the case (see Section 1.3) and using reference yields that are below the potential pollination optimum would lead to an underestimation of the contribution of pollinators; *v)* dependence ratios taken from the literature do not reflect the variability in cultivars, in pollinator density and composition, and need not necessarily fit the agronomic conditions that are found in practice (Melathopoulos et al. 2015); *vi)* value estimates that are solely based on market prices may neglect the non-use value components of an ecosystem service's TEV (cf. Pascual et al. 2010).

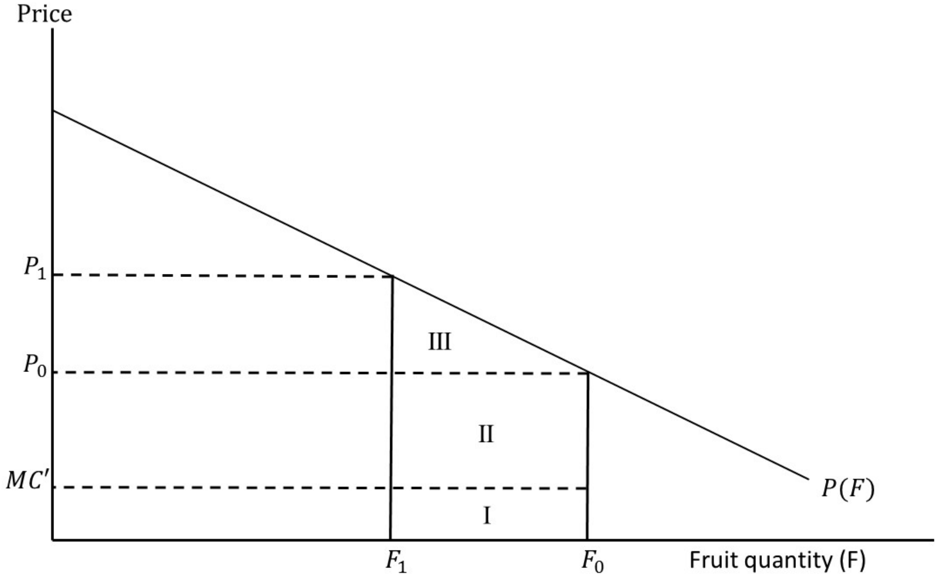

Figure 1.1 Estimating the value of pollinator declines by solely equating them with changes in crop revenues (i.e., area $\boldsymbol{I+II}$) may *i)* lead to an overestimation corresponding to area $\boldsymbol{I}$, as $\boldsymbol{MC'}$ (i.e., the marginal costs that farmers may save when crop output declines from $\boldsymbol{F_0}$ to $\boldsymbol{F_1}$) are neglected and *ii)* neglect the demand function $\boldsymbol{P(F)}$ and thus the consumer surplus corresponding to area $\boldsymbol{III}$.

Evidently, applying pollination dependence ratios to correctly estimate the effects on social welfare from changes in the provision of pollinattion services requires full information on the demand and supply of the crops in question. Southwick and Southwick (1992) constructed demand functions (with long term time series on US agricultural outputs and prices, and household incomes) for an array of agricultural commodities and then calculated the effect that changes in pollination services (and thus yields, according to different pollination dependence ratios) had on consumer surplus *CS* (see also Figure 1.2), as given by:

$$\Delta CS = (P_1F_1 - P_0F_0) + \int_{F_1}^{F_0} [P(F)]\, dF \ , \tag{1.1}$$

where $P(F)$ is the demand function (i.e. prices as a function of fruit quantities), F_1 and P_1 are the fruit quantity and price without insect-mediated pollination, and F_0 and P_0 are the fruit quantity and price with insect-mediated pollination. The first term of the Equation (1.1) is the difference between farmer revenues with and without insect pollination. The second term is the aggregated marginal willingness to pay of consumers (WTP) for the agricultural products that would have been sold at the lower price P_0 (due to optimal pollination levels) and are instead sold at higher prices (up to P_1) after pollinator populations start declining (adding both terms yields areas I and II in Figure 1.2). Gallai et al. (2009) also follow this approach to estimate the worldwide economic value of crop pollination, yet use the dependence ratios offered by Klein et al. (2007) and make assumptions regarding the value and shape of $P(F)$, due to the difficulty of estimating elasticities for each crop at a global scale. By further assuming a long-run (perfectly elastic) aggregate supply curve (i.e., ΔPS=0 and $\Delta CS = \Delta W = I + II$), both studies (Southwick and Southwick 1992; Gallai et al. 2009) address caveats *i)* through *iii)*, yet fail to address caveats *iv)* through *vi)*.

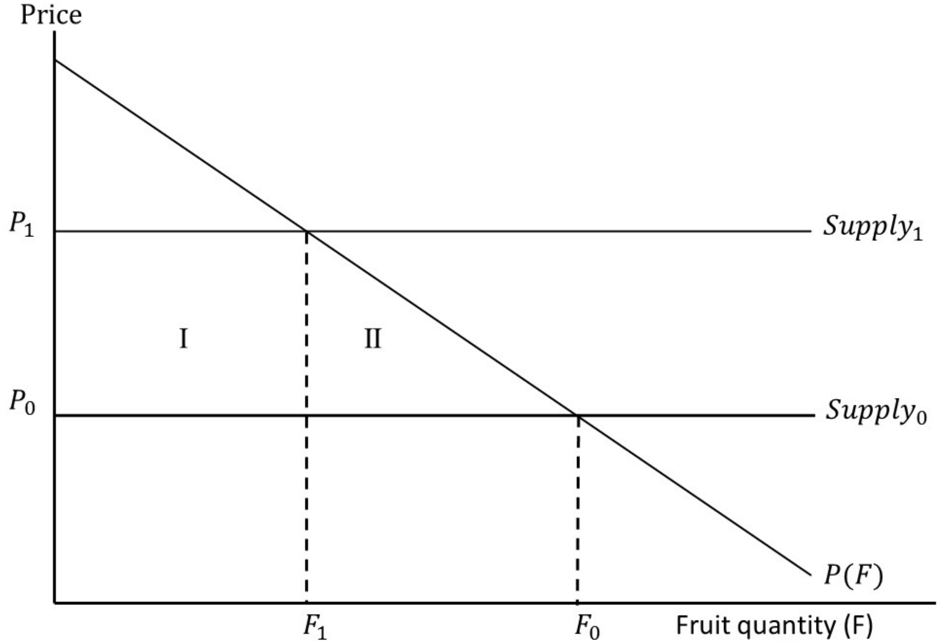

Figure 1.2. Economic valuation of pollinator declines based on consumer preferences (basic approach): crop yield declines ($\boldsymbol{F_0} \rightarrow \boldsymbol{F_1}$), price response ($\boldsymbol{P_0} \rightarrow \boldsymbol{P_1}$) and corresponding consumer surplus loss ($\Delta\boldsymbol{CS} = \boldsymbol{I} + \boldsymbol{II}$) are attributed to a deficient crop pollination. Modified from Southwick and Southwick (1992).

Neglecting important components of the TEV of pollinators [i.e., caveat *vi)*] could lead to a substantial underestimation of ΔCS, thus adding uncertainty to the overall welfare effects. Furthermore, such uncertainty increases with the proportion of crop yields that depends on insect-mediated pollination (Mburu et al. 2006).

Crop pollination is only one of the positive externalities derived from bees, which among others comprise the pollination of wild plants (also an indirect use value) and the existence value associated with specific bee species and/or with bee diversity (i.e. non-use values). If one additionally considers that crop pollination is mostly dominated by a few (often unthreatened) bee species (see Section 1.2), its economic value becomes an argument of less weight for the conservation of pollinator communities. Conservation efforts that are solely guided by this argument may therefore be of limited impact, i.e., they may result in a sub-optimal allocation of bee colonies (see Figure 1.3) and/or may only target a subset of unthreatened bee species (Kleijn et al. 2015; Melathopoulos et al. 2015).

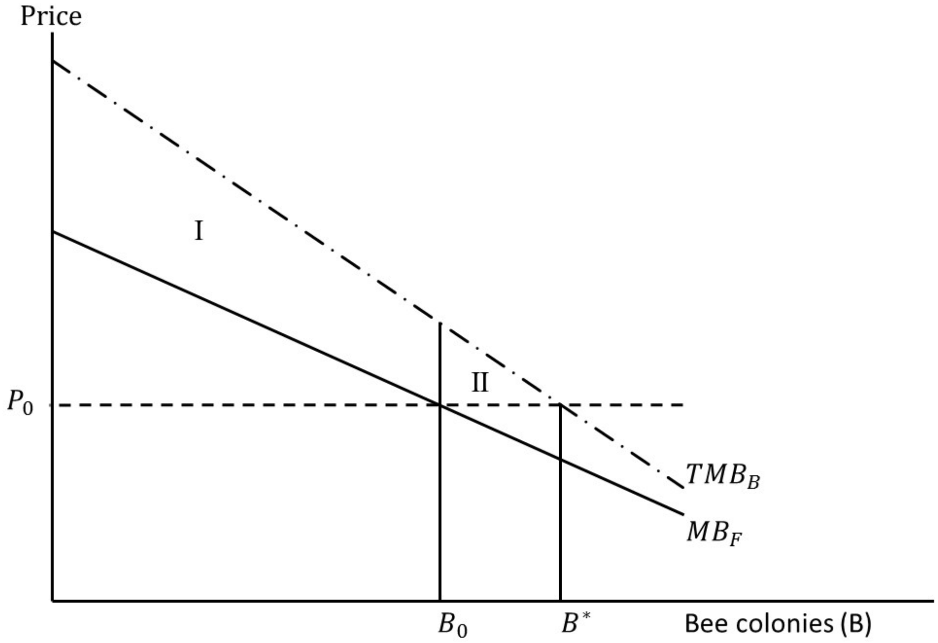

Figure 1.3 The total marginal benefits from bees ($\boldsymbol{TMB_B}$) are the consumers' aggregated willingness to pay for the marginal non-use benefits ($\boldsymbol{MB_{NU}}$, e.g., existence value, not explicitly shown in the diagram) that bees generate, and for the fruits $\boldsymbol{F}$ [$\boldsymbol{MB_F = P(F)}$, where $\boldsymbol{P}$ is the fruit price] resulting from their pollination; thus, $\boldsymbol{TMB_B = MB_{NU} + MB_F}$. At a given fruit price $\boldsymbol{P_0}$, farmers' incentive to conserve bees is limited to the contribution that $\boldsymbol{B_0}$ colonies can make to the corresponding fruit production, disregarding the additional utility (additional externality) that, for instance, may be derived from their existence. Thus, when only use values are considered (e.g., crop pollination), the change in social welfare arising from a loss of $\boldsymbol{B_0}$ colonies is underestimated by the value corresponding to area $\boldsymbol{I}$. Area $\boldsymbol{II}$ represents the net benefits potentially forgone if conservation efforts allocate $\boldsymbol{B_0}$ colonies instead of the optimum level $\boldsymbol{B^*}$.

1.4.1.3 Cost based methods

An example of the third approach that Mburu et al. (2006) list in their review is given by Allsopp et al. (2008), who ascribed value to the wild and managed pollination services in the Western Cape deciduous fruit industry of South Africa, by calculating the costs of replacing them with pollen dusting and hand pollination [using adjusted insect and managed pollinator dependence ratios and thus also being prone to caveats *iv)* and *v)* above]. This valuation approach is manifestly not based on consumer preferences and thus not appropriate for the measurement of welfare changes. Cost-based methods should be applied when maintaining or replacing the ecosystem service in question (or mitigating the damages arising from its deterioration) is imperative and the mitigation measures considered should be implemented at any cost. Section 3.2 also briefly discusses Allsopp et al. (2008) and provides further examples of efforts to estimate the economic value of pollinators and their services.

1.4.2 Challenges for valuing conservation policies with stated preference methods

The so called stated preference (SP) approach consists of letting (potential) consumers directly state their willingness to pay (or preference) for a public good, as opposed to indirectly revealing it through a behavioral or physical link to a marketable good (i.e., revealed preference approach). It encompasses the discrete choice experiments (DCEs)[15] and the contingent valuation (CV) method, which elicit the preferences of a representative consumer population's sample in a survey context and potentially take into account all TEV components when estimating consumer preferences for ecosystem services. In their review of methods, Mburu et al. (2006) mention stated preference methods as a fourth valuation approach that they consider inadequate for the valuation of pollination services. Mburu et al. (2006) sensibly argue that, in order for CV and DCEs to be reliable valuation tools with regard to pollination services, all respondents would have to fully understand how much pollinators contribute to agricultural production, a condition that is unlikely to be fulfilled.

To overcome unfamiliarity with the service to be valued, the SP approach often involves informing lay respondents about the complexities of the ecosystem function in question before the survey is administered. This has raised concerns about the extent to which the provided information may contribute to preference formation (i.e., *information bias*) and thus lead to distorted value estimates (Spash 2002; Barkmann et al. 2008). Another issue of SP methods has to do with their hypothetical nature, which paradoxically is one of their main advantages over other valuation approaches: confronting respondents with hypothetical goods (or hypothetical combinations of goods) allows elicitng consumers' ex-ante preferences for a good before introducing it to the market or for a policy before implementing it. On the other hand, asking respondents about hypothetical goods may introduce a *hypothetical bias,* which refers to the discrepancy between the expressed preferences and the preferences that would be embedded in actual market transactions (Hausman 2012; Rakotonarivo et al. 2016). These biases are some of the caveats[16] of the SP approach that can compromise the validity (i.e., the extent to which a study measured

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[15] Discrete choice experiments are often mistaken for conjoint analysis, which is just a generic term used to refer to several ways of eliciting preferences that have a limited relationship to utility theory (Louviere et al. 2010; Carson and Louviere 2011).

[16] For an in-depth typology of SP methods related caveats, the interested reader is referred to Bateman et al. (2002).

the intended quantity) and reliability (i.e., a measurement's degree of replicability) of its applications. They are often rooted in following assumptions that SP practitioners make: *i)* that respondents have well formed and stable preferences, which they engage when providing their valuation responses; and *ii)* that such preferences are consistent with standard economic theory (Bateman et al. 2002). These caveats also reflect a general attitude of SP economists, who often take preferences as given (regardless of how irrational they may seem), and frequently neglect the behavioral aspects that may motivate respondents to act erratically (e.g., to behave differently during a survey than how they would act in a real market context), or to display seemingly inconsistent preferences [e.g., the commonly found evidence of large disparities between individuals' maximal WTP to acquire a good and the minimal willingness to accept compenstaion to give it up (Tversky and Kahneman 1991)].

Fortunately, many of the issues mentioned above can be dealt with during the experimental and survey design phases of an SP study (e.g., by correctly identifying and specifying the environmental good and corresponding attributes to be valued, and/or by reducing the task complexity on respondents) (Carlsson 2002; Barkmann et al. 2008). Moreover, with regard to the concerns raised by Mburu et al. (2006), the SP approach can be applied to estimate the value of nature conservation measures and thus to inform the design of conservation policies (Lienhoop et al. 2015), rather than to value complex ecosystem functions per se. Perhaps the most prominent example in this context is the case where, after the 1989 infamous Exxon Valdez disaster occured in open waters of the Prince William Sound in Alaska, Carson et al. (2003) conducted a CV to estimate household WTP for a measure that would prevent another marine oil spill. This study aimed at assessing the liability of the *Exxon Valdez* oil tanker's owners to compensate the general public for the related damages (i.e., for the harm to the local fisheries, recreation and tourism industries and for the degraded and lost services of non-use value character). The anticipation of a compensation (or of a fee, or penalty) may nevertheless induce a *strategic bias* in the value estimates, which results from respondents' preferences having been deliberately misrepresented in order to influence the study's outcome and thus the decision making processes it may guide (Bateman et al. 2002; Venkatachalam 2004; Flachaire and Hollard 2007). To circumvent such bias, Carson et al. (2003) used a double-bounded dichotomous choice elicitation format, as proposed by Hanemann et al. (1991). This solution in turn may drive respondents to adopt heuristic decision rules (e.g., the tendency to *yea-saying*), which accentuate the *starting-point bias* that arises when respondents (directly or indirectly) anchor their WTP to the initial bid introduced by the interviewer (Mitchell and Carson 1989; Bateman et al. 2002; Chien et al. 2005).

As DCEs have overcome some of the limitations of CV (e.g., the proneness to strategic bias) (Bennett and Adamowicz 2001) and, in recent years, computational advances have allowed the estimation of increasingly sophisticated econometric models (Train 2003), this method has gained popularity as a valuation tool for nature conservation policies. For isntance, Hanley et al. (2003) applied a DCE to assess public preferences over the design of a policy aimed at conserving wild geese in Scotland. Other DCE applications dealing with the design and public acceptance of conservation policies include Bauer et al. (2004), Colombo et al. (2005) and Greiner et al. (2014).

Chapters 3 and 4 of this dissertation are an attempt to estimate the value of policies directed at conserving wild bees in agro-forest ecosystems of Northern and Eastern Thailand. We defined these policies as a combination of preventive and corrective measures, following the recommendations of the IPI-POA (Byrne and Fitzpatrick 2009), and the potential effect of such measures on the local population of wild (native) bees. Accordingly, three conservation measures were presented in this study's DCE surveys, which consisted of following strategies to be hypothetically implemented at the village level with the support of extension services:

Preventive measures:

i) "bee-friendly pest control", consisting of an educational program under which farmers would get information on methods (e.g., integrated pest management and spraying during times with low bee activity levels) and products that offer an alternative to conventional use of agro-chemicals, thus reducing the risk of bee poisoning.

ii) "improving native bee habitat", which consists of the provision of native tree seedlings and expert advice to promote the rehabilitation and management of natural habitats in public lands and near cropland, aiming at offering nesting sites and food resources for native bees within agro-forest ecosystems.

Corrective measure:

iii) "adopting on-farm native bee husbandry", which entails the transfer of technical knowledge on how to build bee hives that keep native bee species such as stingless bees or the Asian honeybee (*Apis cerana* F.).

1.4.3 Choice modelling: supplementary theoretical background

The hypotheses postulated in Chapters 3 and 4 were tested modelling DCE data. Therefore, the theory underlying the DCE approach and choice modelling is partly explained in Sections 3.3.1 and 3.3.2 for random parameter logit [also known as mixed logit (ML)] models, and in Sections 4.2.1 and 4.2.2 for generalized mixed logit (GMXL) models. In the following subsections, I supplement such theoretical background on choice modelling with a review concerning the derivation of the logit probabilities and the estimation of the marginal contributions to utilitiy that DCE respondents derive from a choice alternative's attributes; such contributions will henceforth also be referred to as *part-worths*.

1.4.3.1 Derivation and properties of the logit probabilities

The utility that individuals derive from the DCE choice alternatives (Equation 3.1 in chapter 3) is a continuous (latent) variable underlying each observed choice decision (Agresti 2007) and thus each choice probability. In the context of choice analysis nevertheless, the absolute value of such utility is irrelevant, as for choice probabilities only differences in utility matter. This can be expressed as

$$P_{ih} = P[(U_{ih} > U_{ij})] = P[(V_{ih} - V_{ij}) > (\varepsilon_{ij} - \varepsilon_{ih})] \quad \forall\, j \neq h \tag{3.2}$$

(cf. Equation 3.2 in Section 3.3.2), which is the probability P_{ih} of an individual i choosing alternative h over any other alternative j (from a given set of $j = 1, \ldots, h, \ldots, J$ choices), provided that the relative standing of the underlying utilities corresponds to the utility maximization rule[17] $U_{ih} > U_{ij}$. The (unobserved) stochastic term (i.e., random error) ε_i represents the difference between the true utility U_i experienced by the individual and the utility portion that the researcher captures in the systematic component V_i, i.e., the representative utility (Train 2003). From Equation (3.2) it becomes apparent that P_{ih} follows a cumulative distribution, as it is the probability of the random term $\varepsilon_{ij} - \varepsilon_{ih}$ taking smaller values than the measured quantity $V_{ih} - V_{ij}$. Taking into account the actual choice observations (from the DCE survey), P_{ih} can be reexpressed as the J dimensional integral

$$P_{ih} = \int_{\varepsilon} \mathbb{I}\left(V_{ih} - V_{ij} > \varepsilon_{ij} - \varepsilon_{ih}\right) f(\varepsilon_i)\, d\varepsilon_i \quad \forall j \neq h \ ,$$

where $\mathbb{I}(.)$ equals one to indicate when the expression in parentheses is empirically true (and zero otherwise), and $f(\varepsilon_i)$ is the joint density of the random error vector $\varepsilon_i = \langle \varepsilon_{ij}, \ldots, \varepsilon_{iJ} \rangle$ (ibid).

The researcher must specify a functional form for $f(\varepsilon_i)$, which in turn defines the distribution she assumes for ε_i and thus the discrete choice model to be estimated. For instance, probit models are derived from assuming a multivariate normal distribution (ibid). The standard logit model, on the other hand, is obtained from assuming that each ε_i is distributed independently and identically (IID) extreme value type 1 (EV1)[18], which corresponds to following cumulative distribution[19]:

$$F\left(\varepsilon_{ij}\right) = e^{-e^{-\varepsilon_{ij}}}, \ \ \varepsilon_{ij} \sim IID\ EV1(0.57721, \pi^2/6). \tag{1.2}$$

The EV1 distribution's variance is $\pi^2/6$ and has implications over how the scale of utility is normalized (which will be discussed in Section 4.2.2). Its mean is non-zero, which nevertheless is immaterial for choice modelling, considering that only differences in utility matter and that the difference between two

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[17] This rule is based on the assumption that individuals behave rationally, in the sense that they are completely aware of their preferences and that they use all the relevant information at their disposal (e.g., on the probabilities of events, and on potential costs and risks) to assist their choice decisions.

[18] The EV1 distribution has slightly fatter tails than the normal, which allows for a moderately more divergent behavior than the normal (Train 2003).

[19] The underlying probability density function for each unobserved utility portion is $f(\varepsilon_i) = e^{-\varepsilon_{ij}} e^{-e^{-\varepsilon_{ij}}}$(Train 2003).

random variables with equal means has in itself a zero mean (Train 2003). The difference between two EV1 errors $\tilde{\varepsilon}_{ijh} = \varepsilon_{ij} - \varepsilon_{ih}$ with density $g(\tilde{\varepsilon}_{ijh})$ follows a logistic distribution, such that the cumulative distribution of its $J-1$ dimernsional vector corresponds to the binary logit model

$$G\big(\tilde{\varepsilon}_{ijh}\big) = \exp(\tilde{\varepsilon}_{ijh})/(1+\exp(\tilde{\varepsilon}_{ijh})) \ . \qquad (1.3)$$

In accordance to McFadden's (1974) derivation of choice probabilites, one can rewrite Equation (3.2) as $P_{ih} = P[(\varepsilon_{ih} + V_{ih} - V_{ij}) > (\varepsilon_{ij})]$, such that for a given ε_{ih} this cumulative distribution can be evaluated for each ε_{ij} at $\varepsilon_{ih} + V_{ih} - V_{ij}$. Furthermore, assuming IID EV1 errors, one can insert ε_{ij} into Equation (1.2) to obtain the cumulative distribution $P_{ih}|\varepsilon_{ih} = \exp(-\exp\big(-(\varepsilon_{ih} + V_{ih} - V_{ij})\big))\ \forall j \neq h$, which is the product of the *individual* cumulative distributions (ibid). As ε_{ih} is unknown, P_{ih} is the integral of $P_{ih}|\varepsilon_{ih}$ over all values of ε_{ih} weighted by $f(\varepsilon_i)$:

$$P_{ih} = \int \left(\prod_{j\neq h} e^{-e^{-(\varepsilon_{ih}+V_{ih}-V_{ij})}} \right) e^{-\varepsilon_{ih}}\, e^{-e^{-\varepsilon_{ih}}}\, d\varepsilon_{ih} \ .$$

This integral can be algebraically manipulated (for the detailed algebra leading to Equation (1.3) see Train 2003, p. 85ff.) to obtain the multinomial (conditional) logit choice model

$$P_{ih} = \exp(V_{ih}) \big/ \textstyle\sum_{j=1}^{J} \exp(V_{ij})\,, \qquad (1.4)$$

which yields probabilities P_{ij} that add up to one for each individual i, and is the basic closed-form expression (with analytically tractable solution) upon which the open-form models applied in Chapters 3 and 4 of this dissertation are built. The standard logit model [according to Equations (1.3) and (1.4)] implies the independence of irrelevant alternatives (IIA) property, which presupposes that for any two alternatives h and g, the ratio $P_{ih}/P_{ig} = e^{V_{ih}}/e^{V_{ig}}$ is independent from alternatives other than h and g. In other words, IIA preserves the relative odds of choosing h over g, regardless of the availability (or attribute composition) of other alternatives. This property and its restrictive substitution patterns originate from the inherently assumed IID condition of standard logit models, which imposes a zero correlation in the part-worths (Train 2003). IIA is restrictive in that it fails to acknowledge that, for the odds ratio (of a subset of choice alternatives) to remain unchanged in the presence of an added irrelevant alternative (i.e., irrelevant in that it may be a perfect substitute for either alternative in the choice subset), the choice probabilities may have to adjust.

The ML and GMXL models result from partly relaxing the IID condition: the unobserved (stochastic) utility portion is decomposed into two parts, the first one of which contains all the correlation and heteroskedasticity of the part-worth utilities, while the second part remains $\sim$ IID EV1. Based on the assumption that the distributions of both parts are *mixed*, the researcher may assign any distribution (e.g., normal, log-normal, etc.) to the first part, which is introduced as additional stochastic elements through the random part-worths [cf. Equation (3.4) in Chapter 3 and Equations (4.4) and (4.5) in Chapter 4]. Thus, each of the n part-worths of the attributes constituting the choice alternatives may be assigned a variance term, which for a model with an assumed unrestricted[20] covariance matrix will generate $k(k-1)$ covariances. This covariance matrix identifies the interdependence of attributes within and between alternatives, depending on whether the part-worths are specified as generic or alternative-specific (Tain 2003; Hensher et al. 2005). When specifying ML or GMXL models, the goal of the researcher is to estimate the parameters that describe the density of the part-worths (not the part-worths themselves), i.e., the mean and covariance of the distribution she assumed for the additional stochastic elements that resulted from decomposing the unexplained utility portion.

1.4.3.2 Maximum likelihood estimation, hypothesis testing and goodness of fit measures

Representative utility is typically specified as a linear function of its part-woths[21] β, as $V_{ij} = \beta' X_j$, where X_j is a vector of observed variables that describe alternative j. Choice probabilities can nevertheless be expressed as a nonlinear function of the (latent) utility parameters (see Section 1.4.3.1), which is conveniently modeled using the maximum likelihood estimation method (MLE), instead of ordinary least squares (OLS). In this section I briefly describe the traditional MLE, which applies to logit probabilities when they take a closed form (as in Equation 1.4), i.e., that are conditional on homogeneous β. MLE also constitutes the basic estimation principle of the maximum simulated likelihood (MSL) procedure, which applies when choice probabilities are integrals with no closed form (i.e., conditioned on the stochastic elements that are added to specify individual-specific part-worths β_i) that have to be approximated via simulation in order to estimate the moments of the probability density function of β_i (i.e., not the β_i themselves). Sections 3.3.2 and 4.2.2 describe MSL in the ML and GMXL contexts respectively.

An observed DCE choice can be related to the modeled choice probabilities through the following expression:

$$\prod_j (P_{ij})^{y_{ij}} \ ,$$

where $y_{ij} = 1$ if respondent i chose alternative j and zero otherwise, and thus represents the probability of a choice actually made (Train 2003; Agresti 2007). Thereby, the probability of the observed data can be expressed as a function of the part-worths β

$$L(\beta) = \prod_{i=1}^{I} \prod_{j \in J} (P_{ij})^{y_{ij}} \ , \tag{1.5}$$

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[20] The researcher may also place structure on the covariance matrix in order to represent specific sources of non-independence (Train 2003).

[21] All choice models in this dissertation have assumed such linear specification.

where I denotes the total number of respondents i. The product of probabilities may yield very small values that are likely to escape detection by computer packages, an issue that is solved by taking the logarithm of Equation (1.5) as

$$LL(\beta) = \sum_{i=1}^{I} \sum_{j} y_{ij} ln(P_{ij}) \ , \tag{1.6}$$

which according to McFadden (1974) is globally concave for utilitiy specifications that are linear in β. The part-worths are thus estimated by maximizing the log-likelihood function $LL(\beta)$ through an iterative process that involves computing a numerical maximization algorithm (e.g., Newton–Raphson or Fisher scoring) that finds the $\hat{\beta}$ which satisfy the first order condition $\partial LL(\beta)/\partial\beta = 0$ (Train 2003; Hensher et al. 2005). Inserting the logit probabilities (1.4) in Equation (1.6), this condition becomes

$$LL(\beta) = \sum_{i} \sum_{j} (y_{ij} - P_{ij}) x_{ij} = 0 \ . \tag{1.7}$$

Considering that $y_{isj} - P_{isj}$ (i.e., the difference between one or zero, depending on an individual's actual choice, and the probability of that choice) are the *residuals*, Equation (1.7) can be interpreted as follows: MLE finds the $\hat{\beta}$ that set the sample covariance of the residuals with the explanatory variables equal to zero. Equation (1.7) can be rearranged and divided by I to obtain

$$\frac{1}{I} \sum_{i} \sum_{j} y_{ij} x_{ij} = \frac{1}{I} \sum_{i} \sum_{j} P_{ij} x_{ij} \ , \tag{1.8}$$

where the left hand side, denoted $\bar{x}$, is the average of x over the *observed* chosen alternatives, while the right hand side, denoted $\hat{x}$, is the average of x over the *predicted* choices. The maximum likelihood estimates are therefore those $\hat{\beta}$ that reproduce the observed sample average $\bar{x}$ with the predicted average $\hat{x}$ (Train 2003).

As shown by McFadden (1974), the distribution of the estimator $\hat{\beta}$ is asymptotically normal with mean β and covariance matrix

$$\Omega = X'PX$$

(Hensher et al. 2005). The large-sample normality of the ML estimator $\hat{\beta}$ can be used to conduct hypothesis testing for any attribute k's part-worth, such as $H_0\colon \beta_k = \beta_0$, by computing the *Wald*-statistic:

$$z = \frac{\hat{\beta}_k - \beta_0}{\sqrt{v\hat{a}r(\hat{\beta}_k)}} \ .$$

This test statistic can be compared to a critical *Wald*-value (at various levels of confidence) and its interpretation is equivalent to that of the OLS *t*-statistic (Hensher et al. 2005; Agresti 2007; Rodríguez

2007). This procedure was applied to test the hypotheses concerning part-worths and other choice modeling parameters that are postulated in Sections 3.3.3 and 4.2.3 of this dissertation.

The overall level of significance of a logit model can be determined when comparing its log likelihood (at MLE convergence) with that of a nested (restricted) model (i.e., the LL of a model where all or some of the parameters are set equal to zero), by computing a likelihood-ratio chi-squared statistic as follows:

$$-2\log(L_{nested}/L_{estimated}) = -2(LL_{nested} - LL_{estimated}) \sim \chi^2 ,$$

with *d.f.* corresponding to the difference in the number of parameters estimated for the two models (Hensher et al. 2005). Similarly the LL function can be used to compute goodness of fit measures such as the McFadden Pseudo R-squared:

$$Pseudo\ R^2 = 1 - \kappa * \frac{LL_{estimated}}{LL_{nested}} .$$

The adjusted Pseudo R-squared measure is obtained by penalizing the K number of free parameters in the full model through the factor $\kappa = (\sum_i \sum_s J - N)/[(\sum_i \sum_s J - N) - K]$, where $\sum_i \sum_s J$ is the number of J choice alternatives per N choice observations (i.e., over I individuals and $s \in S$ choice sets). The non-adjusted measure is thus obtained with $\kappa = 1$. Note that the Pseudo R-squared is not equivalent to the OLS coefficient of determination. A Pseudo R-squared between 0.3 and 0.4 is approximately analogous to an OLS R^2 between 0.6 and 0.8, and thus represents a decent to good model fit (Hensher et al. 2005).

The likelihood-ratio chi-squared statistics and goodness of fit measures presented in Chapters 3 and 4 were computed using nested models that estimate two constants only, i.e., that reproduce with the estimated choice probabilities each of the three choice alternatives' sample shares. Thus, the degrees of freedom of these tests correspond to K parameters minus two.

1.5 Motivation and objectives

Engaging in this doctoral research was motivated by the work I conducted for my master thesis “An economic analysis of beekeeping and honey hunting as additional income alternatives for the rural poor in the Philippine Cordillera, Luzon”, which as its title suggests, examines the potential for rural development found in the hive products of bees, both wild and managed. Thereby, it became apparent to me that, beyond their direct use benefits, wild bees generate values that are neglected in many cost-benefit calculations.

This doctoral thesis aims at contributing to the efforts of estimating the value of pollinators (specifically of wild bees) in view of their sustained declines in recent decades. Thereby, this work supplements the efforts summoned by the IPI-POA under the objective of “assessing the economic value of pollination and the economic impact of pollinator decline” and thus provides economic arguments that may contribute to “getting bees into policy-mainstreaming” (Byrne and Fitzpatrick 2009, cf. Section 1.3).

One specific objective of this thesis is to offer an analytical framework to describe the economics of farmer-beekeeper interactions under different provision levels of wild bee pollination. This objective is pursued in Chapter 2, where the microeconomic model by Rucker et al. (2012), which describes the determinants of equilibrium wages paid by almond farmers to commercial beekeepers in California, is generalized in order to accommodate a broader spectrum of farmer-beekeeper interactions. Classifying a specific farmer-beekeeper interaction into one of the exemplary cases presented in Section 2.2.2 may, for instance, help structuring the research questions before undertaking an economic valuation of pollinating bees: e.g., knowing the magnitude and direction of the payments between farmers and beekeepers can hint at whether the valuation questions should be framed in a context of corrective or preventive action.

A second specific objective of this dissertation is to capture the non-use values of pollinating bees by estimating a lower bound TEV of conserving them and their services in Northern and Eastern Thailand. This was approached with the DCEs presented in Chapters 3 and 4 (the first applications of this kind to attempt at valuating pollinators) that were conducted with farmers of melittophilous crop species in both locations (i.e., Chiang Mai and Chanthaburi provinces, respectively).

Lastly, the third objective of this study is to compare the preferences (and WTP) of farmers for selected bee conservation features (cf. Section 1.4.2) between both provinces. Such comparison is relevant considering that pollinator declines have reportedly impaired crop yields in Chanthaburi (while no such evidence exists for Chiang Mai) and thus may have shifted the value perceptions (and enhanced the knowledge regarding bee-mediated crop pollination) of the affected farmers. To this end, the part-worth estimates obtained for both locations had to be made comparable, especially in consideration of the difference between the experimental designs[22] that underlied the DCEs conducted in both study sites.

1.6 Research questions

Following research questions guided the attainment of this dissertation's specific objectives:

a) How do agro-ecological, institutional and socio-economic factors determine the interaction between beekeepers and bee-dependent crop farmers, ultimately leading to the different equilibria observed globally, which include the cases of Californian almond growers and of Chiang Mai (Northern Thailand) and Chanthaburi (Eastern Thailand) orchardists?

b) How can stated preference methods be applied, or, more precisely, a discrete choice experiment be conceived, in order to capture non-use value components of wild pollinating bees, while taking into account that the concerned farmers (respondents) are likely to lack the knowledge regarding the quantitative contribution of pollination to their agricultural production?

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[22] The part-worths that resulted from modelling the Chiang Mai dataset were used as priors to obtain a more efficient experimental design for the DCE conducted in Chanthaburi.

c) How do the preferences and WTP of Thai farmers for different bee conservation measures and for changes in the local population of wild bees stand relative to each other?
d) Do experiences of past crop pollination deficits (i.e., that are attributed to wild bee population declines) have an effect on farmers' preferences for wild bee conservation measures and, if so, does such effect translate into farmers preferring corrective over preventive measures (cf. Section 1.4.2)?
e) Are farmers' preferences for the conservation policy attributes heterogeneous and, if so, can such heterogeneity be traced back to idiosyncratic effects on preference?
f) Can a choice model specification separate preference heterogeneity from the potential differences of how each individual scales utility (i.e., how each individual's part-worth covariance matrix is weighted), and if so, can the latter effect be explained by regional differences (i.e., differences between Chiang Mai and Chanthaburi) and/or by differences in the overall understanding of the DCE exercise and/or of the ecological underpinning of crop pollination?

The research questions that are particular to choice modelling are postulated as testable hypotheses in Sections 3.3.3 and 4.2.3.

1.7 Study sites

The data used for the analyses presented in Chapters 3 and 4 were obtained from separate discrete choice experiments in the Thai provinces of Chiang Mai and Chanthaburi, respectively (see Figure 1.4). These regions were selected for presenting contrasting beekeeper-farmer interactions that seemingly pertain to socio-economic and agro-ecologic differences, the latter of which may further affect the provision of crop pollination services from wild bees. On the one hand, beekeepers in Chiang Mai tend to pay for the right of placing their bee hives on the land of orchardists; there, surrounding natural habitats seem to sufficiently supply the pollination of crops [especially of longan (*Dimocarpus longan*)], which in turn are a rich nectar source for the large local honey industry. On the other hand, anecdotal evidence points at a deficient natural crop pollination in Chanthaburi, which in the past drove orchardists [especially rambutan (*Nephelium lappaceum*) farmers] to rent hives from beekeepers, a corrective measure that more recently developed into farmers adopting meliponiculture (i.e., the occupation of keeping and managing native stingless bee species). The phenomenon in Chiang Mai has been classified under the particular case of beekeeping-farmer interactions presented in Section 2.2.2.3, while Sections 2.2.2.2 and 2.2.2.7 describe the phenomena that have been reported for Chanthaburi. Additionally, the socio-demographic and agronomic aspects that were considered of relevance to the analyses conducted for Chiang Mai and Chanthaburi are discussed in Sections 3.3.4 and 4.2.4, respectively, while the corresponding descriptive statistics are compared in Section 4.3.1.

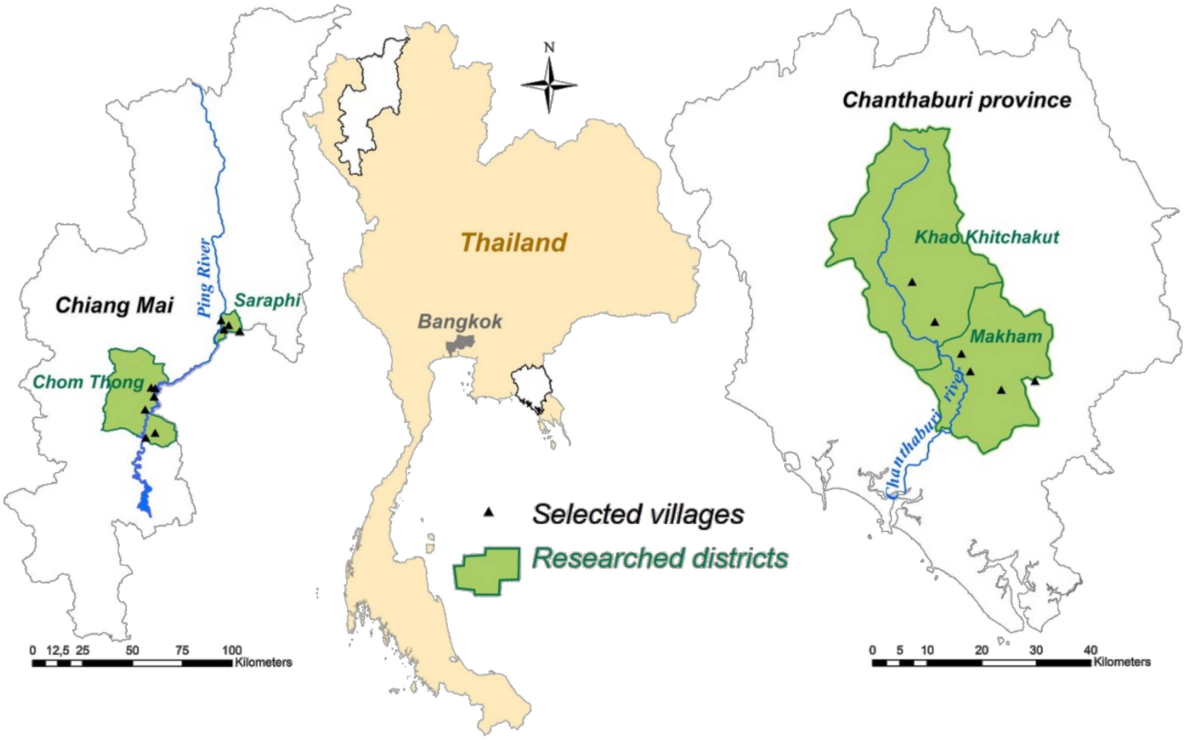

Figure 1.4 The research area comprises 10 villages from two districts of Chiang Mai province (Northern Thailand) and 6 villages from two districts in Chanthaburi province (Eastern Thailand). Source: own representation with data from DIVA-GIS (2014).

1.8 Overview of data and software

The choice models presented in Chapters 3 and 4 were fitted on primary data from the two above mentioned DCE surveys in Chiang Mai, in June 2013, and in Chanthaburi, in November 2013.

A sampling technique of probability proportional to acreage [cultivated with longan (*Dimocarpus longan*)] was applied to select ten villages in Chiang Mai. This was followed by a random selection of households that, at that time, were registered as longan farming in each village's DoAE (Department of Agricultural Extension) office. As a result, the DCE in Chiang Mai yielded a total of 2376 choice observations from 198 household heads each answering 12 choice exercises. In Chanthaburi, in contrast, neither the selection of villages nor of the respondents could be guided by a random procedure, as this was not allowed by the competent authorities (e.g., the provincial administration and each village's head). Accordingly, the Chanthaburi dataset consists of 1524 (non-representative) choice observations from 127 farmers that were selected by village heads from their corresponding fruit (i.e., rambutan, longan and durian) growing communities, the latter of which were designated for the survey by local administrative and academic authorities.

Both DCEs were based on efficient designs for random parameter models that were generated by inserting prior parameter values (i.e., prior part-worth estimates) into Ngene (version 1.1.1) software. The "priors" that informed the generation of the experimental design for Chiang Mai were estimated by fitting a conditional logit model on a DCE pilot that was conducted with 27 respondents from three villages in the district of Saraphi, namely Pa Khed Thee (May 26, 2013), Dea Hgon (May 28, 2013) and Ton Heaw (May 29, 2013). On the other hand, the priors for the experimental design of Chanthaburi were estimated with a conditional logit model that was fitted on the data obtained from the final DCE survey conducted in Chiang Mai.

The raw DCE data was inserted into MS Excel spreadsheets that were converted with Stat/Transfer (version 12) into the .lpj data format that is used to run choice analyses with the NLOGIT 5/LIMDEP 10 statistical package.

1.9 Thesis structure

The body of research of this cumulative dissertation consists of three scientific papers[23] that are presented in logical (not chronological) order in Chapters 2, 3 and 4, in accordance with the sequence of specific objectives that is offered in Section 1.5.

Chapter 2 presents an analytical framework to optimize supply of crop pollination and honey from wild and managed bees for diverse socio-economic and ecological settings. Among this dissertation's articles, it comprehends the (chronologically) last analyses of my research. It builds on the microeconomic model by Rucker et al. (2012) and aims at explaining a more diverse range of equilibria resulting from the interaction between beekeepers and bee-dependent crop farmers.

Chapter 3 presents the random parameter logit models that were fitted on the Chiang Mai dataset alone. Here, we estimated the marginal preferences (i.e., part-worths) and WTP of longan farmers for the attributes constituting the DCE's hypothetical bee conservation policies (cf. Section 1.4.2) and explained the corresponding preference heterogeneity with idiosyncratic variables.

In Chapter 4, we compare the preferences and WTP of orchardists from Chiang Mai and Chanthaburi, by fitting the pooled datasets with generalized mixed logit models. We specified these models such as to control for possible differences in the overall scale of utility between (and within) both samples and to further test whether understanding the DCE exercise and/or possessing the relevant knowledge (regarding the importance of conserving pollinators) may have reduced the subjectiveness of farmers and thus the variability in the unobserved influences on their choice decisions. The GMXL model framework additionally offers a specification that we used to obtain WTP space estimates, which are not prone to the potential biases resulting from calculating WTP point estimates.

Lastly, Chapter 5 offers an overall discussion concerning this dissertation's analyses, contributions and caveats, and concludes with recommendations and an outlook over future research perspectives.

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[23] Cross-references have been added to the scientific articles constituting this thesis and the articles' original numbering format has been modified to fit the span of this cumulative dissertation. Chapters 2, 3 and 4 are otherwise verbatim copies of the submitted (published in the case of Chapter 3) manuscripts.

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Chapter II

The optimal supply of crop pollination and honey from wild and managed bees: an analytical framework for diverse socio-economic and ecological settings[24]

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Keywords: Bee-mediated crop pollination, Honey production, Ecosystem services, Pollination markets, Conservation policy, Transaction costs, Bio-economic model.

Abstract

Hiring pollination services has become an important strategy to secure fruit set in many pollinator-dependent crops, especially in monocultures such as the Californian almond (*Prunus dulcis*) groves, where the European honeybee (*Apis mellifera*) is frequently deployed to compensate deficiencies in the natural provision of insect-mediated pollination. Such beekeeper-farmer cooperation has been the focus of many economic models, although it is only one in a variety of setups under which bees (wild and managed) contribute to agriculture. Based on a thorough literature review we extend the basic model by Rucker et al. (2012) to describe farmer-beekeeper interactions as determined by diverse agro-ecological, institutional and socioeconomic conditions. The generalized model serves as an analytical framework to classify real world farmer-ecosystem-beekeeper interactions, to identify possible causes for a suboptimal (or deficient) crop pollination and to formulate informed policy recommendations.

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2.1 Introduction

With ever diminishing wild pollinator populations worldwide (Potts et al. 2010; González-Varo et al. 2013), the role of pollination markets has increased and, in some instances, commercial crop-pollination has almost completely substituted for this ecosystem service (Klein et al. 2012). This paper builds on the theoretical model proposed by Rucker et al. (2012), which describes the determinants of equilibrium bee wages (i.e., per colony remuneration to beekeepers consisting of both in-kind honey payments and pollination fees) in the specific case of almond (*Prunus dulcis*) pollination markets of California, USA, where commercial pollination services are almost exclusively provided by *Apis mellifera* beekeepers from across the nation. We extend this model by taking into account the economic benefits from a given agro-ecosystem's wild bees and diversity of domesticable bee species, thus covering a broader range of agro-ecological contexts. After an overview on the role of bees in crop pollination (Section 2.1.1) and on pollination related markets (Section 2.1.2), we depart from the model by Rucker et al. (2012) (Section 2.2.1) to gradually add complexity and derive exemplary equilibria (Section 2.2.2), which we analyse and discuss as we compare them with real world cases (Section 2.3). We are convinced that the extension of the well-founded model by Rucker et al. (2012) is necessary if one wants to analyse all relevant farmer-beekeeper-nature interactions. In fact, the simpler model (Rucker et al. 2012) fails to represent many of such interactions that have been described in the literature or that we have observed ourselves, and which will be discussed in this study.

2.1.1 The role of managed and wild bees in crop pollination

A conspicuous example of pollination deficits is provided by the almond (*Prunus dulcis*) groves in California, USA. This state alone currently produces 80% of world's almonds, and has more than doubled the area cultivated with this crop in the last three decades, from 168,000 ha (in 1986) to 364,000 ha (in 2016), in reaction to increasing global demand (United States Department of Agriculture 2016a). Californian almond orchards rely heavily on insect pollinators to set fruit (Connell 2000), yet when managed as intensive monocultures, they create areas of permanent spatial and temporal homogeneity in the agro-ecosystem that restrict wild pollinators to the field margins for most of the year (National Research Council 2007; Marini et al. 2012; Saunders and Luck 2014; Saunders 2016). Wild bee species can therefore only be found visiting almond flowers in orchards with adjacent semi-natural habitat or vegetation strips (Klein et al. 2012). In fact, Californian almond growers rely almost exclusively on managed European honeybees for crop pollination. In 2016, approximately 76% of the 2.59 million nationwide available honeybee colonies were required in the central valleys of California for almond pollination; the state of California itself provided 1.14 million thereof (Goodrich 2016; United States

Department of Agriculture, 2016b), while the remainder were brought by migratory beekeepers[25] from other parts of the USA (Rucker et al. 2012). The reliance on a single pollinator species has nevertheless become problematic in the USA, where the combination of parasites (esp. the parasitic mite *Varroa destructor*) and a syndrome known as colony collapse disorder (CCD) has compromised the health of managed honeybee (*A. mellifera*) colonies (van Engelsdorp et al. 2010; Lee et al. 2015; United States Department of Agriculture 2016b; Watson and Stallins 2016) and consequently their availability for pollination services in the last decade (National Research Council 2007; Goodrich 2016). On the other hand, wild pollinator abundance and diversity offer an insurance when health issues or adverse environmental conditions prevent crop flower visitation by honeybees (Winfree et al. 2007; Brittain et al. 2013; Klein et al. 2012), and may even enhance fruit set regardless of honeybee abundance (Garibaldi et al. 2013).

The Californian almond groves manifest an extreme case of natural pollination services being almost entirely replaced by managed European honeybees. As a matter of fact, in most agro-ecosystems worldwide, wild bees and other wild pollinators continue to play a key role in crop pollination (Roubik 1995; Klein et al. 2007), which may be supplemented, rather than substituted for, by managed honeybees (Garibaldi et al. 2013). In Chiang Mai province (Northern Thailand), for instance, wild pollinators still seem to fulfil the pollination requirements of longan (*Dimocarpus longan*), a perennial tree crop that is widely cultivated in the lowlands of the Chiang Mai–Lamphun valley and depends on flower visitation from honeybees (especially the Eastern honeybee *Apis cerana*) and stingless bees (Hymenoptera: Apidae: Meliponini) to set fruit (Blanche et al. 2006; Pham 2012). To our knowledge, no crop pollination crisis has yet been reported for this region, although its wild pollinator populations may be at risk, considering that in the last several decades the remaining forest cover of Northern Thailand (the largest relative to other regions in Thailand) has experienced a rapid decline due to agricultural expansion and deforestation, among other causes (Giri et al. 2003; Thomas 2006; Trisurat et al. 2010). Moreover, this region's land (i.e., planted in both traditional staples and forestland) is increasingly being converted, giving way to the intensive commercial production of fruit trees, in response to higher returns to land and labor (Ekasingh and Ngamsomsuke 2005; Thomas 2006).

Thailand's National Economic and Social Development plan has oriented the country's agriculture to the production of capital-intensive cash crops (Kasem and Thapa 2012), demanding a fourfold increase in pesticide imports over the past decade (Thapinta and Hudak 2000; Panuwet et al. 2012). This is especially true for Chanthaburi province (Eastern Thailand), which encompasses one of the largest fruit producing

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[25] Migratory beekeeping is the practice of moving bee hives from farm to farm in search of different foraging sources. It often involves traveling long distances and following the blooming periods of different crops throughout the year.

areas in the country (Thai Office of Agricultural Economics 2014). In some farming communities of this province, rambutan (*Nephelium lappaceum*) and durian (*Durio zibethinus*)[26] farmers started managing stingless bees (mainly from the genera *Tetragonula* and *Lepidotrigona*) to compensate for presumed deficits in their crops' pollination (Thanuart and Makhonpas 2013; Boongird 2014; Chuttong et al. 2014). With this respect, experts (Thanuart and Makhonpas 2013) and focus group discussions informed us in 2013 that rambutan and durian farmers preferred keeping stingless bees (a practice known as meliponiculture), over European honeybees, due to their shorter flight range[27], which reduces their exposure to the pesticides that may be applied in neighbouring farms. Phankaew (2016) further reports that *A. mellifera* beekeepers in Soi Dao district (Chanthaburi province) avoided placing their honeybee colonies on off-season longan farms, where flowering is induced with potassium chlorate to produce year-round off-season fruit. This treatment thus results in crop protection schedules that vary over neighboring farms and make it difficult for beekeepers to protect their colonies from pesticide exposure.

2.1.2 Markets for pollination services and floral nectar

Thailand offers an interesting case for how markets can adapt to ecologically different regions with regard to their ability to provide an environmental good. In Chanthaburi province, for instance, a beekeeper had been renting, for already 10 years (in 2014), European honeybee colonies to longan orchardists as a pollination supplement, at a price of 50 Thai baht (i.e., €1.12, as of June 1, 2014) per colony, per season (Phankaew 2016). Nevertheless, many Chanthaburi farmers have opted for becoming meliponiculturists (Chuttong et al. 2014), due to the advantages that stingless bees have over European honeybees, i.e., the reduced hazard of exposure to synthetic pesticides and the relatively lower costs[28] incurred with this practice.

In Chiang Mai province, in contrast, beekeepers often pay longan farmers a fee to let their colonies forage longan nectar on their farm (Wongsiri et al. 2000; Narjes and Lippert 2018, Chapter 4). During our 2013

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[26] Durians (*D. zibethinus*) are exclusively pollinated by nocturnal pollinators such as fruit bats. Stingless bees (Hymenoptera: Apidae: Meliponini) and honeybees are nevertheless the major diurnal flower visitors of durians, yet they do not succeed in transferring pollen onto the stigmata (Bumrungsri et al. 2009). Bees are thus mistakenly credited for the pollination of durians (Boongird 1992), a perception that seems to be shared among Chanthaburi durian farmers (Boongird 2014; Narjes and Lippert 2018).

[27] The flight range of stingless bees is proportional to their body size, which may constrain the dispersion capacity of smaller species within the limits of forest fragments, further increasing the risk of extinction in their local population (Araújo et al. 2004). Compared to the wide foraging range of honeybees (*A. mellifera*), typically reaching distances of 5 km (Beekman and Ratnieks 2000), stingless bees have a considerably smaller flight radius: e.g. ~600 m for the subgenus *Trigona* (Boontop et al. 2008). Boonithee et al. (1991), for instance, reported a high abundance of *Trigona* spp. in a longan orchard in Northern Thailand within distances of 50 and 200 m from the adjoining forest, yet decreasing sharply between 2.5 and 4 km.

[28] Stingless bee colonies can be easily extracted from nearby forests, their propagation is relatively simple and consumes considerably less time and inputs for their maintenance and honey harvest than European honeybees (Narjes and Lippert 2010; Chantawannakul et al. 2004; Chuttong et al. 2014).

survey (Narjes and Lippert 2016, Chapter 3), 57 longan orchardists (from 198 that were interviewed) reported that, each flowering season, they allowed an average of 66.47 colonies on their farm (between 10 and 250 colonies per farm) and received an average payment of 0.56 USD (*SD*=0.60 USD)[29] per honeybee colony per season, and/or an average in kind payment of 7.1 bottles (*SD*=11.3 bottles) of honey (containing each 750 ml ≈ 1 kg) per season. Longan flowers produce abundant nectar of which honey can be sold at a premium price, i.e., 1.5 USD/kg, which is 50% higher than honey from the second major honey plant in the region, i.e., Siam weed (*Chromolaena odorata*, formerly *Eupatorium odoratum*) (Thapa and Wongsiri 1997, Chantawannakul et al. 2004). Moreover, large-scale beekeeping operations have been attracted to this region where, each blooming season (February-March), they move about 120,000 *A. mellifera* colonies to longan farms in the lowlands of the Chiang Mai–Lamphun valley. They produce approximately 4,200 metric tons of longan honey each year, 55-60% of which is destined for the export market (Wongsiri et al. 1998; Chantawannakul et al. 2004; Seanbualuang 2012).

Contrary to almond honey, which is considered unpalatable to humans and therefore commercially uncompetitive (National Research Council 2007; Goodrich 2016), longan honey is highly valued and enjoys a great demand in the Asian (especially in Hong Kong, Taiwan, Malaysia and Singapore) and U.S. markets (Wongsiri et al. 1998). The same is true for stingless bee honey, which is highly regarded for its organoleptic and (presumed) medicinal properties (Chuttong et al. 2014; Chuttong et al. 2016).

The value of honey is an important determinant in the price formation of pollination fees (Cheung 1973; Rucker et al. 2012) and constitutes a defining aspect in the differences between the beekeeper-orchadist dynamics of the Californian almond groves and those of the Thai agro-ecosystems described by Narjes and Lippert (2018). Another such aspect is the role of wild (native) pollinators in crop-pollination, which is marginal in the Californian almond groves, while it is still (seemingly) important in the longan orchards of Chiang Mai province. Moreover, unlike in California, Thailand's native bee fauna includes many wild species that produce honey (i.e., of the genus *Apis: A. dorsata*, *A. florea*, *A. andreniformis* and *A. cerana*, and many stingless bee spp.), of which some can be managed in hive boxes (i.e. kept permanently or baited and temporally kept until they abscond), such as *A. cerana* and some stingless bee species of the genera *Tetragonula* and *Lepidotrigona* (Wongsiri et al. 2000; Chuttong et al. 2016).

The scenario currently found in Chanthaburi province, where many farmers have responded to pollination deficits by becoming beekeepers, is a special case that has not been illustrated by the economic model analyses currently found in the literature, which center their focus on the US American pollination markets with special regard to the Californian almond groves. For instance, building on the theoretical framework first proposed by Cheung (1973), Rucker et al. (2012) developed a theoretical model that explains

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[29] 16.9 Thai baht (THB) per colony; 1 USD=30.21 THB, as of June 1, 2013. These values were calculated over the 57 respondents that allow beekeepers placing honeybee colonies on their orchard.

pollination fees by optimizing the stocking density of honeybee colonies (i.e., bee colonies per unit of cultivated land) and determining their supply as a function of honey prices, beekeeping costs and pollinated crop acreage (cf. Section 2.2.1). They further test these predictions using regression analysis and panel data (12 crops) from the annual survey (1987-2009) of Pacific Northwest beekeepers. Nevertheless, the analyses by Rucker et al. (2012), like those by Willett and French (1991) and by Champetier et al. (2015), which also deal with modeling the economics of crop pollination and honeybee products, do not take into account the contribution of wild (native) bees to crop pollination.

In a simpler set of models, Ward et al. (2010) recognize the role of alternative pollinators (i.e., with respect to *A. mellifera*) in the formation of pollination prices in the Californian almond groves and the sweet cherry (*Prunus avium*, also highly dependent on insect pollination) orchards of California and the Pacific Northwest. They regressed pollination fees on the total cultivated acreage and the number of available European honeybee colonies and then simulated the effect of a reduced demand of commercial pollination services that would result from a partial contribution of the blue orchard bee (*Osmia lignaria*, a native pollinator that can be managed, yet does not produce honey) to crop pollination. Not surprisingly, their analyses predict that the availability of an alternative pollinator would result in smaller pollination fees, and that this effect would be greater for almonds, since (unlike cherries) almond honey cannot be commercialized for human consumption.

Rucker et al. (2012) denote as “uncommon” the case of beekeepers paying farmers for the privilege of placing their colonies on cultivated land. Nevertheless, beyond the boundaries of the Californian almond monocultures, this case may be less uncommon than perceived. For instance, according to a 2016 survey, the contracts between German beekeepers and farmers concerning the remuneration for the effected crop pollination services are crucially based on the proportion to which the resulting honey yields stand relative to the additional crop yields that are attributed to bee-mediated pollination. For crops with abundant nectar such as canola, buckwheat or sunflower, instead of a pollination fee, the agreement between farmer and beekeeper often results in a payment to the former that typically consists of honey (Wolf, 2016). Similarly, in the theoretical model by Rucker et al. (2012), this equilibrium results from a substantial marketable honey output or a limited marginal contribution of pollination to fruit production at the equilibrium stocking density of honeybee colonies (e.g., for crops with low or zero pollinator dependence). In Chiang Mai, a limited contribution of European honeybees to longan pollination may result from this service already being delivered close to its maximum by abundant wild bees (i.e., further marginal contributions to fruit pollination approach zero). Modelling the development of pollination fees in Thailand’s (or similar) agro-ecological context would thus require accounting for the contribution of crop pollination from commercial beekeepers (i.e., European honeybees) relative to the various levels of pollination services that may be delivered by wild bees from adjacent natural and semi-natural habitats. Moreover, the analysis by Rucker et al. (2012) does not take into account the transaction and production costs of various beekeeping setups locally available to the farmers as alternative crop-pollination supplements and additional income sources.

2.2 A general theoretical framework for the optimal allocation of pollinating bees

In view of the deficiencies identified in the previous section, we propose a comprehensive modelling framework that can accommodate the complexity and diversity of the agro-ecosystems cultivated with melittophilous crops, which offer floral resources to both managed and wild bees and thus determine the interaction of farmers with beekeepers.

We start by identifying the following exemplary cases, which in accordance to the different underlying agro-ecological and economic variables, our model should be able to describe as bioeconomic (market) equilibria:

1) *No cooperation occurs between farmers and beekeepers*
2) *Farmers pay beekeepers for pollination services*
3) *Beekeepers pay farmers for the privilege of placing their bee colonies on cultivated land*
4) *Farmers and beekeepers cooperate with no monetary payments involved*
5) *Beekeepers deliver crop pollination as an externality*
6) *Beekeepers interact with farmers in an agro-ecosystem that sustains wild bees*
7) *Farmers consider engaging in on-farm beekeeping*

2.2.1 Basic economic model: deriving equilibrium stocking densities

Our analysis builds on the theoretical model proposed by Rucker et al. (2012), which parts from assuming a competitive, hypothetical market in which multi-product firms rent A acres of land and B bee colonies to produce both fruit and honey in respective quantities F and H, as given by the following pair of constant-returns-to-scale functions:

$$H = G_H(A, B) \text{ and } F = G_F(A, B). \tag{2.1}$$

Rucker et al. (2012) further define the bee stocking density (number of bee colonies per unit of land) $b := B/A$ as the sole determinant of per-acre outputs $g_H(b)$ and $g_F(b)$, such that the total (linearly homogeneous) outputs G_H and G_F proportionately increase by the factor A. Thus, the objective function of a profit maximizing farmer with given acreage is:

$$\max_b \pi(b) = P_H * g_H(b) + P_F * g_F(b) - w * b, \tag{2.2}$$

where P_H and P_F are the market prices of honey and fruit and $w = P_p + P_H * g_H/b$ represents a market “bee wage” that consists of a per-colony pollination fee (P_p) and an in-kind honey payment.

Restricting (2.2) to be concave, the first order condition for a maximum implies that w equates the sum of values of marginal products (VMPs) of honey and fruit, i.e., the total value marginal product of bees ($TVMP$). A price-taking firm's optimal decision is therefore a function of factor prices, i.e., the optimum stocking density $b^* = b^*(w, P_H, P_F)$, as

$$TVMP := P_H * g'_H(b^*) + P_F * g'_F(b^*) := VMP^H(b^*) + VMP^F(b^*) = w. \qquad (2.3)$$

The equilibrium bee wage w^* results from the demand (i.e., $TVMP$ aggregated over all farmers) meeting the supply $Q_S(w, P_H, k)$, where per-colony cost k sums up factors affecting the costs of beekeeping; both functions are aggregated over A^* equilibrium homogenous acres (Rucker et al. 2012). In equilibrium, the property rights of honey are assigned to beekeepers, while orchardists retain ownership of the fruit (ibid). According to Barzel (1997), this arrangement should maximize the value of the economic rights over these assets, as price and production risk tend to be borne by the actors with most influence over the respective sources of income variability.

Notwithstanding that the functional forms of $g_H(b)$ and $g_F(b)$ are unknown, we reasonably assume that their marginal products (accordingly their corresponding VMPs) diminish with each additional bee colony. This model also represents a short- to middle-run optimization problem, as it assumes that the per-acre density of fruit trees to be pollinated is given.

2.2.2 Bioeconomic equilibria and extensions to the basic model

2.2.2.1 No cooperation occurs between farmers and beekeepers

We begin our analysis examining equilibrium ***1)***, which may result from either one or a combination of the following two circumstances:

a) The bee wage w^* may be prohibitive vis-à-vis the relatively low potential $TVMP$ that a cooperation with commercial beekeepers could generate, if they would place their bee colonies in the orchard (Figure 2.1). In a competitive market, w^* can be seen as the opportunity costs that a beekeeper perceives from failing to deploy her colonies on another location that yields a higher $TVMP$, e.g., in another orchard that has a higher demand for pollination services and/or which offers more valuable nectar. Accordingly, the equilibrium stocking density b^* (supposedly reached when fruit and honey production is optimised) will equal zero.

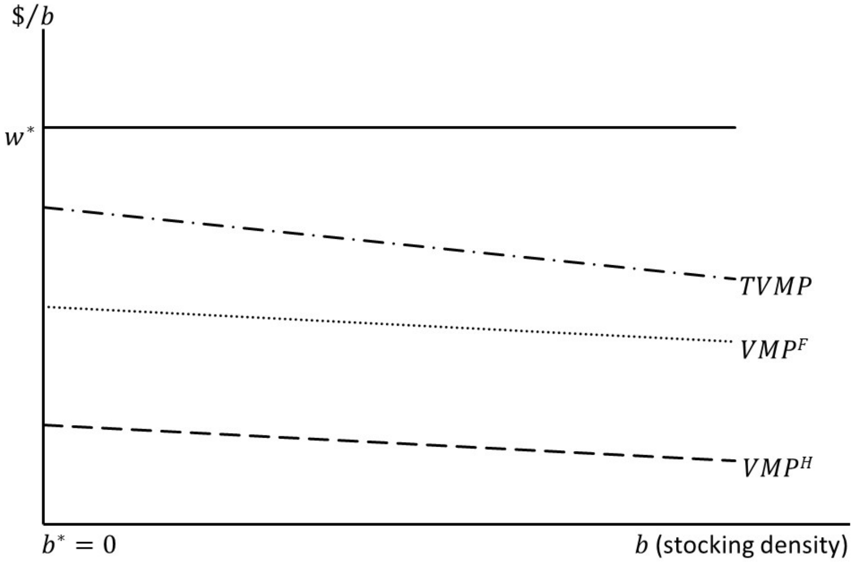


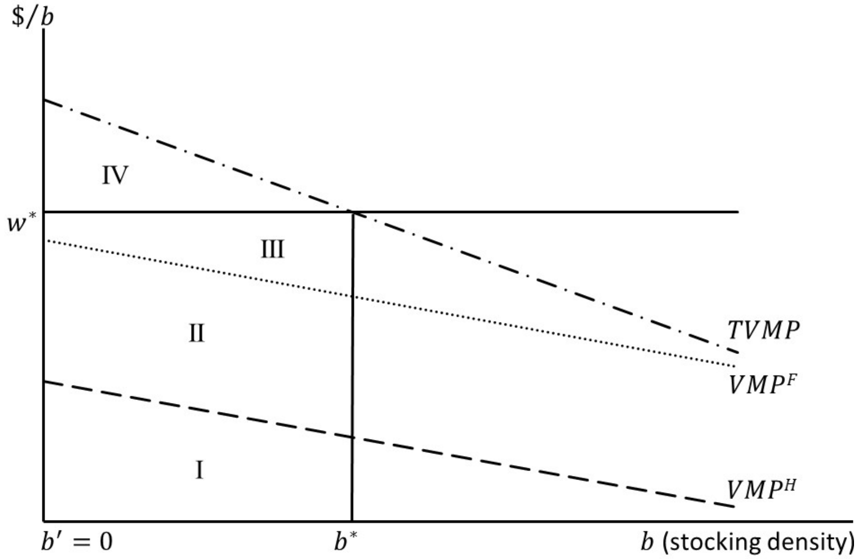
Figure 2.1. In equilibrium, $\mathbf{b}^*$colonies are placed ($\mathbf{b} :=$ bee colonies/acre); no cooperation occurs between farmers and beekeepers. The total value of marginal products ($\boldsymbol{TVMP}$) equals the sum of values of marginal product of fruit ($\boldsymbol{VMP^F}$) and of honey ($\boldsymbol{VMP^H}$). Per-colony bee wages ($\boldsymbol{w}$) consist of a pollination fee and an in-kind honey payment.

b) Contracts between farmers and beekeepers can entail prohibitive transaction costs. These may result from bringing beekeepers and farmers together and from monitoring and enforcing the agreements, all time-consuming tasks that may or may not be relegated to pollination brokers and third-party inspection services in exchange for a fee. Additionally, the intricacies of bee-mediated pollination and crop pollination requirements impose information costs on farmers, while beekeepers face transportation costs and the costs of coordinating pollination services and honey production across different crops and blooming seasons (Cheung 1973; Rucker et al. 2001; Rucker et al. 2012; Goodrich 2016). Whereas some transaction costs, such as monitoring for hive strength, arise on a per-colony basis, we will for now discuss those of fixed character. At any rate, if the fixed transaction costs are higher than the potential welfare gains from cooperating, an interaction with farmers and beekeepers will not take place. As follows, the welfare gains illustrated in Figure 2.2 (i.e., area IV) and Figure 2.3 (i.e., area IV+V), potentially realizable under equilibria ***2)*** and ***3)*** respectively, would become welfare deadweight losses if prohibitive transaction costs would force the equilibrium stocking density to $b' = 0$.

In the following, we present equilibria ***2)*** and ***3)***, which Rucker et al. (2012) also use to illustrate contrasting payment signs, i.e., the direction at which payments between farmers and beekeepers will flow.

2.2.2.2 Farmers pay beekeepers for pollination services

Figure 2.2 depicts equilibrium ***2)***, where farmers pay beekeepers with both all the honey produced, i.e., area I, and a pollination fee amounting to II+III. After subtracting the pollination fee from the value of the additional fruit produced resulting from bee-pollination (equivalent to I+II=II+III+IV), farmers retain the surplus IV (cf. Figure 2.2, welfare change for $b\colon 0 \rightarrow b^{*}$), which may be used to cover their transaction costs[30] (and those of the beekeepers) and possibly as an additional payment to incentivize beekeepers, beyond the mere coverage of per-hive opportunity costs w^{*}, to place their colonies in the orchard.



Welfare changes for
b: $0 \rightarrow b^{*}$
Beekeepers:
$\mathrm{I} + \mathrm{II} + \mathrm{III} = w^{*} \times b^{*}$
Farmer:
$\mathrm{I} + \mathrm{II} - (\mathrm{II} + \mathrm{III}) = \mathrm{I} - \mathrm{III}$
$= \mathrm{IV}$ (as $\mathrm{I} = \mathrm{III} + \mathrm{IV}$)

Figure 2.2. In equilibrium, $\boldsymbol{b^{*}}$colonies are placed ($\boldsymbol{b} :=$ bee colonies/acre); farmers pay beekeepers for crop pollination services. Prohibitive transaction costs may nevertheless force the equilibrium to $\boldsymbol{b'}$. The total value of marginal products ($\boldsymbol{TVMP}$) equals the sum of values of marginal product of fruit ($\boldsymbol{VMP^{F}}$) and of honey ($\boldsymbol{VMP^{H}}$). Per-colony bee wages ($\boldsymbol{w}$) consist of a pollination fee and an in-kind honey payment. Modified from Rucker et al. (2012).

2.2.2.3 Beekeepers pay farmers for the privilege of placing their bee colonies on cultivated land

A converse scenario, where beekeepers pay farmers, is conceivable for an orchard that, at equilibrium, delivers marginal benefits from honey that are superior to the marginal benefits from fruit, such as oranges (*Citrus aurantium*). Figure 2.3 illustrates equilibrium ***3)***, one such scenario where beekeepers would place $\tilde{b}$ colonies (per acre) for free (i.e., until the marginal benefits from nectar equal their opportunity costs w^{*}). However, under negligible transaction costs, farmers would demand additional colonies (i.e., beyond $\tilde{b}$) until reaching the equilibrium b^{*}. In that case, beekeepers would at maximum pay farmers a value equivalent to IV minus III for the exclusive right to place their colonies in the orchard, while farmers would retain the surplus from pollination V, plus a certain share from the honey production surplus.

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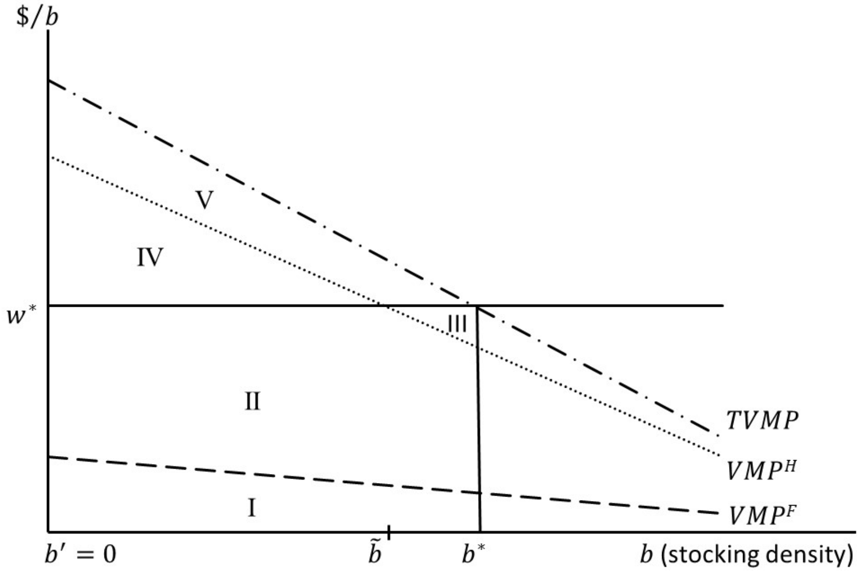
[30] Notice: finding the optimum b^{*} and the magnitude of the payment II+III is not possible without incurring a certain amount of transaction costs.



Welfare changes for b: $0 \rightarrow b^*$
Beekeepers:
$\mathrm{I} + \mathrm{II} + \mathrm{IV} - (\mathrm{IV} - \mathrm{III})$
$= \mathrm{I} + \mathrm{II} + \mathrm{III} = w^* \times b^*$
Farmer:
$\mathrm{I} + (\mathrm{IV} - \mathrm{III})$
$= \mathrm{IV} + \mathrm{V}$ (as $\mathrm{V} + \mathrm{III} = \mathrm{I}$)

Figure 2.3. In equilibrium, beekeepers will place $\boldsymbol{\tilde{b}}$ colonies for free, yet the farmer will demand additional colonies until reaching the optimum $\boldsymbol{b}^*$ ($\boldsymbol{b} :=$ bee colonies/acre); beekeepers pay farmers for exclusive rights to floral resources. Prohibitive transaction costs may nevertheless force the equilibrium to $\boldsymbol{b'}$. The total value of marginal products ($\boldsymbol{TVMP}$) equals the sum of values of marginal product of fruit ($\boldsymbol{VMP^F}$) and of honey ($\boldsymbol{VMP^H}$). Per-colony bee wages ($\boldsymbol{w}$) consist of a pollination fee and an in-kind honey payment. Modified from Rucker et al. (2012).

2.2.2.4 Farmers and beekeepers cooperate with no monetary payments involved

Decisive for the direction of payments is thus the proportion to which (net of transaction costs) the honey surplus (available to the beekeepers to compensate the farmer) stands relative to the share of total crop pollination benefits that the farmer has to invest to balance the opportunity costs of beekeepers that could not be met with honey benefits, i.e., the ratio of area IV to area III in Figure 2.3. If IV > III, beekeepers will pay farmers, while the opposite is true when III > IV. In light of this, it becomes evident that case ***4)***, an interaction between farmers and beekeepers involving no monetary payments, is likelier to take place as III ≈ IV (Figure 2.3).

2.2.2.5 Beekeepers deliver crop pollination as an externality

For an orchard with relatively low VMP^F, the fixed transaction costs are, at least from the farmer's perspective, unlikely to be trivial: they may be prohibitive or leave little incentive for the farmer to enter into an agreement that also accommodates the additional benefits from crop pollination. In other words, the costs of e.g. estimating the actual VMP^F and enforcing the agreement may be perceived by the farmer as higher than the aggregated VMP^Fthey would expect. Such circumstance leads us to equilibrium ***5)***, the case where, in an orchard with high VMP^H (relative to w^*), beekeepers and farmers agree to interact, contemplating solely the marginal benefits of honey. Figure 2.4 shows the equilibrium stocking density being settled at $\tilde{b}$, as beekeepers compare w^* to the VMP^H. After covering their opportunity costs (area

I+II), beekeepers are left with a surplus of honey benefits (area III), while farmers enjoy the externality of crop pollination (area IV, equivalent to area I). Here too, beekeepers may use part of (or at maximum) the surplus III (e.g., as in-kind honey payments) as an additional incentive for the farmer to give them exclusive rights to the orchard's nectar. In this case, failing to reach the optimum b^* due to transaction costs entails deadweight losses amounting to V.

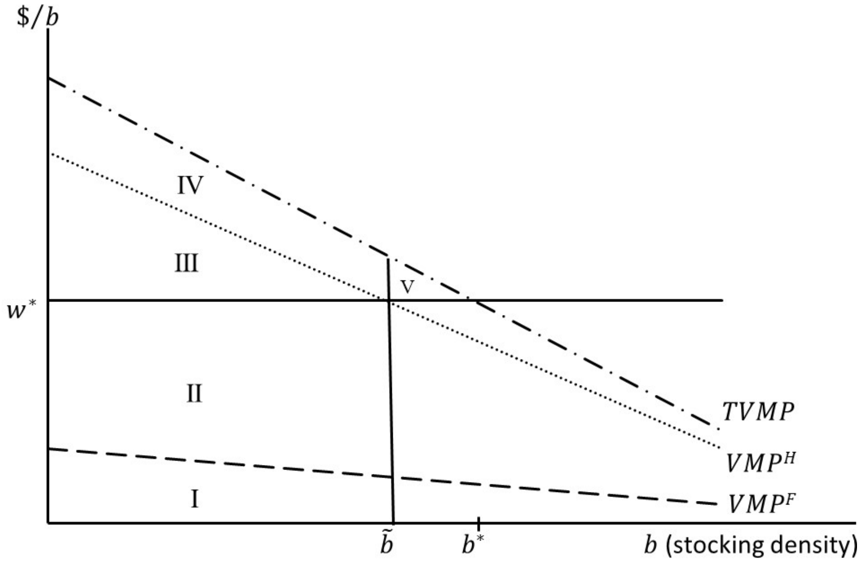


Figure 2.4. Pollination as an externality: in equilibrium, beekeepers place $\tilde{\boldsymbol{b}}$ colonies ($\boldsymbol{b} :=$ bee colonies/acre); beekeepers pay farmers for exclusive rights to floral resources and allocate the stocking density solely based on honey. The optimum $\boldsymbol{b}^*$ is missed due to transaction costs relevant to $\boldsymbol{VMP^F}$. The total value of marginal products ($\boldsymbol{TVMP}$) equals the sum of values of marginal product of fruit ($\boldsymbol{VMP^F}$) and of honey ($\boldsymbol{VMP^H}$). Per-colony bee wages ($\boldsymbol{w}$) consist of a pollination fee and an in-kind honey payment.

2.2.2.6 Beekeepers interact with farmers in an agro-ecosystem that sustains wild bees

Until now, the marginal benefits from an interaction between the orchard and wild bees have not been made explicit in our analysis. In fact, the VMP curves we have discussed so far are implicitly shaped by the agro-ecological conditions that determine, among other factors, the local population of wild bees.

Our analysis proceeds including the marginal benefits of wild bees, by redefining the variable b (Eq. 2.2) such that it captures the sum of wild and managed bee colonies (respectively b_w and b_m) that in a given agro-ecosystem are available to pollinate each orchard's acre and to produce honey from its nectar. The distinction between wild and managed bees may seem arbitrary, being that some species can simultaneously fall into both categories. For the sake of simplicity one may nevertheless assume that, in a given agro-ecosystem, the marginal benefits from all bee species within each category (i.e., wild and managed) can be derived from the per-acre production functions $f_H(b_w)$ and $f_F(b_w)$ and $h_H(b_m)$ and $h_F(b_m)$ respectively. In doing so, one has to be wary about the possible differences between each category's per-colony marginal benefit curves from pollination and honey. Therein, the issue arises of which bees, i.e., wild or managed, will visit the orchard first, thus yielding the highest potential marginal benefits. This is immaterial with regard to crop pollination, since all of its related benefits are retained by the farm regardless of which bees are delivering them. On the other hand, the issue remains of which bees will produce honey from the first units of available nectar, given that the honey produced by wild bees may or may not be harvested. We therefore introduce another simplifying yet sensible assumption, namely that the proximity to natural or semi-natural habitats will give wild bees priority access to the orchard's floral resources. Consequently, our analyses ascribe the first units of nectar to the production of honey from wild bees.

We further assume that an orchardist will only consider hiring a commercial beekeeper's pollination services if wild bees leave unrealized positive marginal benefits from crop pollination after having finished visiting the orchard's flowers[31]. At any rate, the marginal benefits in the equilibria presented above set their baseline (i.e., $b=0$) immediately after the last available wild bee colony has delivered its benefits. It thus becomes apparent that $g_F(b)$ and $g_H(b)$ (Eq. 2.2) are piecewise per-acre production functions that are defined by the sub-functions and corresponding intervals specific to b_w and b_m in the main function's domain. For our analyses, the value of b_w is assumed to be a parameter given by each particular agro-ecosystem (i.e., regardless of whether wild bees are displaced by managed bees)[32], while b_m is the variable to be optimized.

We represent the marginal benefits of the two distinct bee categories in a rectangular coordinate system, by setting one colony of managed bees as the common unit. Thereby, we arbitrarily set the equivalence between both bee categories at the VMP^F, i.e., b_w^e is equivalent to the number of managed bee colonies required to deliver the per-acre pollination services of the locally available bee colonies[33]; the VMP^H of wild bees are accordingly adjusted to b_w^e. We further assume that the VMP^F diminishes at an equal rate for b_m and b_w^e, thus leading to a smooth transition down the pollination demand curve.

Figure 2.5 illustrates one possible case for equilibrium ***6)***, in which the VMP^H curve radically changes when the managed bee colonies are placed in the orchard. The relatively small VMP^H corresponding to b_w^e could represent the case of a farmer being able to only recover a small portion of the wild honey produced

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[31] Beekeepers may nevertheless see in rich floral resources an incentive to bring their colonies to the orchard, regardless of the abundance of wild bees. In the case of beekeeping with European honeybees (*A. mellifera*), which have been extensively introduced to almost every region in the world, it has been widely reported that this species tends to displace other bee species from the richest source of forage to less profitable ones (Goulson 2003).

[32] Bearing in mind that managed bees are likely to be the dominant orchard visitor, the number of available wild bee colonies (if there are any) could theoretically be displaced closer to the origin, i.e., a level below the full potential range of b_w.

[33] Kakutani et al. (1993) for instance conclude from their fertilization experiments that stingless bees (*Trigona minangkabau*) require 1.8 times the amount of workers of a honeybee (*Apis mellifera*) colony (i.e., 2:1 mature or 8:1 small stingless bee colonies per honeybee colony) to attain an equivalently efficient pollination of strawberries (*Fragaria chiloensis* × *ananassa*).

by directly gathering it from the wild bee's natural habitat, a labor-intensive practice known as honey hunting[34]. Here, the diminishing marginal benefits could exemplify the increased effort the farmer has to invest as she searches farther away from her orchard to find the next wild bee colony. The VMP^H increases from the point at which managed bees are brought to the orchard, as more of the orchard's nectar benefits can be harvested and transformed into honey more efficiently. This step in the VMP^H curve (Figure 2.5), indicates the baseline for equilibrium ***3)*** ($b' = 0$ in Figure 2.3). It is worthwhile mentioning that in this example, the VMP^F are low relative to the marginal benefits from honey of managed bees (VMP_m^H), due to a significant contribution to the orchard's pollination needs on account of wild bees, i.e., a positive environmental externality.

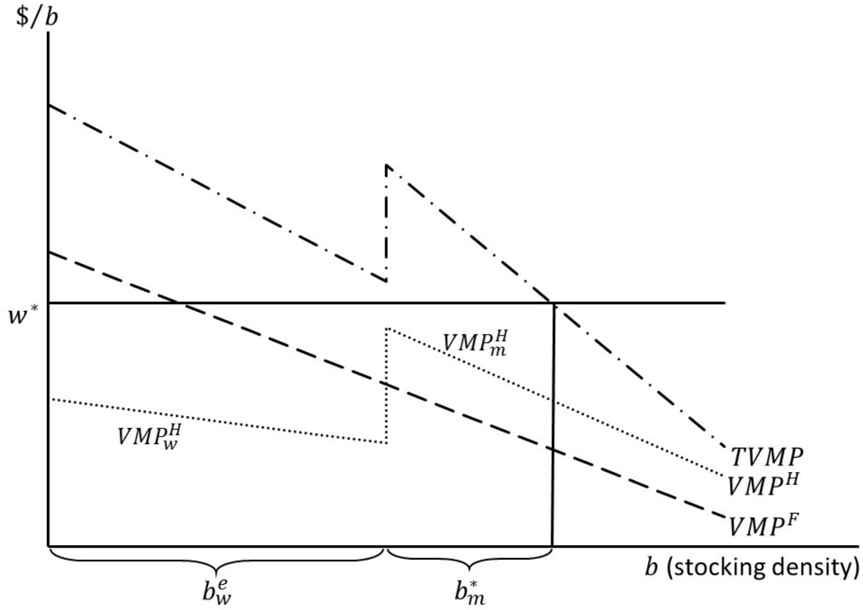

Figure 2.5. Managed bees are brought to the orchard after wild bees have finished delivering their pollination services. In equilibrium, $\boldsymbol{b_m^*}$ colonies are placed ($\boldsymbol{b} :=$ bee colonies/acre); $\boldsymbol{b_w^e}$ is the equivalent number of managed bee colonies that would deliver the pollination services of the locally available wild bee colonies ($\boldsymbol{b_w}$). The total value of marginal products ($\boldsymbol{TVMP}$) equals the sum of values of marginal product of fruit ($\boldsymbol{VMP^F}$) and of honey ($\boldsymbol{VMP^H}$). Per-colony bee wages ($\boldsymbol{w}$) consist of a pollination fee and an in-kind honey payment.

The equilibria presented above could be based on the assumption that multi-product firms optimize the stocking density of managed bees, i.e. regardless of how the benefits will ultimately be distributed between farmers and beekeepers in the real world. At this stage, nevertheless, a sensible question would be whether the farmer can extract a greater benefit from the orchard's nectar, i.e., capturing the VMP^H without requiring the intermediation of external commercial beekeepers. This consideration is important, since it has implications on how the costs of beekeeping enter the optimization.

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[34] This practice requires a special set of skills and is very time consuming. It is rarely practiced by farmers as an important source of additional income (Narjes and Lippert 2010).

2.2.2.7 Farmers consider engaging in on-farm beekeeping

So far, the marginal costs of beekeeping k have implicitly been assumed to be zero (in the profit function) and the VMPs from managed bees have diminished only to account for the constraints inherent to the agro-ecosystem. In the profit maximization problem offered by Rucker et al. (2012), the costs of beekeeping (e.g., per colony hive box investments and maintenance) can be assumed to be equal in every location. If we rewrite Equation (2.2), such to make the costs of commercial beekeeping k explicit as follows

$$\max_{b} \pi(b) = P_H * g_H(b) + P_F * g_F(b) - k * b - (w - k) * b, \tag{2.4}$$

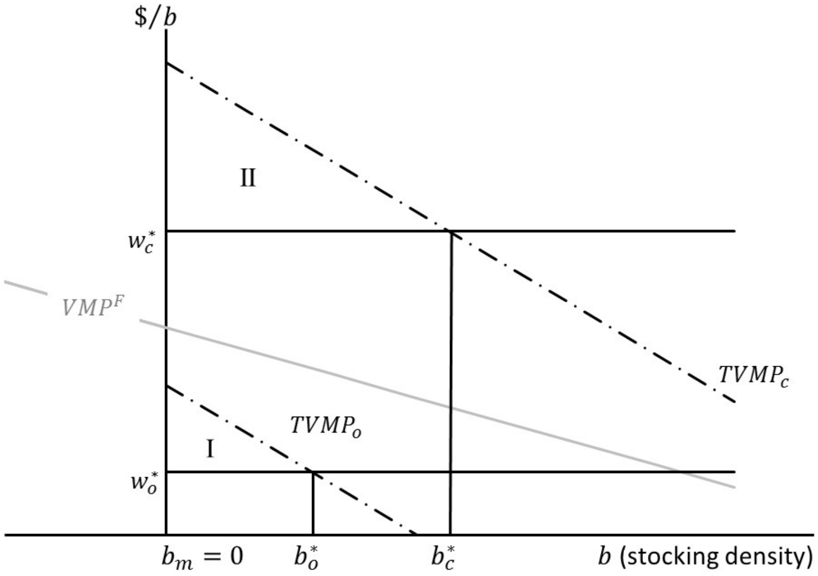
it becomes evident that k is cancelled out by the identical costs that would be incurred at any alternative location that offers the opportunity to earn w^*. Nevertheless, on-farm beekeeping generates a different type of opportunity costs. As a part-time beekeeper, a farmer is typically bound to leave her bee colonies on (or near) the farm and can hardly practice migratory beekeeping with the same flexibility that commercial operations enjoy, due to e.g., infrastructure and time limitations. Therefore, instead of failing to capture the benefits from alternative beekeeping locations, a part-time beekeeper's opportunity costs mainly consist of the forgone benefits from investing her labor capacity in other activities, e.g. on the own farm. We therefore distinguish the opportunity costs of commercial beekeeping w_c from those of on-farm beekeeping w_o (i.e., per colony forgone labor wage) and specify the on-farm beekeeping profit maximization problem as follows:

$$\max_{b} \pi(b) = P_H * h_H(b_{os}) + P_F * h_F(b_{os}) - k_{os} * b_{os} - w_{os} * b_{os}, \tag{2.5}$$

where $h_H(b_{os})$ and $h_f(b_{os})$ are the production sub-functions corresponding to on-farm beekeeping, as denoted by the sub-index o, for $s = 1, \dots, S$ bee species that are capable of being domesticated and available to the farmer. It is apparent that, in contrast to commercial beekeeping (Eq. 2.4), k is now relevant for on-farm beekeeping: commercial beekeepers already own the necessary infrastructure, while for farmers acquiring it involves a new investment. In fact, if in the VMP^F for equilibrium ***6)*** (Figure 2.5) we would have otherwise accounted for the difference in marginal costs between on-farm beekeeping and allowing wild bees in the orchard, its curve would have depicted a sudden downward shift (and a steeper slope if increasing marginal costs were assumed) in the segment that corresponds to managed bees. However, to make the analysis of the next equilibria simpler, we deduct the entire costs of either beekeeping activity entirely from the VMP^H. Thereby, the VMP^F is identical for commercial and on-farm beekeeping and the smooth transition on its curve (between wild and managed bees) is conveniently maintained, thus characterizing the nature of positive externality that pollination potentially assumes.

Figure 2.6 and Figure 2.7 represent equilibrium ***7)***, the case of farmers considering the possibility of becoming themselves beekeepers. They depict the total value of marginal products ($TVMP$s) of any bee species s that can be managed both on the farm (as an additional farming activity), or on a larger scale for commercial purposes (including renting out colonies for crop-pollination). Both figures set their baselines ($b_m = 0$) immediately after the last wild bee has delivered its pollination benefits (and neglect the benefits from wild honey). For simplicity's sake, we assume that both beekeeping activities (i.e., on-farm and commercial) use the same bee species and therefore produce equal honey yields that fetch the same honey price. Thereby, equilibria ***7.a)*** and ***7.b)*** (respectively Figure 2.6 and Figure 2.7) differ in the (constant) per colony investments of on-farm beekeeping (k_o): in both figures $k_o > k_c$, yet in the former k_o is much higher than in the latter. Accordingly, only the resulting $TVMP_o$ is affected; the VMP^H curves are not depicted, yet can be inferred with the common VMP^F curve. To make a comparison possible, both equilibria are also based on the same wild bee populations (i.e., remaining crop-pollination requirements) and same opportunity costs w_c^* and w_o^*, where $w_c^* > w_o^*$.

In equilibrium ***7.a)***, it becomes clear that commercial beekeeping is preferred over on-farm beekeeping: the aggregated $TVMP$ of on-farm beekeeping is much lower than the aggregated marginal benefits from commercial beekeeping (corresponding to I < II in Figure 2.6). The farmer would thus be better off hiring commercial beekeeping and retaining a surplus equivalent to area II (i.e., provided she has sufficient bargaining power to keep all the surplus II).

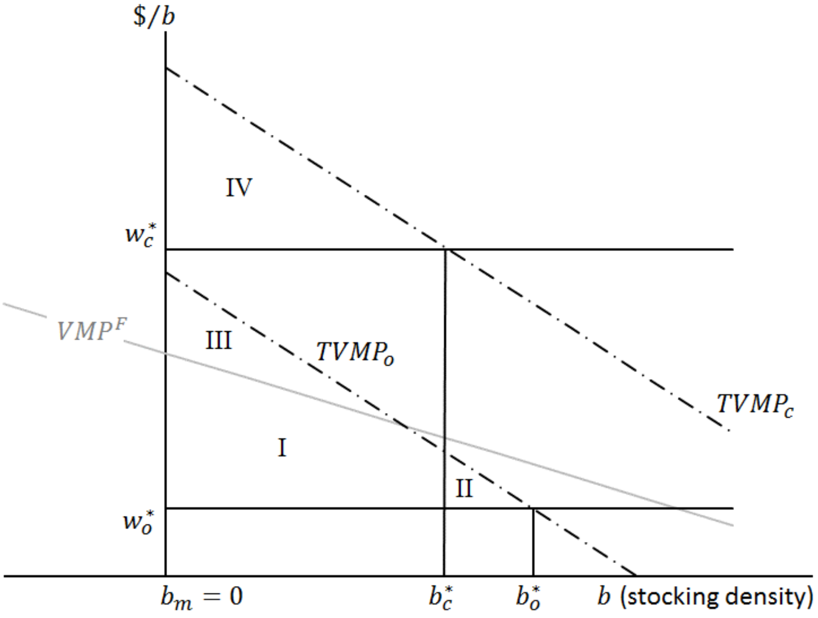


Hiring crop-pollination services from commercial beekeepers is preferred over on-farm beekeeping: the surplus of the former exceeds that of the latter, i.e., II > I.

Figure 2.6. On-farm vs. commercial beekeeping: the alternatives' characteristic opportunity costs ($\boldsymbol{w_o^*}$ for the former and $\boldsymbol{w_c^*}$ for the latter), lead to the corresponding equilibria, i.e., $\boldsymbol{b_o^*}$ and $\boldsymbol{b_c^*}$, respectively. The total value of marginal products ($\boldsymbol{TVMP}$) equals the sum of values of marginal product of fruit ($\boldsymbol{VMP^F}$) and of honey ($\boldsymbol{VMP^H}$). The $\boldsymbol{VMP^H}$ for on-farm and commercial beekeeping (here not depicted for simplicity in visual representation), only differ in the corresponding marginal costs of beekeeping $\boldsymbol{k}$, i.e., $\boldsymbol{k_o > k_c}$. $\boldsymbol{VMP_o^H - k_o}$ and $\boldsymbol{VMP_c^H}$ can be inferred by subtracting $\boldsymbol{VMP^F}$ from their corresponding $\boldsymbol{TVMP}$s. Per-colony bee wages ($\boldsymbol{w_c}$) consist of a pollination fee and an in-kind honey payment; $\boldsymbol{w_o}$ represents the per-colony forgone labor wage of the beekeeping farmer.

Equilibrium ***7.b)*** (Figure 2.7) reverses the relative positions of on-farm and commercial beekeeping, as the former delivers positive benefits from honey production that result in a surplus equivalent to area I+II+III, which is greater than the surplus of the latter, i.e., area IV.

It is conceivable that farmers will engage in modern beekeeping (i.e., with *A. mellifera*) as an additional farming activity, i.e., at a small scale or as a hobby (thus accounting for relatively small or even disregarding opportunity costs of labor w_o). For a smallholder farmer nonetheless, especially in regions where *A. mellifera* is a foreign species, the investments k_o required to adopt and run modern beekeeping, even at a small scale, may be prohibitive. On the other hand, traditional beekeeping with native bees (if such are available), typically requires marginal infrastructure, little capital and easy-to-learn skills, and can thus be an affordable alternative for smallholders (especially in developing countries) to derive greater value from the local floral resources and bee fauna (Narjes and Lippert 2010; Gupta et al. 2014; Kasangaki et al. 2014).



On-farm beekeeping is preferred over hiring crop-pollination services from commercial beekeepers: the surplus of the former exceeds that of the latter, i.e., I+II+III > IV.

Figure 2.7. On-farm vs. commercial beekeeping: the alternatives' characteristic opportunity costs ($\boldsymbol{w_o^*}$ for the former and $\boldsymbol{w_c^*}$ for the latter), lead to the corresponding equilibria, i.e., $\boldsymbol{b_o^*}$ and $\boldsymbol{b_c^*}$, respectively. The total value of marginal products ($\boldsymbol{TVMP}$) equals the sum of values of marginal product of fruit ($\boldsymbol{VMP^F}$) and of honey ($\boldsymbol{VMP^H}$). The $\boldsymbol{VMP^H}$ for on-farm and commercial beekeeping (here not depicted for simplicity in visual representation), only differ in the corresponding marginal costs of beekeeping $\boldsymbol{k}$, i.e., $\boldsymbol{k_o > k_c}$. $\boldsymbol{VMP_o^H - k_o}$ and $\boldsymbol{VMP_c^H}$ can be inferred by subtracting $\boldsymbol{VMP^F}$ from their corresponding $\boldsymbol{TVMP}$s. Per-colony bee wages ($\boldsymbol{w_c}$) consist of a pollination fee and an in-kind honey payment; $\boldsymbol{w_o}$ represents the per-colony forgone labor wage of the beekeeping farmer.

2.3 Applying the analytical framework to real world beekeeping-farming interactions

In the following, we associate the above presented equilibria with real world examples of interactions between beekeeping and farming activities that we obtained from a literature review and our own observations. Although based on anecdotal evidence, these associations and their classification according to our analytical framework may further guide the formulation of future testable empirical research questions. In this regard, giving context to the proposed theoretical equilibria may further hint at the agro-ecological and socio-economic conditions that favour the cooperation between beekeepers and farmers or the adoption of beekeeping by the latter.

We start with ***1.a)***, which we compare to the case of crops with overlapping flowering periods, especially if there are differences in the value of their corresponding marginal products of fruit and/or honey. This may result in the under-provision of pollination services for crops that compete for pollinators with crops that have comparatively higher total value marginal product. For instance, apples and strawberries, both members of the Rosacea family, are likely to overlap in their pollination faunas. Both crops are cultivated in the Mid-Atlantic region of the USA, where Grab et al. (2017) determined a negative effect on strawberry yields when this crop's bloom coincided with the mass flowering of apples during early and peak bloom periods. Grab et al. (2017) also report that *A. mellifera* beekeepers often bring their colonies to pollinate apples but not strawberries. Similarly in California, avocadoes and early blooming cherries compete for honeybee pollination with almond orchards (Sumner and Boriss 2006); such competition is especially relevant for avocadoes, whose nectar and honey are relatively unattractive to honeybees (i.e., low VMP^H curve) when other foraging sources are available, thus often leading to an insufficient pollination (Ish-Am and Eiskowitch 1998; Akik et al. 2006). With the acreage expansion of Californian almonds, the demand for pollination services has exceeded the supply of honeybee colonies (Sumner and Boriss 2006; Pilati and Prestamburgo 2016), resulting in almond pollination fees (167.0 USD/colony) that in 2016 were far superior to those paid for cherries and avocadoes (68.6 USD/colony and 40.8 USD/colony, respectively) (United States Department of Agriculture 2016c); for avocados this means a high w^*. Although, to our knowledge, a decline in avocado and cherry yields has not been reported in connection with uncompetitive pollination fees or honey revenues, it is to be expected that migratory commercial beekeepers will respond to such price signals and corresponding opportunity costs by deploying as many bee colonies on almond orchards. In the absence of wild pollinators this may result in reduced avocado and cherry yields.

As described under ***1.b)***, transaction costs may also discourage the cooperation between beekeepers and farmers. This could be the case for the Lake Constance fruit producing region in Southern Germany, where small-scale agriculture is the dominant land use and, according to Wolf (2016), beekeepers are mostly hobbyists with no more than 30 colonies. Wolf (2016) further reports that the first contact between farmers and beekeepers in this region is typically not facilitated by intermediaries, and pollination contracts are usually informal (e.g., verbal agreements). Transaction costs may thus arise from establishing a first contact, from contract enforcement (especially concerning the risk of pesticide exposure to bees) and from the uncertainty regarding the demand for pollination services and the corresponding fees. In this regard, some beekeepers refused pursuing a contact to offer farmers their pollination services, arguing that the pollinator density of this region is high and that it is therefore difficult to establish a pollination fee based on their bees' marginal contribution to crop yields (VMP^F being already quite low at $b = 0$; cf. Figure 2.5). Moreover, some beekeepers stated that they offer honey as an in-kind compensation to the farmer. Notwithstanding these accounts, a pollination demand seems to exist in some parts of this region: large-scale migratory beekeepers (>300 colonies) travel to Lake Constance every year to offer the pollination services that, due to a lack of mobility, many local beekeepers cannot deliver. The average pollination fee paid by farmers is €21/colony (*SD*=€21/colony)[35]. This price may nevertheless indicate a low marginal productivity of managed honeybee pollination, relative to the average per-colony pollination fee (€47; *SD*=€4) that is paid in the *Altes Land*, Germany's largest coherent fruit producing region. The pollination fees in one of Lake Constance's districts may also be influenced by a local beekeepers' association whose members, motivated by the ideal of a non-monetary cooperation with farmers, deliberately renounce to a remuneration for their pollination services as a measure against the formation of a pollination market (ibid).

At any rate, the marginal productivity of honeybee-mediated crop pollination seems to be heterogeneous over Lake Constance's fruit producing subregions, as suggested by the pollination fees' standard deviation. Furthermore, the proportion of honeybee hives' density to pollinator-dependent orchards' density varies between Lake Constance's subregions (Dabbert et al. 2017), thus hinting at a heterogeneous provision of managed pollination services. In view of this, some subregions are likelier to be described by equilibrium ***5)***, where rather than hindering a cooperation between farmers and beekeepers altogether, transaction costs lead to a cooperation that is only based on honey benefits and render the concomitant pollination services an externality (cf. Figure 2.4). After all, beekeepers depend on floral resources from different orchards to at least guarantee the subsistence of their bee colonies throughout the year, thus setting a basis on honey benefits for a beekeeper-farmer cooperation. On the other hand, some subregions of Lake Constance may present an incipient pollination market resembling equilibrium ***2)*** (cf. Figure 2.2).

Equilibrium ***2)*** may also describe the *Altes Land* region, where a pollination market started to emerge in the 1940s, two decades after the introduction of synthetic pesticides entailed an important reduction of pollinators in this region's sweet cherry orchards (Hausschildt 2003; Reise 2005). A centralized organization for pollination services (the only one in Germany) has been established there (since the *Esteburg Fruit-Growing Centre* was founded in Jork in 1929) to procure from migratory beekeepers of other regions the additional 3,000 to 4,000 honeybee colonies that are required for the pollination of local orchards (Dabbert et al. 2017).

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[35] These figures were calculated over all surveyed contracts in this region (*N*=10), of which 50% involved a pollination fee of €0/colony. Neglecting the zero values, the average pollination fee changes to €41/colony (*SD* = €5/colony).

A situation where beekeepers pay orchadists, i.e., equilibrium ***3)*** (corresponding to IV > III in Figure 2.3), may partly explain the case of the Chiang Mai-Lamphun valley lowlands (Northern Thailand), where a large *A. mellifera* honey industry that competes for exclusive rights to the floral resources of longan (*Dimocarpus longan*) orchards has been established. The heterogeneous remuneration to longan farmers (cf. Sections 2.1.1 and 2.1.2) could nevertheless also reflect regional differences in the marginal productivity of honeybee hives: it is for instance conceivable that orchadists who have perceived appreciable yield improvements from managed *A. mellifera* pollination would be willing to accept a lower remuneration from beekeepers than those farmers whose orchards yield no further benefits from crop pollination. One could thus further hypothesize that instances approaching equilibrium ***4)*** (corresponding to IV ≈ III in Figure 2.3) are also represented in the Chiang Mai case: some farmers and beekeepers may decide to cooperate without involving a payment. Equilibrium ***4)*** is instead less pertinent to Lake Constance, as most beekeepers there are hobbyists and thus less likely to incur important opportunity costs.

Thailand also offers examples for the proposed model extensions that lead to equilibria ***6)*** and ***7)***. Chiang Mai's presumably abundant wild bee endowment can for instance be taken into account as an important contributor to the local production of fruit: longan's high demand for bee-mediated pollination seems to be, in general, adequately covered by wild bees, yet many orchards may also be delivering additional benefits attributable to managed honeybees. In fact, 81% of the interviewed longan farmers that allow honeybee colonies being placed in their orchards stated having experienced improved yields due to pollination services from managed honeybees; for obvious reasons, they could not specify the magnitude of such contribution and instead provided a subjective account, i.e., reporting either a moderate (67%), a great (14%), or no (19%) yield improvement (calculations with data from our research project). These figures suggest that, in most reported cases in Chiang Mai, wild bees delivered the bulk of crop pollination, relegating a remainder of relatively low or negligible marginal benefits from this task to managed honeybees.

Our model accounts for a possible direct influence of local wild bee populations (*ceteris paribus*) on the productivity of managed honeybees, which can thus partly explain differences in the magnitude and direction of payments between farmers and beekeepers within and between regions. In this respect, the pollination deficits that reportedly affected the productivity of orchards in Chanthaburi province (Eastern Thailand) can be represented by setting the equivalent population of locally available pollinating wild bees (b_w^e) closer to the origin $b = 0$ (cf. Figure 2.5), thus making the supplement of crop-pollination with managed honeybees more relevant. Under this circumstance, many Chanthaburi farmers were confronted with the choice of either renting *A. mellifera* colonies from professional beekeepers, or adopting on-farm beekeeping themselves. Ultimately, 55% of the interviewed farmers who blamed past yield declines on wild bee pollination deficits decided to supplement their orchards' insufficient pollination by adopting meliponiculture (calculation with data from our research project), a practice that consists of keeping and propagating stingless bee colonies that have typically been first collected in the wild. This choice may reflect the rationale of equilibrium ***7.b)*** (cf. b_o^* in Figure 2.7), especially considering that, relative to *A. mellifera* beekeeping, meliponiculture entails low marginal production and opportunity costs (cf. Section 2.1.2). On the contrary, ***7.a)*** (cf. b_c^* in Figure 2.6) may rather represent the case of the *Altes Land* region in Northern Germany and the Californian almond groves in the USA. In both cases, the opportunity costs of on-farm beekeeping, at a scale that covers an orchard's pollination demands, are evidently too high.

2.4 Discussion and conclusions

The above presented equilibria result from a theoretical model that rests on the fundamental assumptions of neoclassical economics and perfect competition. This basic principle shall be subject to scrutiny. We have already included a qualitative discussion on the possible effects of transaction costs (cf. Sections 2.2.2.1.b and 2.2.2.5), which are otherwise assumed to be zero. Evidently, the assumptions of perfect information (at zero cost) and profit maximization are often unrealistic: regarding the latter, the fact that many individuals engage in beekeeping as hobbyists or idealists casts further doubt on its plausibility. Moreover, being that the production factors for honey and crop pollination (i.e., labor, land and bees) are supplied by both transacting parties (and that wild pollinators may also contribute to crop pollination) imposes a challenge on the assignation and exchange of property rights; this for instance, seems to have hampered the establishment of pollination fees in some districts of the Lake Constance fruit producing region (cf. Section 2.3).

For our model extensions we assume that the marginal benefits of managed bees set their baseline where wild bees have finished delivering their benefits (cf. Section 2.2.2.6). It is worth mentioning that this assumption undermines the fact that most introduced bee species (especially those with polylectic diet[36]) potentially compete with (and may marginalize) a great variety of native bee species, a competition that is especially asymmetric in favour of managed honeybees, as these are often given supplementary feeds when floral resources are scarce (Goulson 2003). The marginal benefits baseline for managed bees b_w^e (cf. Figure 2.5) may thus move closer to the origin as managed bees displace wild bees. Kleczkowski et al. (2017) further point out that the availability of commercial bees can mask a decrease in pollination services from wild pollinators, possibly past a local extinction threshold, which would otherwise encourage farmers to adopt conservation measures.

Notwithstanding these caveats, the basic model by Rucker et al. (2012), in combination with our proposed extensions, may help better classifying and analyzing farmer-beekeeper interactions of particular agroecosystems and social contexts as we develop a more encompassing array of possible bioeconomic

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[36] A polylectic diet in bees (e.g., *A. mellifera*) consists of floral resources from many plant families and genera.

equilibria. This analytical framework provides a basis to identify the causes for a possible crop pollination under-provision and for hypothesis testing. For instance, associating a bee-mediated pollination dependent crop with an equilibrium where colony stocking density b^* equals zero could hint at either relatively high transaction costs (that are prohibitive for a beekeeper-farmer cooperation), or at a healthy provision of pollination services from wild pollinators (i.e., low $TVMP$ from managed bees vis-à-vis high opportunity costs of commercial beekeeping w^*, cf. Section 2.2.2.1).

As a diagnostic tool, the proposed analytical framework can also inform the formulation of policies. Among the possible interventions, two are adequate for the following exemplary cases (identified according to Section 2.3):

i) Given an important (already existing) crop pollination deficit (such as that which occurred in Chanthaburi, Eastern Thailand), to prepare for urgent corrective action by improving the institutions and information flow that facilitate the interaction between farmers and beekeepers (i.e., disburdening them from transaction costs), or which may help farmers adopting on-farm beekeeping.

ii) Confronted with eventual wild pollinator declines due to intensified agriculture (as conceivable in Chiang Mai, Northern Thailand), a sensible preventive intervention could be implementing policies for the assessment and monitoring of bee declines and to support the adoption of adaptive management techniques that enhance pollinator conservation and habitat restoration (Byrne and Fitzpatrick 2009). In this regard, Isaacs et al. (2017) recently introduced a comprehensive pollinator management concept that they coined “Integrated Crop Pollination”, which among other strategies recommends developing management protocols for native bee species and incorporates considerations by Biddinger and Rajotte (2015) regarding adjustments to Integrated Pest Management programs to protect pollinator health (i.e., both managed and wild pollinators), without compromising pest control.

As a decision-supporting tool, our analytical framework is also potentially useful for comparing the benefits from hiring commercial crop pollination services with those from adopting on-farm beekeeping with the locally available domesticable bee species (cf. Section 2.2.2.7). In practice, this requires collecting information on the marginal productivity of the relevant bee species and for the crops in question, which may also frequently involve increasing the stock of pertinent knowledge with more meticulous and context-specific trials such as those offered by Ricketts and Lonsdorf (2013) for coffee in Costa Rica and by Blanche et al. (2006) for longan in Australia.

Our analyses are founded on direct and indirect benefits provided by bees to humans through their hive products and pollination services respectively. Nevertheless, as noted by Kleijn et al. (2015), the delivery of crop pollination services is an insufficient argument for the conservation of wild pollinators as it discriminates in favor of a few species of dominant crop pollinators, while neglecting in the conservation efforts the threatened species that are rarely observed near crops. Bearing in mind that human preferences are not necessarily in keeping with the ecologic balance, the question arises as to which other benefits (corresponding to e.g., option or existence values)[37] might be neglected when optimizing the allocation of pollinating bees based solely on their honey and crop pollination services.

The cases cited and reported in this study illustrate the global diversity of beekeeper-farmer interactions, which reflect how these actors adapt to the agro-ecological, socioeconomic and legal conditions that are peculiar to each region. So far, the economic literature has focused on describing and modeling well established crop pollination markets (such as in the Californian almond groves, where for instance beekeepers paying farmers, i.e., for exclusive rights to floral resources, is an exception), thus neglecting the possibility of other bioeconomic equilibria. Our extensions to the model by Rucker et al (2012) offer a broader analytical framework that accommodates the diverse circumstances under which bee-mediated pollination and apiculture may contribute to a region's agricultural production.

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[37] Narjes and Lippert (2016, Chapter 3) attempted at estimating the value of conserving native pollinating bees (thus partly capturing non-use value components), by letting longan farmers of Northern Thailand state their preferences for several bee conservation measures (and their potential effects on the population of native bees) within a discrete choice experiment.

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Chapter III

Longan fruit farmers' demand for policies aimed at conserving native pollinating bees in Northern Thailand[38]

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Keywords: Northern Thailand, Native bees, Crop pollination, Choice experiment, Dimocarpus longan, Valuation.

Abstract

Northern Thailand is orienting its agriculture towards intensive production systems at risk of being subjected to the current worldwide pollinator crisis. Bee-friendly pest management, improving native bee habitats within agro-forest ecosystems and fostering the husbandry of native bee species are three widely recognized strategies to conserve the local pollinating fauna. We attempted at eliciting farmers' valuation of these measures and that of their potential effect on local native bees, by conducting a choice experiment with 198 longan (*Dimocarpus longan*) farmers. The results of a mixed logit model indicate a significant heterogeneity in farmers' preferences, part of which was explained by the respondents' attitude towards native bees, among other idiosyncratic variables such as gender. We also determined a generally positive willingness to pay for the above mentioned conservation measures, which implemented together were valued at approx. €18.1 by the average household, all else equal. Additionally, avoiding a 50% native bee population decline was valued in average at €40.5 per household. These estimates stand in strong contrast with the comparatively high economic losses such a decline could potentially entail in terms of reduced longan production and the relatively low investment costs to implement a conservation strategy aimed at preventing such losses.

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3.1 Introduction

The international community is showing increasing concerns regarding the continued decline of both wild and managed pollinator populations worldwide (Dias et al. 1999; Ricketts 2004; Steffan-Dewenter et al. 2005; Kluser & Peduzzi 2007; FAO 2008; Gallai et al. 2009; Potts et al. 2010). Agricultural intensification has been recognized as the main driver for the decline of wild bee populations, especially due to the inappropriate use of pesticides and by reducing natural habitats through land-use change (Kremen et al. 2002; Potts et al. 2010).

Thailand is located in a bee diversity hotspot. With the exception of the European honeybee (*Apis mellifera* L), all other 8 honeybee species of the world are indigenous to Southeast Asia (Hepburn & Radloff 2011). There is also a great diversity of stingless bees in this subcontinent, with a large number of species recorded in Thailand, particularly in its northern provinces (Rajitparinya et al. 2001; Klakasikorn et al. 2005; Jongjitvimol et al. 2005). The region has therefore historically been a cluster for traditional beekeeping, which is mainly practiced by smallholders with rather rudimentary technologies that have been developed around the culture of the Asian honeybee (*Apis cerana* F.) and that of stingless bees.

Northern Thailand is also rapidly orienting its agriculture to the production of high-value crops under intensive systems that are often characterized by the overuse of synthetic pesticides (Schreinemachers et al. 2011), which in connection with deforestation (Delang 2002) risk reproducing the case of other regions in the world, where intensive agriculture has driven pollinator populations to substantial declines (Biesmeijer et al. 2006; National Research Council 2007; Potts et al. 2010). Thailand has also responded to the continuously growing demand for longan (*Dimocarpus longan* L.), a fruit obtained from a bee-pollination dependent crop (Blanche et al. 2006; Pham 2012), by dramatically expanding its cultivated area and its yields, i.e. from 12,094 ha (corresponding to 45,756 tons per annum) in 1983 to 168,517 ha (i.e. 976,729 tons per annum) in 2014 (Anupunt & Sukhivibul 2005; Thai Office of Agricultural Economics 2014). Currently, ~82% of the longan land is cultivated by 206,328 households in Northern Thailand, ~30% thereof by 69,330 households in Chiang Mai province (Thai Office of Agricultural Economics 2014), rendering this region the leading exporter of longan worldwide and its economy highly dependent on this crop (Anupunt & Sukhivibul 2005; Menzel & Waite 2005).

Although there are yet no official reports on a pollinator crisis in Thailand, in June 2011 we collected anecdotal evidence from the eastern Thai province of Chanthaburi that supports the suspicion that a pollinator problem might exist, in at least that region: according to accounts from many local rambutan

(*Nephelium lappaceum* L.) farmers, habitat encroachment due to agriculture and pesticide overuse have dramatically reduced the population of wild bees with economically important consequences on their crop yields. In response, a local initiative is correcting such pollination deficiencies by promoting on-farm meliponiculture (i.e. keeping stingless bees).

Against this background, Northern Thailand could benefit from a policy directed at conserving native pollinating bees. Such policy should take into account the perceptions of longan farmers with regards to the benefits of its implementation and the expected yield losses that could arise in the event of an important decline of pollination services. A pollinator conservation policy could consist of following measures: *i)* offering farmers bee-friendly alternatives to conventional pesticides (e.g., biological control and integrated pest management), *ii)* encouraging the protection and improvement of natural bee habitats within agro-forest ecosystems and *iii)* fostering the husbandry of native bee species. Expert interviews and focus group discussions with farmers helped us recognize that, among the recommendations of the Plan of Action of the International Pollinator Initiative (IPI-POA) (Byrne & Fitzpatrick 2009, cf. Section 1.3), these measures potentially have the greatest impact and implementation chances in Thailand’s current agricultural and political context.

We conducted a discrete choice experiment (DCE) in Chiang Mai Province in order to understand the preferences of longan farmers with regards to the conservation strategies mentioned above and to hypothetical changes in the local population of native bees. The respondents had to choose between a status quo scenario, associated with an assumed 50% native bee population decline, and a series of alternative hypothetical policy scenarios in which the implementation of different conservation strategy combinations would avoid such declines. The choice decision also involved a single-payment tax hypothetically incured by the respondents before any conservation policy bundle could be implemented. Our analyses include by this design willingness to pay (WTP) estimates for the individual conservation policy attributes and for the bee population declines assumed in the DCE. Furthermore, we confront the value estimates obtained with the potential costs that would arise if some or all of the ecosystem services provided by local native bees to longan orchards would be lost, as calculated based on the bee-pollination dependence ratios given by Blanche et al. (2006) and Pham (2012) (cf. Section 3.2).

3.2 The economic value of pollination services

In several studies, the economic value of the contribution of pollinators to agricultural production has been estimated using a dependence ratio that accounts for the partial production loss of specific crops, attributed to the complete absence of pollinators (Morse & Calderone 2000; Losey & Vaughan 2006; Gallai et al. 2009). Gallai et al. (2009) for instance estimated the total economic value of pollination services worldwide at €153 billion. Building upon this approach and having estimated the demand functions of a variety of insect-pollination dependent crops, the potential welfare losses from increases of food prices that would result from the effect of insect pollination shortages on crop yields can be considered (Kevan & Phillips 2001). Accordingly, Southwick and Southwick Jr. (1992) estimated the annual value of crop pollination by managed honeybees (*A. mellifera*) in the USA to range between USD1.6 and USD5.7 billion.

Pollination experiments along replicated distance gradients have also been used to estimate the economic value of tropical forest patches that, serving as nesting sites for bees, contribute to the pollination of crops, such as coffee (*Coffea arabica* L. and *Coffea robusta* P.) (Ricketts 2004; Olschewski et al. 2006). Blanche et al. (2006) conducted similar experiments with longan (*Dimocarpus longan*) orchards in north Queensland, Australia, where they assessed the effect of their proximity to rainforests (as beneficial insect reservoirs) on this crop's pollination. No monetary results were offered by this study, yet it concludes that initial fruit set in longan is substantially enhanced by insect pollination (i.e. 62% contribution), prominently from stingless bees. A similar result obtained by Pham (2012) for four different longan cultivars in Quoc Oai, Vietnam attributes 67% of their yields to floral visits by Asian honeybees (*A. cerana*), amounting to EUR 0.34 per kg of fruit in 2011.

Other studies have measured the economic value of pollination services by directly observing the market prices of existing commercial pollination services that are contracted by farmers to substitute their failing ecosystem service counterpart, such as it occurs in the almond groves of California, USA (Rucker et al. 2012). Another approach consists in calculating the cost of potentially having to replace pollination services with labor or capital (e.g., hand pollination, or pollen dusting, respectively), such as to maintain crop production at the same levels that are attained with pollination services from a healthy natural ecosystem (Allsopp et al. 2008).

More recent studies have integrated the estimation of economic values for pollination services with spatial analyses. Ricketts and Lonsdorf (2013), for instance, calculated (discrete) marginal values for unit changes in pollinator habitats by combining the pollen limitation experiment results for coffee fields in Costa Rica from Ricketts et al. (2004) with a model by Lonsdorf et al. (2009) that predicts the supply of pollinators based on the surrounding land cover's suitability to provide nesting sites and floral resources. On the other hand, Barfield et al. (2015) and Lautenbach et al. (2012) applied the pollination dependence ratio and crop vulnerability ratio approaches to plot economic value estimates at local and global scales, respectively; the former using a farm gate dataset for 55 crops in the US state of Georgia, while the latter combined FAO country-specific data for the years 1993 through 2009 with the global crop distribution maps of Monfreda, Ramankutty and Foley (2008).

3.3 Material and methods

3.3.1 The discrete choice experiment

The studies reported above (Section 3.2) estimate the so-called use value of pollination services relying upon market price observations of either pollination dependent crops or commercial pollination services. In contrast to such studies, DCEs have been deemed not suitable for the estimation of the economic benefits of pollination services, with the sensible argument that such stated preference methods would require respondents to possess a sound knowledge of the quantitative contribution that pollination delivers to their agricultural production (Mburu et al. 2006), i.e. a lack in ecological knowledge may hinder them from correctly assessing the use value of pollination. We do not dispute such argument, nor do we consider DCEs an alternative to studies that estimate the market value of pollination services. On the contrary, we think both approaches can complement each other: market-based valuation methods are important tools to estimate the use value of pollination, whereas DCEs can be used to assess peoples' current preferences for measures to conserve bees and for avoiding their declines. After all, policy makers should take into account stakeholders' preferences for the implementation and implications of the conservation policies considered in order to ensure some degree of public support. Thus, similar to the studies concerning wild geese conservation by Hanley et al. (2003) and compensatory wetland mitigation by Bauer et al. (2004), we propose approaching the economic valuation of pollination services from a perspective of *public demand for policies* aiming at conserving the native bees that deliver this ecosystem service in agro-forest landscapes. To this effect, the trade-offs that are stimulated in a DCE can capture the economic value of *measures* to conserve native bees. Furthermore, DCEs can also capture the existence value of pollinators and the option value of preserving them, disregarding the awareness that respondents may or may not have about how much pollinating bees contribute to the production of their crops. In this sense, one must be careful when interpreting DCE value estimates for changes in the population of native bees; these encompass several components of the total economic value of pollinating bees, contingent to socio-demographic characteristics of the respondents, and must not necessarily be equivalent to the true use values realized at specific levels of bee abundance.

Hanley et al. (2015) insist in the importance of capturing non-market benefits when estimating the economic value of pollinators and that this may only be approached by means of stated preference methods such as the DCE. To our knowledge, this is the first study to apply the DCE method to obtain economic value estimates for the conservation of pollinating bees.

3.3.2 Economic theory of discrete choice modelling

According to random utility theory (RUT) (Thurstone 1927; Marschak 1960), human choice can be explained by the utility maximizing behavior of individuals when they are confronted with paired or multiple comparisons of discrete choice alternatives. Each alternative potentially yields a certain level of utility that is known to the decision-maker, but unknown to the researcher. From the researcher's perspective, the utility that an individual i derives from a choice alternative j ($U_{ij}, j = 1, \ldots, J$) can be decomposed into a systematic (explainable) component (V) and a stochastic (unexplainable) component (ε) that represents unobservable influences over the decision-maker's choice. This can be formalized as follows:

$$U_{ij} = V_{ij}(X_j) + \varepsilon_{ij} = \beta' X_j + \varepsilon_{ij} \,, \tag{3.1}$$

where X_j is a vector of observed variables that relate to the choice alternative and are weighted by parameters β to account for their relative contribution to an individual's utility (i.e. part-worth utilities). The decision-maker i chooses from a given set of J choices the alternative h that maximizes her utility, strictly holding that the utility associated with alternative h is superior to that of any other alternative j. The probability P_{ih} of this choice outcome can be expressed as follows:

$$P_{ih} = P[(U_{ih} > U_{ij})] = P[(V_{ih} - V_{ij}) > (\varepsilon_{ij} - \varepsilon_{ih})] \quad \forall\, j \neq h \tag{3.2}$$

Assuming independent and identically distributed (IID) extreme value distribution type I error terms ε_{ij}, the choice probabilities can be expressed as the standard logit model (Train 2009). The mixed (random parameter) logit (ML) model (Hensher & Greene 2003) is an extension of the standard logit model that allows for taste variation in the utility function with parameters β_i. As the researcher cannot observe individual parameters β_i, the (unconditional) choice probability P_{ih} is the expected value of the standard logit probability over all the possible values of β_i, weighted by the continuous mixing distribution $f(\beta)$, the functional form of which is specified by the researcher. In this study the density of all attribute parameters β is assumed to be normal. As such, the choice probability (2) is given by

$$P_{ih} = \int \frac{e^{\beta' X_h}}{\sum_{j=1}^{J} e^{\beta' X_j}} \phi(\beta|\theta) d\beta, \tag{3.3}$$

where θ collectively denotes the moments of the normal density, which are the parameters to be estimated. Normally distributed random parameters enter the model as follows

$$\beta_i = \beta + \delta' w_i + \sigma v_i, \;\; v_i \sim N(0,1) \,, \tag{3.4}$$

where β is the fixed population mean, w_i are (observed) individual-specific characteristics that induce heterogeneity around the mean, v_i is the individual (unobserved) specific heterogeneity and σ is the standard deviation of β_i around β. Some random coefficients may only present unobserved heterogeneity (homogeneous parameter means), in which cases the vector δ is set to zero. The introduction of additional stochastic elements through β_i in the utility function that may be correlated across alternatives and choice situations partially relaxes the restrictive IID assumption (Hensher & Greene 2003; Hensher et al. 2005).

The integral (3.3) does not have a closed form and the choice probabilities P_{ih} must therefore be approximated through computational simulation: for different moments θ, values of β are drawn from

$f(\beta)$, with which the values behind the integral sign (Eq. 3.3) are calculated. This process yields the simulated probability $\check{P}_{ih}$ as a weighted mean of the probabilities calculated from the different draws of β. The parameters of the distribution ϕ are optimized by iteratively inserting (for different parameters of θ) the resulting $\check{P}_{ih}$ into the log-likelihood function, yielding the maximum simulated likelihood estimator (MSLE).

The estimated coefficients β can be used to derive welfare measures, such as the change in the expected consumer surplus $E(CS_i)$, due to changes in the alternatives and/or choice set, as given by

$$\Delta E(CS_i) = -\frac{1}{\beta_c}\left[ln\left(\sum_{j=1}^{J^1} e^{V_{ij}^1}\right) - ln\left(\sum_{j=1}^{J^0} e^{V_{ij}^0}\right)\right], \tag{3.5}$$

where the log sums $ln\left(\sum_{j=1}^{J^t} e^{V_{ij}^t}\right)$ express the expected maximum utility at the initial situation $t = 0$ and after the changes at $t = 1$, and their difference is divided by the negative cost coefficient β_c (the marginal disutility of cost). The total change in consumer surplus in the population can be computed as the weighted sum of $\Delta E(CS_i)$, with the weights reflecting the share of individuals in the population who share the same representative utilities as the sampled individual. The $\Delta E(CS_i)$ is consistent with RUT and often referred to as the willingness to pay (WTP) for changes in a choice alternative (McConnell, 1995; Lancsar & Savage 2004; Train 2009). Furthermore, it collapses to its simplest form, namely the marginal WTP (implicit price) for an attribute k, given equal changes in such attribute in all alternatives, *ceteris paribus* (*c.p.*):

$$MWTP_k = -\frac{\beta_k}{\beta_c}, \tag{3.6}$$

where β_k is the estimated coefficient of the attribute of interest. Obtaining economically meaningful WTP estimates becomes complex when it involves analyzing the ratios of two random parameters. When choosing a normal distribution, also the issue arises of how to handle extremely high WTP estimates as β_c approaches zero. To solve this problem, β_c can be fixed in the model and point estimates of a normally distributed WTP obtained with mean $-\beta_k/\beta_c$ and standard deviation σ_k/β_c (Hensher et al. 2005; Train 2009).

3.3.3 Hypotheses underlying this study and experimental design

Unstructured interviews, with the experts that are listed below, helped in determining the general models We aim at explaining the choices made by longan farmers, regarding alternative policy profiles for the conservation of native bees. Accordingly, the alternative hypotheses stated in this study are:

H1: The presence of each of the three proposed bee conservation strategies has a positive contribution *c.p.* to the utility derived from the conservation policy alternatives that contain them. Similarly, an increase in the population of native bees increases *c.p.* the probability that a policy presenting this attribute level will be chosen, while the opposite is true for a decline.

H2: The preference for the attributes constituting the choice alternative profiles varies among the population of longan farmers. This heterogeneity should be reflected in parameter standard deviations that are significantly different from zero and in parameter means that may interact with socio-demographic characteristics of the respondents.

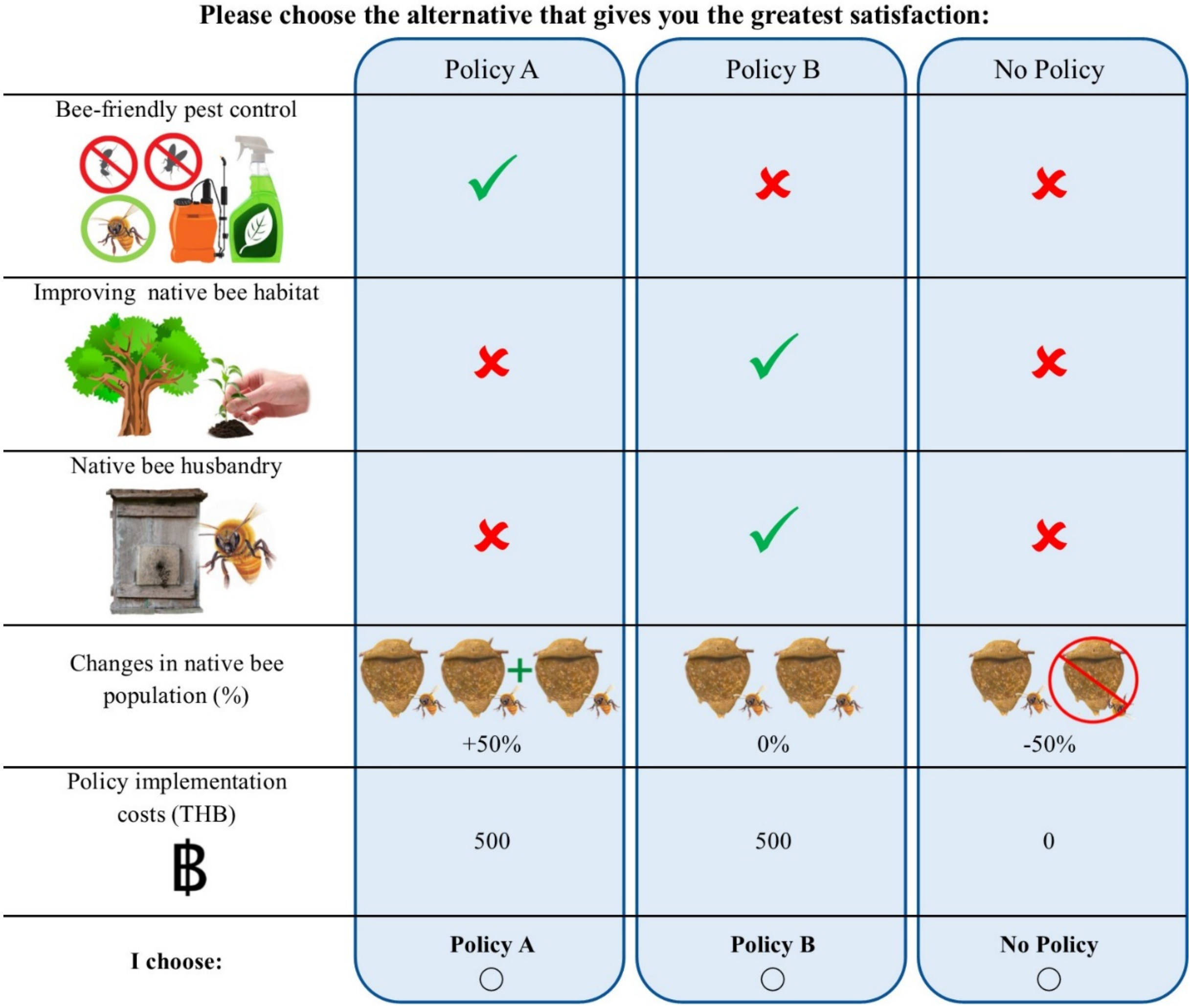
We defined the choice attributes with the assistance of provincial officers from the Thai Ministry of Natural Resources and Environment and from the Department of Agricultural Extension, who helped identifying the IPI-POA recommended conservation measures that could be implemented under the local political infrastructure. Focus group discussions with local longan farmers additionally contributed to formulating and phrasing plausible attribute levels that could be easily comprehended by the DCE participants. Consequently, we defined the attributes and levels (Table 3.1) as measures that would hypothetically be implemented at the village level and take effect with the support of extension services. With the implementation of a "bee-friendly pest control" program, the farmers would get information on methods (e.g., integrated pest management and spraying during times with low bee activity levels) and products that offer an alternative to conventional agro-chemicals, reducing the risk of bee poisoning. The "improving native bee habitat" measure would consist of the provision of expertise and native tree seedlings to promote local reforestation and habitat management campaigns in public lands and near cropland, aiming at offering nesting sites and food sources for native bees within agro-forest ecosystems. Extension services would also transfer technical knowledge on how to build bee hives to keep native bee species such as the Asian honeybee (*Apis cerana* F.) and stingless bee spp. (Heard 1999; Hepburn & Radloff 2011) on farm, under the "native bee husbandry" measure. The cost attribute represents a one-time fee that the farming households would pay to the local authorities for the implementation of the chosen policy alternative.

Table 3.1. Choice alternative attributes and corresponding design levels

Bee conservation policy attribute	*Levels*
Bee-friendly pest control	no [a)], yes
Improving native bee habitat	no [a)], yes
Native bee husbandry	no [a)], yes
Changes in native bee population (%)	−50 [a)], 0, +50
Policy implementation costs (THB) [b)]	0 [a)], 250, 500, 750

a) Attributes fixed at these levels for the status quo alternative.
b) €1 = 39.3048 Thai baht (THB), as of June 1, 2013.

A Bayesian efficient (Chaloner & Verdinelli 1995) subset of the full factorial design was generated using the Ngene 1.1.1 software. Efficient designs, in contrast with the traditionally preferred orthogonal designs, aim at data results that generate parameter estimates with as small as possible standard errors. Bayesian efficiency is achieved by pivoting the design around prior parameter estimates that are generated using data obtained from a pilot study (Rose et al. 2008; ChoiceMetrics Ltd. 2012). The prior parameter estimates used to generate the design for this study were based on a pilot study that we conducted with 27 respondents. Finally, we generated 12 choice sets, one of which is presented below (Figure 3.1).



Please choose the alternative that gives you the greatest satisfaction:

	Policy A	Policy B	No Policy
Bee-friendly pest control	✓	✗	✗
Improving native bee habitat	✗	✓	✗
Native bee husbandry	✗	✓	✗
Changes in native bee population (%)	+50%	0%	-50%
Policy implementation costs (THB) ฿	500	500	0
I choose:	**Policy A** ○	**Policy B** ○	**No Policy** ○

Figure 3.1. An example choice set.

At the beginning of each DCE, we asked the respondents to imagine a hypothetical scenario under which a conservation plan was not instituted, therefore leading the population of native bees to decline to half of its current population. This scenario was presented as the status-quo alternative "No Policy" and it did not entail policy implementation costs. Alternatively, the respondents had the option to choose one of two unlabeled policies (i.e. Policy A or Policy B, whose names are not meaningful sources of utility and which are only distinguishable by their attribute level combinations), containing *at least one* of the three proposed conservation measures, which if implemented could avoid a native bee population decline (0% change from the current population), or even increase it by 50%. Nevertheless, some of the policy implementation profiles also included the 50% native bee population decrease level. The levels describing the changes in abundance of native bees were defined with support of the focus group discussions and after pre-testing the questionnaire. Their three-level specification aims at reducing the respondents' cognitive burden and the design and sample size requirements for their estimation. The implementation of a conservation policy was always bound to a single-payment implementation cost ranging between THB250 and THB750. We randomized choice sets across questionnaires before administering them to the interviewees, in order to avoid biases from order effects.

In addition to the choice questions, the respondents were asked to provide information on their farm, socio-demographic characteristics and on their attitude towards the proposed native bee conservation measures. Previous to each interview, the respondents were informed about the importance of bee-mediated pollination for the fruit-set of longan and about the current trends and consequences of pollinator declines worldwide. This supporting information was complemented with text and illustrations that, similar to the choice cards, were conveyed in colored cards.

3.3.4 Survey and sampling

The DCE survey was conducted in May-June 2013, in 10 villages of the districts of Chom Thong and Saraphi, which are located along the Upper Ping River Basin, in the lowlands of the Chiang Mai–Lamphun valley (Figure 3.2). With 7,862 and 5,269 longan farming households (5,284 ha and 1,794 ha cultivated with longan) registered in the Thai Department of Agricultural Extension databases of Chom Thong and Saraphi respectively, these two districts amount the greatest extension of land cultivated with longan in Chiang Mai Province (DoAE 2015). We selected the villages randomly with the sampling technique of probability proportional to size, using the villages' total longan acreage as the allocation criterion. Thereby, six villages were drawn from Chom Thong, while the other four were drawn from Saraphi.

From 899 longan farming households registered under the 10 selected villages (DoAE 2015), a total of 198 randomly selected individual heads of household (with a total of 187.6 ha cultivated with longan) understood and completed the choice exercise. Each respondent faced twelve choices, resulting in 2,376 observations. A selection of variables that describe the sampled population is listed below (Table 3.2).

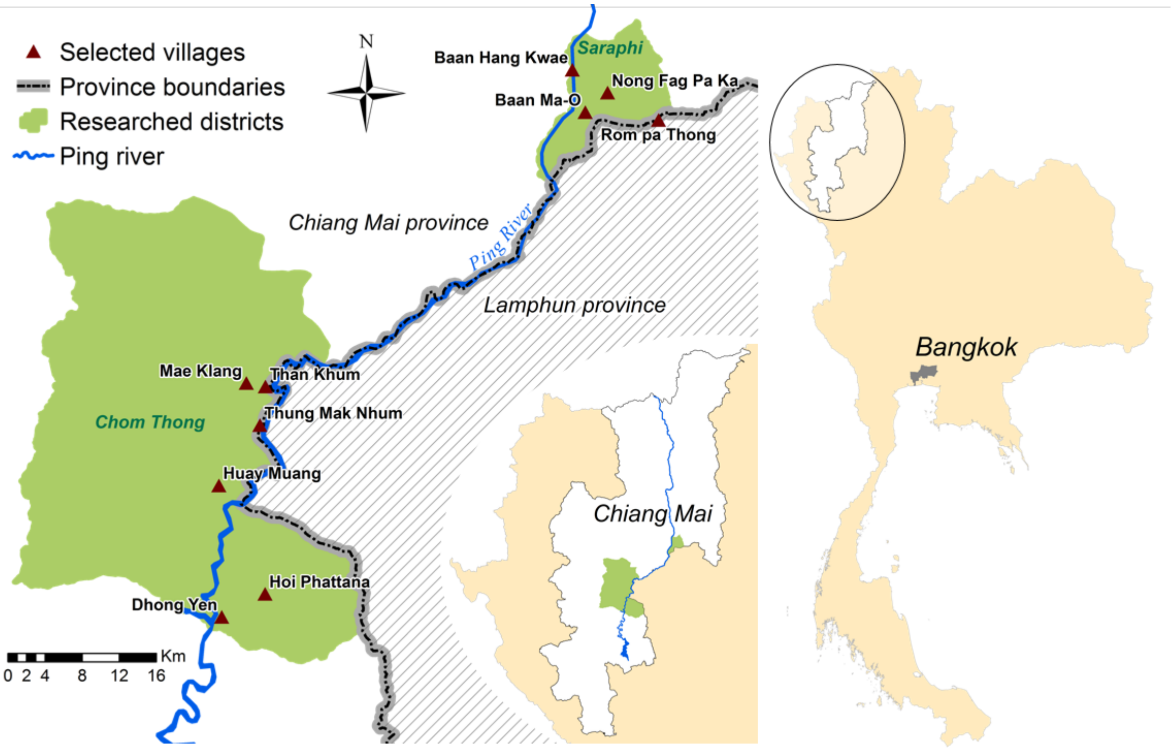


Figure 3.2. Research area: 10 villages along the lowlands of the Chiang Mai-Lamphun valley. Source: Own representation using vector data from the DCW and GADM databases (DIVA-GIS 2014).

Table 3.2. Descriptive statistics of the sampled population

Variable	*Mean*		*SD*
Age (years)	55.76		11.98
Total cultivated acreage (rai) a)	7.15		7.49
Longan cultivated acreage (rai) a)	5.92		6.09
Annual revenue from longan (THB) b)	91,119		105,080
Net annual agricultural income (THB) b)	76,415		96,822
Net total annual income (THB) b)			
Male (N_{male}= 116)	302,085*		836,880
Female (N_{female}= 82)	188,405*		191,008
	Sample shares (%)		
Regard longan as their main income source		68.18	
Completed six years of primary school only [or less]		71.71 [5.55]	
Self-employed in agriculture		85.35	
Aware of positive effect of native bees on longan yields		87.37	
Engage in at least one of the bee-related activities below:		50.00	
Allowing migratory beekeepers in their farm		28.79	
Beekeeping c)		17.68	
Honey hunting d)		20.71	
Individuals favoring [policy alternative]	45.58 [A]	43.52 [B]	10.90 [None]

N = 198 respondents = 2376 choice observations. Total longan cultivated area = 187.63 ha. a) 1 rai = 0.16 ha. b) €1 = 39.3048 Thai baht (THB), as of June 1, 2013. c) *A. mellifera* or native bees (i.e. *A. cerana* and/or stingless bee spp.). d) Harvesting honey from wild bees in the forest. Respondent's per household estimate. *Statistically significantly different: $F(1, 196) = 9.04$, $p = 0.003$.

3.4 Results

We analyzed the 2,376 choice observations using NLOGIT 5/LIMDEP 10 econometric software. All attributes were assigned generic coefficients (i.e. the utility specification was the same for the three choice alternatives). The three conservation measures entered the estimated models as dummy variables that take the values zero, if absent, and one if implemented. The cost attribute on the other hand was assigned a continuous variable, the coefficient of which relates changes in the utility to a cost increase of one Thai baht. We coded the native bee population change levels with two dummies representing a 50% increase and a 50% decrease respectively. We did not include alternative specific constants (ASCs) in our models due to the unlabeled nature of the presented policy alternatives. One could arguably regard the status-quo as a label and thus specify a constant term that is common for policy alternatives A and B, capturing possible preferences for any native bee conservation intervention being implemented over doing nothing. Nevertheless, in our study's design we included in each policy alternative *at least one* of the three proposed conservation measures in every choice set; modelling ASCs would thus result in their estimates being confounded with the effects of the single attributes.

We calculated two ML models, accounting for panel data structure and an assumed normal distribution for the random parameters: one with the full parameter vector set to be random and, in order to also find economically meaningful WTP estimates, a second model in which only the cost attribute was held fixed (Table 3.3) (Train 2009; Hensher et al. 2005). Following a stepwise approach we finally explained part of the heterogeneity in the random parameter means by interacting them with four idiosyncratic covariates; a dummy indicating whether or not the respondent is aware of the positive effect that native bees may have on crop yields interacted with both the bee-friendly pest control and improving native bee habitat attributes. Another dummy indicating whether or not bees (i.e. *A. mellifera* and/or native bees) are kept in the household interacted with the native bee husbandry attribute in the "fixed-cost model" only. The native bee husbandry attribute (in the "all-parameters-random model" only) and the 50% increase in the population of native bees interacted with a third dummy indicative of whether or not the household engages in at least one of three economic activities involving bees, i.e. beekeeping, harvesting honey from wild bees in the forest (honey hunting) and/or charging migratory beekeepers a fee for bringing honeybees (typically *A. mellifera*) to forage longan nectar on their farms. Lastly, a fourth dummy representing the respondent's gender interacted with the cost variable in the "all-parameters-random model" only. The simulations to approximate P_{ih} (Eq. 3.3) were done using 100 Halton draw sequences (Train 2000).

Table 3.3. Mixed logit coefficients for an all-parameters-random model and for a fixed-cost model with corresponding WTP estimates

		All-parameters-random model		*Fixed-cost model*		
Variable		*Coefficient* a)	*(SE)*	*Coefficient* a)	*(SE)*	*MWTP* e)
Bee-friendly pest control	*Mean*	0.48746	(0.3737)	−0.23226	(0.3635)	0.0 f)
	SD	1.39573***	(0.2046)	2.15956***	(0.1936)	629.4***
Improving native bee habitat	*Mean*	0.62168*	(0.3100)	0.09699	(0.2901)	0.0 f)
	SD	0.18831	(0.1786)	1.27936***	(0.1664)	372.9***
Native bee husbandry	*Mean*	0.60467***	(0.1798)	0.48032***	(0.1466)	140.0***
	SD	1.26305***	(0.1621)	1.35414***	(0.1475)	394.6***
−50% bee pop. (vs 0%)	*Mean*	−5.67399***	(0.4507)	−5.46216***	(0.4873)	−1591.9***
	SD	3.25209***	(0.3795)	3.19020***	(0.3658)	929.7***
+50% bee pop. (vs 0%)	*Mean*	2.83981***	(0.3304)	2.36089***	(0.2876)	688.1***
	SD	2.30053***	(0.3018)	1.82806***	(0.3603)	532.8***
Costs (THB)	*Mean*	−0.00540***	(0.0005)	−0.00343***	(0.0003)	−1.0
	SD	0.00369***	(0.0003)	Fixed parameter		
Heterogeneity in random parameter mean (Covariate × Attribute)						
Thinks native bees boost yields (yes)						
× Bee-friendly pest control		0.90528*	(0.3645)	1.12966**	(0.3530)	329.2**
× Improving native bee habitat		0.82310**	(0.3096)	0.99251***	(0.2854)	289.3***
Beekeeper (yes)						
× Native bee husbandry		Fixed at zero		0.60022**	(0.2145)	174.9**
Economic activity bees (yes)						
× Native bee husbandry		0.43869*	(0.2068)	Fixed at zero		
× 50% bee pop. increase (vs 0%)		1.37934***	(0.3760)	0.79309*	(0.3247)	231.1*
Male (yes)						
× Costs		0.00133**	(0.0005)	Fixed at zero		
Log-Likelihood (LL)		−1455.6685		−1513.8718		
AIC/*N* b)		1.2522		1.2945		
McFadden pseudo R-squared c)		0.3630		0.3375		
LRT c) d) χ^2 (*df*)		(30) 1659.0624***		(22) 1542.656***		

a) Significance levels: * $p < 0.05$, ** $p < 0.01$, *** $p < 0.001$. b) AIC = Akaike information criterion; $N = 2376$. c) Based on the LL function of a restricted model with two constants only, i.e. with choice probabilities set at each alternative's sample shares. d) Likelihood ratio test. e) Thai baht (THB): €1 = 39.3048 THB, as of June 1, 2013. f) MWTP of attributes with coefficient estimates not significantly different from zero were calculated at zero THB.

These models (Table 3.3) are statistically significant and their simulated probabilities correctly predict 66.41% of the observed choices, in the full random parameter vector model, and 66.25% in the model with fixed cost coefficient. Their corresponding AICs and pseudo R^2 indicate a good model fit.

The estimate for the bee-friendly pest control parameter mean came out non-significant in both models. The estimates corresponding to the heterogeneity around the mean (i.e. std. deviation and covariate) of this parameter on the other hand indicate a highly significant large spread in the respondent's value perception for this conservation measure and that being aware of the contribution of native bees to longan production has a significant positive effect on the utility it generates. A similar interpretation can be offered for the coefficients corresponding to the native bee habitat improvement measure in the fixed-cost model, which resulted in a non-significant parameter mean estimate with significant high estimates for the heterogeneity around the mean.

We obtained statistically significant estimates for the remaining parameter means and for the coefficients corresponding to the covariates and standard deviations in both models; the latter resulted smaller than in previous models that were estimated without interaction terms. Observing the assumption of normally distributed random parameters, we standardized the mean estimates $[(\beta + \delta' w_i)/\sigma]$ from the all-parameters-random model to obtain information on the share of respondents placing a positive value on the different attributes. Accordingly, the bee-friendly pest control measure generated a positive utility in 50% of the respondents who did not believe in native bees positively contributing to their crop yields, while this was true for 74% of those thinking otherwise. Similarly, the shares of respondents (presenting the corresponding covariate characteristics) who increased their utility with the improving native bee habitat and native bee husbandry measures and with a 50% increase in the population of native bees were 100%, 80% and 97% respectively. Only 4% of the entire sample expressed a positive preference for a 50% decrease in the population of native bees.

Considering the expected negative sign in the cost coefficient, we used the significant values of the fixed-cost model with confidence to calculate meaningful MWTP mean estimates along with their corresponding heterogeneity (Eq. 3.6). The mean MWTP estimates of the attributes with coefficient estimates that resulted not significantly different from zero were calculated at zero Thai baht. Thereby, each respondent's expected MWTP_{ik} was calculated (Eq. 3.4) and aggregated over the whole sample, using the representative shares for each covariate (Table 3.2) as weights, in order to obtain the average longan farming household's MWTP for each attribute k. We aggregated these values over the surveyed sample and the total population of the research area, and added them up to obtain average WTP estimates for a conservation policy bundle with all measures implemented (Table 3.4).

Table 3.4. Average and aggregated consumer surplus change estimates (THB) [a)]

Attribute	*Average household*	*Surveyed sample (N=198)*	*Total research area (N=13,131)* [b)]
−50% bee pop. (vs 0%)	−1,591.87	−315,190.17	−20,902,876.76
+50% bee pop. (vs 0%)	803.62	159,116.17	6,344,185.19
Bee-friendly pest control	287.66	56,955.77	3,777,201.21
Improving native bee habitat	252.73	50,040.87	3,318,617.97
Native bee husbandry	170.90	33,838.88	2,244,133.58
All measures implemented	711.29	140,835.51	9,339,952.77

Weighted with sample shares of idiosyncratic covariates (Table 3.2). [a)] €1 = 39.3048 Thai baht (THB), as of June 1, 2013. [b)] Number of registered households in Chom Thong and Saraphi districts (DoAE 2015).

To compare these WTP estimates with the potential costs of losing a proportion of the pollination services provided by local native bees, we calculated the resulting longan yield reductions (Table 3.5) using a 60% bee-pollination dependence ratio [slightly conservative as compared to the ratios provided in the literature (Blanche et al. 2006; Pham 2012)], an average production of 5.55 metric tons per hectare per year and a farm gate price of 29 THB per kg of longan in Chiang Mai Province, in 2013 (Thai Office of Agricultural Economics 2014). Accordingly, the economic value of longan production attributed to bee pollination can be estimated at 17.4 THB per kg (0.44 EUR per kg) and 96,570 THB per hectare (2,457 EUR per ha) per year.

Table 3.5. Estimated longan production losses $^{a)}$ attributed to declining bee pollination services (THB) $^{b)}$

Native bee population loss (%)	*Average farm (1.144 ha)*	*Surveyed sample (188 ha)*	*Total research area (7,078 ha)*
100.00	−110,476.08	−18,155,160.00	−683,522,460.00
50.00	−55,238.04	−9,077,580.00	−341,761,230.00
5.00	−5,523.80	−907,758.00	−34,176,123.00

a) Calculated with a 60% bee-pollination dependence ratio. b) €1 = 39.3048 Thai baht (THB), as of June 1, 2013.

Our calculations (Table 3.5) suggest that the actual forgone longan revenues incurred by the average farm from a 5% decline (assuming that natural pollination services are being provided at the ecologically necessary levels for a normal longan production) are much higher than our estimated WTP of an average household for avoiding a 50% native bee population decline and for the implementation of a policy that combines all three conservation measures; we also determined a great (but smaller) divergence between these values when aggregating them at the surveyed sample and the whole research area levels (Table 3.4). Nevertheless, one must be careful when interpreting the estimated WTP for a 50% native bee population decrease, as the abosulute value of this estimate, i.e. 1591.9 Thai baht, substantially exceeds the range of policy implementation costs that was presented to the respondents during the DCE (Table 3.1).

3.5 Discussion

We could reject the hypothesis of a zero mean preference (cf. *H1*) for all attributes, other than for the bee-friendly pest control (in both models) and the improving native bee habitat (in the fixed-cost model only) measures (Table 3.3); an indifferent taste perception for these two measures could not be dismissed with confidence in the 12.63% share of the population who did not believe in a positive contribution of native bees to their crop yields (Table 3.2). Furthermore, the substantial discrepancy between the estimated WTP to avoid a 50% native bee population decline and the actual costs that such loss would imply in terms of forgone longan yield revenues (Table 3.4 and Table 3.5) hints at the respondents' unawareness of the extent by which bees contribute to their crop production. These results accentuate the importance of education in the formation of value perceptions and thus that informing farmers about their crops dependence on pollination services could be seen as an effective policy measure to have them deploy more of their own resources in conserving this ecosystem service.

As suggested by the relevant interaction term coefficient, engaging farmers in economic activities involving native bees may also sensitize them for native bee population changes by attaching a direct use value to natural bee abundance (Table 3.3). Besides the obvious relation existing between being a beekeeper and an expected higher preference for participating in a native bee husbandry program, the significant result for this interaction may point out a demand for more sophisticated technologies and skills than those currently available to beekeepers. On-farm native husbandry is indeed considered a more sustainable alternative to honey hunting, as the latter activity (if not carefully carried out) may deplete wild bee colonies and damage their habitat (Partap, 2011). Moreover, on-farm beekeeping has encouraged orchard farmers in Chanthaburi province to use less pesticides on their crops as these would also harm their bees. The significant interaction of gender with costs, which suggests a lower reluctance to spending money *c.p.* by male respondents than by females, may relate to the significant income differences between male and female headed households (Table 3.2).

The resulting standard deviation estimates indicate a statistically significant preference heterogeneity among the population of longan farmers, leading to the rejection of the hypothesis of homogeneous taste parameters (cf. *H2*). The variance in the value perceived by the population for the “native bee husbandry” measure could, for instance, be related to the difference in opinions that members of some of the surveyed communities expressed (in the additional comments section of the DCE survey) regarding bee husbandry. Modern beekeeping with the European honeybee (*A. mellifera*) is widely practiced in this region due to the valuable honey that can be obtained from longan nectar. Beekeepers therefore practice migratory beekeeping (i.e. relocating the hives in search for new bee foraging sources). In some villages, the respondents expressed a negative opinion regarding modern bee husbandry, based on their belief that honeybees carry the parasites that serve as vectors for *witches' broom* (*Candidatus* Phytoplasma spp.), a disease that affects longan and lychee (*Litchi chinensis* S.) trees in the region. Some individuals additionally stated believing that bees harm their yields by eating their crop’s flowers. On the other hand, these negative opinions contrasted with a general interest in native beekeeping that could be especially determined in individuals from communities that had an already established tradition for this activity.

The parameter means of the “changes in native bee population” attribute suggest that avoiding a 50% *loss* of native bee populations is valued twice as much as an equally sized population *increase*, a result that is consistent with loss aversion behavior; a Wald test for linear restrictions led to the rejection of the hypothesis that the part-worth of these dummies could be captured as a single linear effect [$\chi^2(1) = 36.56$, p = 0.000]. In this regard, and considering the prospects of a local pollinator crisis such as we suspect might have stricken Chanthaburi province, avoiding the losses that could arise from even a small decline in the population of native bees (Table 3.5) justifies the comparably small investment in its prevention: the cost of implementing a conservation policy that includes all the proposed measures amounts to less than five thousand Thai baht per village, which includes an extension service officer’s salary (72% of total costs), contracted farmers’ wages, tree seedlings and transportation expenses. Every household interested in native beekeeping could additionally acquire bee boxes at 150 Thai baht each, an investment that can readily be amortized with the sales of honey (THB 80/kg) and other valuable hive products; moreover, distributed among all longan farming households in a village (an average of 1,130 in the research area), the costs of implementing the full project would lay far below the sum corresponding to the average household’s WTP for such purpose.

Placing the derived WTP estimates in the context of the average income earned by the sampled population of respondents (Table 3.2) also leads to the conclusion that these values conform to their expenditure capacity, especially due to the single-payment nature of the policy implementation costs.

3.6 Conclusions

The discrete choice experiment (DCE) approach allowed us obtaining economic value estimates for longan farmers’ preferences for conservation measures of wild pollinators and for preventing a decline in the pollinator population. Investigating the perspective of farmers is of utmost importance, as they would ultimately be the most directly concerned stakeholders regarding conservation policies. The results of our study thus inform Thai policy makers about which conservation strategies require a greater government intervention and which ones can be expected to engage more efforts and resources from the targeted farming communities. In this regard, we estimated that, on average, the share of longan farmers who did not believe in the positive contribution of native bees to their crop production valued both the bee-friendly pest control and the improving native bee habitat strategies at zero THB. In contrast, an on average considerably higher willingness to pay (WTP) for these strategies could be determined for those farmers who were aware of the importance of bee pollination to their yields. Most of the surveyed longan farmers were willing to pay for the implementation of the native bee husbandry conservation strategy and an even higher bid was estimated for those individuals, who already engaged in beekeeping (or in other economic activities involving bees).

We show that longan farmers very likely underestimate the true use value of pollination, when comparing their aggregated WTP for avoiding the presented hypothetical pollinator population declines (resulting from the DCE) with the expected production losses as calculated with a suitable pollination-dependence ratio. Indeed, the obtained WTP estimates for changes in the abundance of pollinating bees can be considered a byproduct of this study that, when compared to the actual costs of implementing the proposed conservation strategies, indicates how worthwhile this investment would be from the concerned farmers’ perspective. Accordingly, implementing an adequate bundle of conservation measures costs less than the farmers’ elicited WTPs for avoiding a 50 % decline in bee population. Hence, from a social point of view and explicitly taking into account the preferences of stakeholders, a tailored conservation policy is worthwhile.

Our results suggest that to galvanize a greater stakeholder engagement in the implementation of the proposed bee conservation strategies, Thai policy makers should start by educating farmers about the role of pollination in agricultural production, such as to bring their preferences for native bee abundance closer to the true use value of this ecosystem service. This could be achieved by using the already existing agricultural extension services provided by the Thai government and by the Royal Project Foundation to communicate to the farmers the benefits that pollination represent to their respective crops.

Furthermore, longan flowers produce abundant nectar, of which honey can be sold at a premium price. This has attracted individuals and enterprises with great investment power to this region to develop a local beekeeping industry with the imported European honeybee (*A. mellifera*). Nevertheless, the potential for an additional income that beekeeping offers remains currently untapped by the great majority of longan farmers. Thus, fostering on-farm beekeeping with native bees can be seen as a strategy to reconcile private economic incentives of smallholders with the goals to conserve native pollinators and their habitats.

We conclude by recognizing that Northern Thailand’s crop diversity presents a wide dependence range on animal-mediated pollination: from wind- or self-pollinated crops (e.g. rice and maize), through modestly animal-pollination dependent or profiting crops (e.g. strawberries and coffee), to crops substantially relying on pollination services (e.g. longan, litchi, squashes and pumpkins) (Klein et al. 2007). Therefore, realizing the full potential of a pollinator conservation policy in Northern Thailand would require promoting further research on the interdependency between the local pollinator fauna and the region’s broader agricultural landscape, assessing its economic implications from the production perspective and accounting for the preferences of the relevant farming communities.

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3.8 Errata

This chapter has been published as an article in a scientific journal [Ecosystem Services 18 (2016) 58–67][39]. Here, corrigenda are offered to the errata that have been identified, which nevertheless do not affect the key messages resulting from the published analyses.

i) The final DCE survey was conducted in Chiang Mai, only during the month of June of 2013 (cp. Section 3.3.4). The month of May was spent carrying out the DCE pilot to obtain prior estimates for the final experimental design.

ii) On Section 3.3.3 (page 71), it is incorrectly stated that a Bayesian efficient design was generated for the DCE survey. Instead, the underlying experimental design was generated to be efficient for a random parameters model without making use of Bayesian prior distributions.

iii) The descriptive statistics on Table 3.2 mistakenly suggest that the total *net* annual income is significantly different between male and female headed households. The statistics presented to support such inference [$t = 3.01$, $p = 0.03$; $F(1, 196) = 9.04$, $p = 0.003$] actually correspond to the difference between male and female headed households with regard to the *gross* annual income (revenue) from selling their longan production, i.e., excluding any other agricultural product (Female = THB 64,884.15 ; Male = THB 109,663.79). These statistics were accidentally put in place of those corresponding to the test associated to the presented claim [$t = 1.20$, $p = 0.23$; $F(1, 196) = 1.44$, $p = 0.23$], which is thus contradicted. There is nonetheless further evidence for income differences between male and female headed households, namely for differences in the total *net* agricultural income (Female = THB 49,937.80 ; Male = THB 95,131.35), which tested significant [$t = 3.31$, $p = 0.011$; $F(1, 196) = 10.94$, $p = 0.001$]. In view of this, it is still not farfetched to suggest that the significant interaction of gender with the cost attribute (i.e., reluctance to spending money) may be related to the significant household income differences brought about by gender differences (cp. Section 3.5, page 78).

iv) The fixed-cost (MWTP) model presented in Table 3.3 estimates 25 parameters. The corresponding likelihood ratio test and pseudo R-squared were computed with the LL function of a restricted model with two constants only, thus resulting in $K - 2 = 23$ degrees of freedom, instead of 22 as mistakenly stated in Table 3.3

v) During data entry, the binary variable “Beekeeping” (1 = yes; 0 = no) was wrongly coded for two respondents as 2, instead of 0. Accordingly, the percentage of respondents who are beekeepers changes from 17.68% to 15.66%, i.e. a difference of 4 counts resulting from two individuals that were each formerly counted as two. This variable also entered the fixed-cost (MWTP) model

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[39] https://doi.org/10.1016/j.ecoser.2015.10.010

presented in Table 3.3 as an interaction term, thus slightly changing its results and those presented in Table 3.4 as follows:

Table 3.6. Corrigendum for fixed-cost model (with corresponding WTP estimates) of **Table 3.3**

		Fixed-cost model		
Variable		*Coefficient* [a]	*(SE)*	*MWTP* [e]
Bee-friendly pest control	*Mean*	0.12722	(0.4058)	0.0[f]
	SD	2.55661***	(0.2402)	662.9***
Improving native bee habitat	*Mean*	0.22298	(0.3292)	0.0[f]
	SD	1.73698***	(0.1862)	450.4***
Native bee husbandry	*Mean*	0.67626***	(0.1561)	175.4***
	SD	1.59412***	(0.1717)	413.3***
−50% bee pop. (vs 0%)	*Mean*	−5.29531***	(0.4294)	−1373.0***
	SD	2.70436***	(0.3431)	701.2***
+50% bee pop. (vs 0%)	*Mean*	2.64634***	(0.3064)	686.2***
	SD	1.99020***	(0.2468)	516.0***
Costs (THB)	*Mean*	−0.00386***	(0.0003)	−1.0
	SD	Fixed parameter		
Heterogeneity in random parameter mean (Covariate × Attribute)				
Thinks native bees boost yields (yes)				
× Bee-friendly pest control		1.01974**	(0.3921)	264.4*
× Improving native bee habitat		1.03041**	(0.3245)	267.2**
Beekeeper (yes)				
× Native bee husbandry		0.92703**	(0.2957)	240.4**
Economic activity bees (yes)				
× Native bee husbandry		Fixed at zero		
× 50% bee pop. increase (vs 0%)		0.93692**	(0.3256)	242.9**
Male (yes)				
× Costs		Fixed at zero		
Log-Likelihood (LL)		−1500.7022		
AIC/*N* [b]		1.2843		
McFadden pseudo R-squared [c]		0.3433		
LRT [c] [d] χ^2 (*df*)		(23) 1568.995***		

[a] Significance levels: * $p < 0.05$, ** $p < 0.01$, *** $p < 0.001$. [b] AIC = Akaike information criterion; $N = 2376$. [c] Based on the LL function of a restricted model with two constants only, i.e. with choice probabilities set at each alternative's sample shares. [d] Likelihood ratio test. [e] Thai baht (THB): €1 = 39.3048 THB, as of June 1, 2013. [f] MWTP of attributes with coefficient estimates not significantly different from zero were calculated at zero THB.

Table 3.7. Corrigendum for **Table 3.4**: Avg. and aggregated consumer surplus change estimates (THB) [a)]

Attribute	*Average household*	*Surveyed sample (N=198)*	*Total research area (N=13,131)* [b)]
−50% bee pop. (vs 0%)	−1,591.87	−315,190.17	−20,902,876.76
+50% bee pop. (vs 0%)	803.62	159,116.17	6,344,185.19
Bee-friendly pest control	287.66	56,955.77	3,777,201.21
Improving native bee habitat	252.73	50,040.87	3,318,617.97
Native bee husbandry	167.37	33,139.17	2,160,595.70
All measures implemented	707.76	140,135.81	9,256,417.92

Weighted with sample shares of idiosyncratic covariates (Table 3.2). [a)] €1 = 39.3048 Thai baht (THB), as of June 1, 2013. [b)] Number of registered households in Chom Thong and Saraphi districts (DoAE 2015).

vi) Lastly, in Table 3.5, the estimated longan production losses for the average farm (second column) were calculated using an average farm size of 1.144 ha (i.e., 7.15 rai). This value nevertheless corresponds to the average total cultivated land of the farms in the sample, which was mistakenly put in place of the farms' average longan acreage corresponding to 0.948 ha (i.e., 5.92 rai). Accordingly, the correct estimated longan production losses are presented in following Table (3.8):

Table 3.8. Corrigendum for Table 3.5: estimated longan production losses[a)] attributed to declining bee pollination services (THB) [b)]

Native bee population loss (%)	*Average farm (0.948 ha)*	*Surveyed sample (188 ha)*	*Total research area (7,078 ha)*
100.00	−91,513.24	−18,155,160.00	−683,522,460.00
50.00	−45,756.62	−9,077,580.00	−341,761,230.00
5.00	−4,575.6	−907,758.00	−34,176,123.00

a) Calculated with a 60% bee-pollination dependence ratio. b) €1 = 39.3048 Thai baht (THB), as of June 1, 2013.

Chapter IV

Regional differences in farmers' preferences for a native bee conservation policy: the case of farming communities in Northern and Eastern Thailand[40]

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Keywords: Thailand, Native bees, Crop pollination, Conservation policy, Generalized mixed logit, Scale heterogeneity.

Abstract

Evidence points to past bee-mediated crop pollination deficits in Chanthaburi province, Eastern Thailand. Conversely, no such evidence has yet been reported for Chiang Mai province (Northern Thailand), suggesting that wild pollination is delivered there above the requirements of local orchards. Discrete choice experiments (DCE) were conducted to elicit the preferences of pollinator-dependent orchard farmers with regard to three pollinator conservation measures and their possible effects on native bee populations in each region. We fitted mixed logit (ML) models on the resulting data to capture preference heterogeneity and to obtain willingness to pay (WTP) point estimates. To test our results' robustness, we also inspected for scale heterogeneity by fitting generalized mixed logit (GMXL) models on the pooled and individual datasets. This yielded WTP space estimates (i.e., directly from WTP distributions) and made possible the comparison of farmers' preferences for a native bee conservation policy in both regions. The results hint at significant WTP differences for some of the conservation policy attributes between both provinces. Furthermore, unobserved contributions to choice seem to have been more random in Chiang Mai. Our analyses also suggest that farmers who engage in bee-related activities are WTP more for a conservation policy that includes bee husbandry.

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4.1 Introduction

We elicited the preferences of orchard farmers in Chiang Mai province (Northern Thailand) and in Chanthaburi province (Eastern Thailand) regarding alternative policy scenarios aimed at conserving native pollinating bees. Two discrete choice experiments (DCEs) were conducted in 2013, the resulting data of which was fitted with the mixed logit (ML) and the generalized mixed logit (GMXL) models to identify possible sources of heterogeneity in respondents' willingness to pay (WTP) for three recommended conservation measures (cf. Section 4.1.2) and for their potential effect on local native bees. The WTP for such measures are expected to be higher among those farmers who are beekeepers, considering that a previous, firsthand exposure to honeybees interacting with an agro-ecosystem could increase the awareness about the importance of conserving their native equivalents and habitat.

We also inspected for choice behavior differences between the two regions and whether or not such differences could be traced back to individuals who *believe* having experienced an insufficient crop pollination due to native bee population declines, as this has been reported for Chanthaburi. In this regard, we emphasize that choice analysis addresses the value of *perceived*, rather than actual, changes in the provision of an environmental good. Furthermore, we inspect the potential influence of idiosyncratic characteristics on the overall importance that respondents place on the conservation policy attributes that we propose, relative to the unaccounted factors that also influenced their choices.

4.1.1 Context: current status of beekeeping and pollination services in Thailand

With the exception of the European honeybee, all other eight honeybee species are native to Southeast Asia (Kastberger et al. 2011). This region is also characterized by its stingless bee (Apidae, Meliponinae) diversity, with 32 species of the genus *Trigona* identified in Thailand to date (Klakasikorn et al. 2005; Rasmussen 2008). Beekeeping in Thailand traditionally consists of attracting wild swarms of the Eastern honeybee (*Apis cerana* F.) to bait-hives (typically an unsophisticated wooden box or hollowed-out trunk), where the colony will reside until a disturbance (e.g., honey harvest) causes them to abscond (Oldroyd and Wongsiri 2009). It is also a custom in rural Thailand to keep stingless bees (a practice known as meliponiculture), which has gained economic relevance in the past few decades. Farmers collect colonies from several stingless bee species in the forests and place them in artificial hives (of varying degrees of sophistication) on their farms for their pollination services and to harvest their honey (Heard 1999; Chuttong et al. 2014).

Native bee husbandry has gained popularity in the Eastern Thai provinces of Chanthaburi and Trat, where orchardists started managing stingless bees to compensate for past pollination deficits of rambutan

(*Nephelium lappaceum*) (Thanuart and Makhonpas 2013; Boongird 2014; Chuttong et al. 2014). Moreover, in our 2013 survey in Chanthaburi, some respondents informed us that prior to finding this solution to the pollination deficit, some orchard farmers of the region had experimented with renting *A. mellifera* colonies from beekeepers. They preferred keeping stingless bees over honeybees due to their relatively simple maintenance and shorter flight range[41], which can protect them from possible contact with pesticides from neighboring farms. These anecdotes corroborate a similar report by Boongird (2014) and raise the suspicion that a local pollination crisis might have taken place in this region in the past.

Habitat loss and fragmentation (through, e.g., deforestation), pesticide overuse and pathogens are among the main drivers of pollinator declines worldwide (Priess et al. 2007; Potts et al. 2010; Goulson et al. 2015). The occurrence of localized pollinator crises in Thailand is thus not unlikely, given the country's sustained deforestation rates (Johnson 2015) and the four-fold increase in pesticide imports for agricultural applications over the past decade (Thapinta and Hudak 2000; Panuwet, et al. 2012; Siriwong and Robson 2014). The official promotion of capital-intensive commercial crops within the framework of a national development strategy was one of the main drivers of deforestation in Thailand in the past (Delang 2002; Forest Carbon Partnership Facility 2013). More recently, the continuing increase in the price of rubber (*Hevea brasiliensis*) has led to the conversion of forests in the east, northeast and north of the country, as the rubber plantation area has become limited in Southern Thailand (Jawjit et al. 2010; Trisurat et al. 2010). Furthermore, the overuse of pesticides in Thailand may be a result of the preventive (rather than curative) pesticide application strategy that cash-crop farmers have adopted (Schreinemachers et al. 2011; Grovermann et al. 2013).

Currently, the Northern Thai lowlands are extensively cultivated with longan (*Dimocarpus longan*), a Sapindaceae fruit tree that relies heavily on bee-mediated pollination, particularly from *A. cerana* and stingless bees (Boonithee et al. 1991; Blanche et al. 2006; Pham 2012). Northern Thailand is the leading exporter of longan worldwide (US$274.3 million in fresh fruit in 2013), with approximately 138,500 ha and 206,000 households (47,300 ha and 69,330 households in Chiang Mai province) devoted to producing this fruit in 2013 (Thai Office of Agricultural Economics 2014), thus rendering this region's economy highly dependent on this crop (Anupunt and Sukhivibul 2005; Menzel and Waite 2005; Ministry of Commerce 2015). The extensive cultivation of longan in Northern Thailand has also attracted large-scale beekeeping operations, which (February-March) move their ~120,000 *A. mellifera* hives to farms around

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[41] The flight range of stingless bees is proportional to their body size, which may constrain the dispersion capacity of smaller species within the limits of forest fragments, further increasing the risk of extinction in their local population (Araújo et al. 2004). Compared to the wide foraging range of honeybees (*A. mellifera*), typically reaching distances of 5 km (Beekman and Ratnieks 2000), stingless bees have a considerably smaller flight radius: e.g. ~600 m for the subgenus *Trigona* (Boontop et al. 2008). Boonithee et al. (1991), for instance, reported a high abundance of *Trigona* sp. in a longan orchard in Northern Thailand within distances of 50 and 200 m from the adjoining forest, yet decreasing sharply between 2.5 and 4 km.

the Chiang Mai-Lamphun valley each season in search of longan nectar foraging sources. They produce ~4,200 metric tons of longan honey per annum, which is highly valued in the Asian markets, where it can be sold at a premium price (Wongsiri et al. 1998; Wongsiri et al. 2000; Seanbualuang 2012).

Thailand is also one of the world's leading producers of rambutan and durian, the former of which is a close relative of longan that also depends on bee pollination, i.e., predominantly from stingless bees and *A. cerana* (Shivaramu et al. 2012; Chuttong et al. 2014; Rincón-Rabanales et al. 2015). Rambutans are mostly consumed domestically (i.e., ~91% of the country's total production) and partly exported fresh and canned (~US$20.2 million in 2013). The cultivation of this fruit tree employed 116,200 households and 47,900 ha of land in 2013 and is mainly established in Eastern Thailand, with ~45% of the country's production concentrated in Chanthaburi province (Salakpetch 2005; Thai Office of Agricultural Economics 2014; Ministry of Commerce 2015). In Chanthaburi, rambutans are commonly intercropped with mangosteens (*Garcinia mangostana*), which are often heavily sprayed with insecticides to comply with the high aesthetic standards (that, e.g., reject fruit presenting any signs of insect damage on its surface) imposed on fruits destined to export markets (Methaphat 2010; Chaisrichonlathan and Noomhorm 2011).

Although the accuracy of this information is disputed, honeybees (*Apis*) and stingless bees have also been credited with the pollination of durians, a belief that is commonly shared among durian farmers in Chanthaburi (Boongird 2014). In truth, durians owe this service exclusively to nocturnal pollinators (Bumrungsri et al. 2009). For this reason, a decline in the population of bees would have no consequence on durian yields. Nevertheless, we mention durian in this context as in our study we focus on farmers' *perceptions* regarding the pollination of their crops.

4.1.2 A policy to conserve native bees in Thailand

Considering the economic importance of the pollination services provided by native bees to Thailand's orchards, it would make sense to implement a policy to conserve them and their habitats. This is especially relevant for regions such as Northern and Eastern Thailand, where the agricultural output is vulnerable to future pollination shortages. Moreover, by taking into account the preferences of pollination-dependent crop farmers with regards to the conservation strategies constituting such a policy, one could make ex-ante policy recommendations based on which strategies can be expected to engage more efforts and resources from the targeted farming communities and which ones will require greater government intervention.

The International Pollinator Initiative's Plan of Action (IPI-POA, cf. Section 1.3) provides guidelines to enhance wild pollinator conservation and habitat restoration. Its *adaptive management* pillar recommends, among others, the following conservation strategies: *i)* offering farmers bee-friendly alternatives to conventional pesticides (e.g., biological control and integrated pest management); *ii)* encouraging the protection and improvement of natural bee habitats within agro-forest ecosystems; and *iii)* fostering the husbandry of native bee species (Byrne and Fitzpatrick 2009). We conducted expert interviews and focus group discussions with farmers where we identified these measures as potentially having the greatest impact and implementation chances in Thailand’s current agricultural and political context.

Besides assessing farmers’ preferences for measures to conserve local native bee populations, we attempt at eliciting the existence value of native bees and the option value of preserving the pollination services they provide (among other value components), both of which are neglected by market prices. Indeed, this study does not attempt at estimating the market value of crop pollination by bees, nor does it offer a detailed examination of markets for pollination services. After all, the arguments for the preservation of bee diversity should reach beyond the crop pollination services that a set of dominant bee species may provide (Kleijn et al. 2015).

4.2 Materials and methods

4.2.1 Discrete choice experiments

Hanley et al. (2015) have thoroughly reviewed the methods that have been applied to measure the economic value of pollination services and draw attention to the, until recently, unnoticed knowledge gap regarding the non-market benefits of this ecosystem service. They suggest that only stated preference methods, such as contingent valuation (CVM) and discrete choice experiments (DCE), may fill in this gap, and they proceed to present a first attempt by Mwebaze et al. (2010) at capturing non-market values within a CVM. Mwebaze et al. (2010) deliver WTP estimates for the existence value of protecting honeybees in the UK; however, due to a small and non-random sample, these results are unreliable (Hanley et al. 2015).

The validity and reliability of CVM results have often been criticized for the inherent proneness of this method to estimation biases, such as the *hypothetical bias* and most notably the *strategic bias* (i.e., when respondents may deliberately misrepresent their preferences in the hope of influencing the succeeding decision-making process in their favor). In this regard, DCEs offer the advantage of overcoming some of these limitations and are especially suited to assessing trade-offs that involve multi-dimensional changes (Bennett and Adamowicz 2001; Hanley et al. 2001; Tisdell 2010).

In a recent study, Narjes and Lippert (2016, Chapter 3) conducted a DCE with longan farmers in Chiang Mai province. Their per capita WTP for the combined implementation of the native bee conservation measures mentioned above (cf. Section 4.1.2) and for avoiding a potential 50% native bee population decline were estimated at €18.1 and €40.5, respectively. These estimates strongly contrast with the comparatively high economic losses from a potential pollination deficit in longan orchards, as approximated using the pollination-dependence ratios given by Blanche et al. (2006) and Pham (2012). On the other hand, the estimated farmers’ aggregated WTP exceeds the relatively low investment that implementing such conservation policy would actually cost. The DCE approach thus informs policy makers about the relative support that each conservation strategy demands given the preferences of pollination-dependent crop farmers, who are ultimately the most directly concerned stakeholders.

4.2.2 Mixed Logit (ML) and Generalized Mixed Logit (GMXL) models

Narjes and Lippert (2016) fitted a mixed logit (ML) model that also allowed determining significant dispersions around the mean preferences, which they further explained with idiosyncratic factors such as the respondents' gender and attitude towards native bees.

However, the ML model has recently come under criticism for neglecting the fact that choice behavior may be more random for some respondents than for others (i.e., respondent-specific heteroscedastic errors). In other words, the heterogeneity in the preference for a single conservation policy attribute may actually (or partly) result from a scale effect, i.e., all attribute weights are scaled up or down proportionately across individuals (Louviere et al. 2002; Fiebig et al. 2010). Ignoring such a source of variation (i.e., confounding heteroscedasticity with preference heterogeneity) may result in biased estimates and thus lead to erroneous interpretation and policy conclusions (Scarpa et al. 2008). The ML model indeed accommodates heterogeneous scaling when all parameters are specified to be random and their corresponding errors are allowed to correlate (Rose et al. 2013). Nevertheless, a common practice to obtain WTP estimates in the ML context is to estimate a non-random cost coefficient (cf. Eq. 4.3 below). With the strong assumption of a homogeneous cost parameter, the researcher is implicitly assuming homogeneous scaling over the population; this would lead to biased estimates (i.e., from confounding scale and preference heterogeneity) in the likely case that either the true scale or cost parameters were indeed random (ibid).

Fiebig et al. (2010) propose tackling this issue with the generalized mixed logit (GMXL) model, which explicitly specifies a scale parameter and thereby can disentangle the sources of preference heterogeneity into randomness in the attribute coefficients and randomness in the overall scale of utility.

To comprehend the scale parameter, one has to first formulate an individual farmer's behavioral choice rule from the researcher's perspective. The researcher only controls V_{ij}, the "representative" portion of the indirect utility U_{ij}^* that a farmer i derives from a conservation policy alternative j ($U_{ij}^*, j = 1, \dots, J$), while the unobserved random "residual" term ε_{ij}^* remains exclusively known to the farmer (Thurstone 1927; Marschak 1960). Assuming that V_{ij} is linear additive in the conservation policy attributes X_j and their corresponding taste-weights β (henceforth referred to as part-worths), as given by $U_{ij}^* = V_{ij}(X_j) + \varepsilon_{ij}^* = \beta' X_j + \varepsilon_{ij}^*$, a utility maximizing farmer would choose the alternative h with superior utility from a given set of *J* conservation policy alternatives. Modeling this choice decision requires knowing the density of the unobserved residuals $f(\varepsilon_{ij}^*)$, which for the *standard logit* model are assumed to be distributed independently and identically (IID), following an extreme value type 1 (EV1) distribution that exhibits

$var(\varepsilon_{ij}^{*}) = \sigma^2(\pi^2/6)$ (cf. Section 1.4.3.1). Thereby, the parameter σ becomes a scale parameter of the underlying standard EV1 distribution and is therefore often referred to as the scale of utility in the choice analysis literature. Normalizing the residual variance to that of a known theoretical distribution requires a standardization of the utility expression, i.e., moving the unknown scale factor to divide the representative utility V_{ij}. The standardized utility

$$U_{ij} = V_{ij}/\sigma + \varepsilon_{ij} = (\beta'/\sigma)X_j + \varepsilon_{ij} \tag{4.1}$$

therefore results from $U_{ij} = U_{ij}^{*}/\sigma$ and has $var(\varepsilon_{ij}^{*}/\sigma) = var(\varepsilon_{ij}) = \pi^2/6$. From this transformation, the likelihood of the choice outcome described above can be expressed as the *standard* logit choice probability:

$$P_{ih} = \exp(V_{ih}/\sigma) / \textstyle\sum_{j=1}^{J} \exp(V_{ij}/\sigma) \ . \tag{4.2}$$

The variance of the unobserved residuals is definitionally linked to the implicit scale of utility. In fact, the standard logit model is usually modeled in its scaled form, resulting in part-worth estimates $\beta^{*} = \beta/\sigma$ that are not separately identifiable from scale (Train, 2009). Nonetheless, given the general IID assumption (i.e., σ is constant in the population) one can implicitly cancel out scale by dividing the part-worths of any two attributes $k = 1, \dots, K$. Such coefficient ratios are calculated to, e.g., obtain marginal WTP estimates, as

$$WTP_k = -\frac{\beta_k/\sigma}{\beta_c/\sigma} = -\frac{\beta_k}{\beta_c}\,, \tag{4.3}$$

where β_c weights the attribute related to the costs of implementing the conservation policy.

The unobserved utility variance can be accounted for by *explicitly* parameterizing the standardization step described above as the factor $\lambda = 1/\sigma$ that scales the vector of part-worths, i.e., $U_{ij} = \lambda V_{ij} + \varepsilon_{ij}$. Thereby, it also becomes apparent that, for *standard logit*, the scale parameter λ equals unity. (Train, 2009; Fiebig et al. 2010; Hensher and Greene 2015). Intuitively, λ is the weight that the respondents (equally) place on the utility they derive from V_{ij}, relative to the residual utility they derive from unobserved factors that contributed to their choice. In other words, the larger the parameter λ, the smaller *must* be $var(\varepsilon_{ij}^{*})$.

In reality, the residual variance may differ for different decision makers, i.e., $var(\varepsilon_{ij}^{*}) = {\sigma_i}^2(\pi^2/6)$, resulting in a heteroskedastic scale that is strictly inversely proportional to the standard deviation of the residuals ε_{ij}^{*}. The *GMXL* specification handles the complexity that arises when simultaneously contemplating heterogeneity in scale and in the part-worths (Fiebig et al. 2010). It maintains the IID assumption for the residual term ε_{ij}, yet assigns part of the unobserved heterogeneity to the scale and part to the part-worths (respectively subscripted to each individual as λ_i and β_i), as follows:

$$\beta_i = \lambda_i[\beta + \Delta z_i] + [\gamma + \lambda_i(1-\gamma)]\eta_i, \ \gamma \in [0,1] \ . \tag{4.4}$$

Here z_i are observed individual-specific characteristics (indicating, e.g., whether the respondent believes she has experienced native bee pollination deficits) that induce heterogeneity in the part-worth mean, and η_i embodies i-specific (unobserved) deviations from the mean β (thus capturing part-worth heterogeneity[42]). Some β_i may only present unobserved heterogeneity (i.e., homogeneous parameter means), in which cases the vector Δ is set to zero. The vector η_i can take any distributional form, but in this study it is assumed to be multivariate normal. The extent to which the standard deviation of η_i depends on the scaling of β is controlled by the parameter γ. As such, setting $\gamma = 1$ results in GMXL I, a special case of GMXL that assumes that $(\beta + \Delta z_i)$ is independently scaled from the standard deviation of η_i, whereas $\gamma = 0$ (GMXL II) imposes proportional scaling for both the part-worth means and their variances[43] (Fiebig et al. 2010).

Scale heterogeneity, on the other hand, may be expressed as a function of measurable individual-specific variables g_i (indicating, e.g., whether the respondent keeps bees on his farm or not) as follows:

$$\lambda_i = exp\left(\bar{\lambda} + \delta' g_i + \tau\varepsilon_{0i}\right),\ \varepsilon_{0i} \sim N(0,1)\ , \tag{4.5}$$

where $\bar{\lambda}$ and τ respectively denote a mean parameter and standard deviation of λ_i and ε_{0i} represents standard normally distributed unobserved heterogeneity. While the exponential form restricts λ_i to being positive, its expected value must be normalized to 1 to identify β as the mean vector of utility weights: given that $E[\lambda_i] = \exp(\bar{\lambda} + \tau^2/2)$, we set $\bar{\lambda} = -\tau^2/2$. This results in scale heterogeneity λ_i that is $\sim LN(1, \tau^2)$. We thus estimate the parameters that describe the variance of scale and not λ_i itself (Fiebig et al. 2010).

The GMXL choice probabilities are conditioned on the unobserved η_i and ε_{0i}. The analytical estimation of such a model would imply solving Eq. 4.2 in a multiple integral that does not have a closed form and therefore must be approximated through computational simulation. Indicating with $y_{ijt} = 1$ policy j chosen by farmer i in choice situation t, and with $y_{ijt} = 0$ the alternatives not chosen, the simulated probability $\hat{P}_i$ of observing farmer i making a sequence of choices $\{y_{ijt}\}_{t=1}^{T}$ is obtained as follows:

$$\hat{P}_i = \frac{1}{R}\sum_{r=1}^{R}\prod_t \prod_j \left(P\left(j\middle|V(X_{jt}), \varepsilon_0^r, \eta^r\right)\right)^{y_{ijt}}\ . \tag{4.6}$$

Here, the term $P\left(j\middle|V(X_{jt}), \varepsilon_0^r, \eta^r\right)$ results from inserting (4.4) and (4.5) into the logit formula (4.2) and is solved for R random draws $\{\varepsilon_0^r, \eta^r\}$ that are sampled from the distributions underlying η_i and ε_{0i}. This simulation is made iteratively for different population moments of the assumed distributions (i.e., mean and variance-covariance of β, collectively denoted as θ) and inserted into the log-likelihood function for

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[42] In this study, η_i were allowed to induce correlation in the random coefficients β_i.

[43] GMXL I ($\gamma = 1$): $\beta_i = \lambda_i(\beta + \Delta z_i) + \eta_i$; GMXL II ($\gamma = 0$): $\beta_i = \lambda_i(\beta + \Delta z_i + \eta_i)$.

all n farmers. The model estimation consists of finding the parameters θ that maximize the simulated log-likelihood function (Train 2009; Fiebig et al. 2010).

The GMXL II case offers a framework that handles the challenges resulting from estimating WTP as the ratio of two random part-worths (e.g., the moments of the WTP ratio distribution are undefined). We may first rewrite utility as separable in the monetary attribute $costs_j$ and the *non-monetary* attributes X_j^{nm}: $U_{ij} = -\lambda_i \beta_{ic} costs_j + \lambda_i \beta'_{ik} X_j^{nm} + \varepsilon_{ij}$, where X_j^{nm} contains all elements of X_j except x_1, the cost attribute, and $k = 2, \dots, K$. Assuming that the cost's preference heterogeneity is entirely captured by the scale parameter λ_i (i.e., by setting β_{ic} to one and its corresponding row in η_i to zero), utility can be reformulated into the following equivalent specification (referred to in the literature as *WTP space*[44]):

$$U_{ij} = -\lambda_i \left[costs_j + (1/\beta_{ic}) \beta'_{ik} X_j^{nm} \right] + \varepsilon_{ij} = -\lambda_i \left[costs_j + \phi'_{ik} X_j^{nm} \right] + \varepsilon_{ij} \ . \tag{4.7}$$

The random marginal WTP estimates ϕ_{ik} are directly estimated and independent of scale, while the scaled monetary part-worth $\lambda_i = exp\left(\bar{\lambda} + \delta' g_i + \tau \varepsilon_{0i} \right)$ is estimated by relaxing the $-\tau^2/2$ restriction on $\bar{\lambda}$ (Train and Weeks 2005; Fiebig et al. 2010; Greene and Hensher 2010; Hensher and Greene 2011).

4.2.3 Hypotheses underlying this study

In this study, we contemplate heterogeneity in farmers' choice behavior regarding alternative interventions to conserve native bees, which should be reflected in part-worth and scale standard deviations that are significantly different from zero. We thus postulate:

H3.1: There is significant part-worth heterogeneity, i.e., $var(\eta_i) \neq 0$ (cf. Eq. 4.4) and

H3.2: There is significant heterogeneity in scale, i.e., $\tau \neq 0$ (cf. Eq. 4.5).

We further hypothesize that such heterogeneity is partly explained by selected idiosyncratic variables that enter the part-worth and scale specifications as the vectors $\mathbf{z}_i$ and $\mathbf{g}_i$, and therefore we also postulate *H4:* $\Delta \neq 0$ and *H5*: $\delta \neq 0$ as follows:

H4.1: Farmers who are beekeepers have higher preferences for the conservation of native bees.

H4.2: Whether founded in evidence or not, the *notion* of a possible pollination deficit attributed by concerned farmers to native bee population declines may also have a positive effect on farmers' value perceptions.

H4.3: There are regional differences (i.e., between the two sampled provinces) in the preferences of farmers for the proposed conservation measures.

H5.1: The presence of scale heterogeneity partly results from a subsample of respondents, who ascribe past pollination deficits to native bee population declines, applying a higher weight on the explained utility

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[44] Model specifications of this type owe their name to the distributions (e.g. normal or log-normal), which are assumed for the WTP measures that can thus be directly estimated. In contrast, when these distributions are assumed for the estimation of random part-worths β_i, the model is said to be specified in *preference space*.

V_{ij} than the others. In other words, we hypothesize that the choice behavior is more consistent among farmers whose memory of past pollination deficits may have imparted them greater preference consensus.

H5.2: There are differences, between both sampled regions, in the relative importance that farmers place on the unobserved factors that contributed to their choices (i.e., $E[\lambda_i]_{Chanthaburi} \neq E[\lambda_i]_{Chiang\ Mai}$).

Similarly, scale heterogeneity may be also attributed to:

H5.3: Gender differences and

H5.4: A subsample of farmers who allow someone else's bee hives on their farms and/or are beekeepers themselves. Such direct exposure to honeybees in an agro-ecosystem may sensitize farmers about the importance of conserving native bees and their habitats and/or lead to more informed and thus consistent choices.

Although our analyses address farmers' perceptions, we would like to stress that the notion of a past pollination deficit attributed to declines in the population of native bees may be more justified for farmers in Chanthaburi than in Chiang Mai: there is scientific and anecdotal evidence indicating that an actual crop-pollination crisis took place in the former (cf. Section 4.1.1), whereas no such evidence has been reported for the latter. Moreover, we hypothesize that

H6: Compared to Chanthaburi farmers, Chiang Mai farmers are more likely to engage in activities involving bees manly for the direct economic benefits from the hive (e.g., honey), rather than to supplement their crops' pollination.

4.2.4 Experimental design and survey

This study analyses the datasets resulting from two separate discrete choice experiments (DCEs). A conditional logit model fitted on the data of the first DCE[45] delivered the prior parameter estimates that were assumed for the efficient design (Chaloner and Verdinelli 1995) underlying the second DCE. The latter was then conducted in November 2013 with 127 farmers in Chanthaburi province (Eastern Thailand). We interviewed 90 respondents in four villages of Makham district, while the remaining 37 were questioned in two villages of Khao Khitchakut province (Figure 4.1).

In January 2016, the database of the Thai Department of Agricultural Extension (DoAE) registered 3,369 households in Makham, who together farm ~4,380 ha of rambutan, the largest extension of land dedicated to this fruit's cultivation in a single district of Chanthaburi. The same database registered 1,456 farmers in Khao Khitchakut, who in total cultivate ~1,169 ha of rambutan, rendering this district Chanthaburi's third largest rambutan producer.

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[45] The first DCE was conducted with 198 randomly selected individuals from 10 villages in Chiang Mai province (Northern Thailand) in June 2013, as described in Narjes and Lippert (2016).

In contrast to the first DCE, a random procedure was not applied to the selection of the Chanthaburi sample. The villages were mandatorily assigned by the provincial administration, and all of the village heads insisted on providing their own selection of respondents. A selection bias may thus affect the representativeness of the results obtained from this dataset.

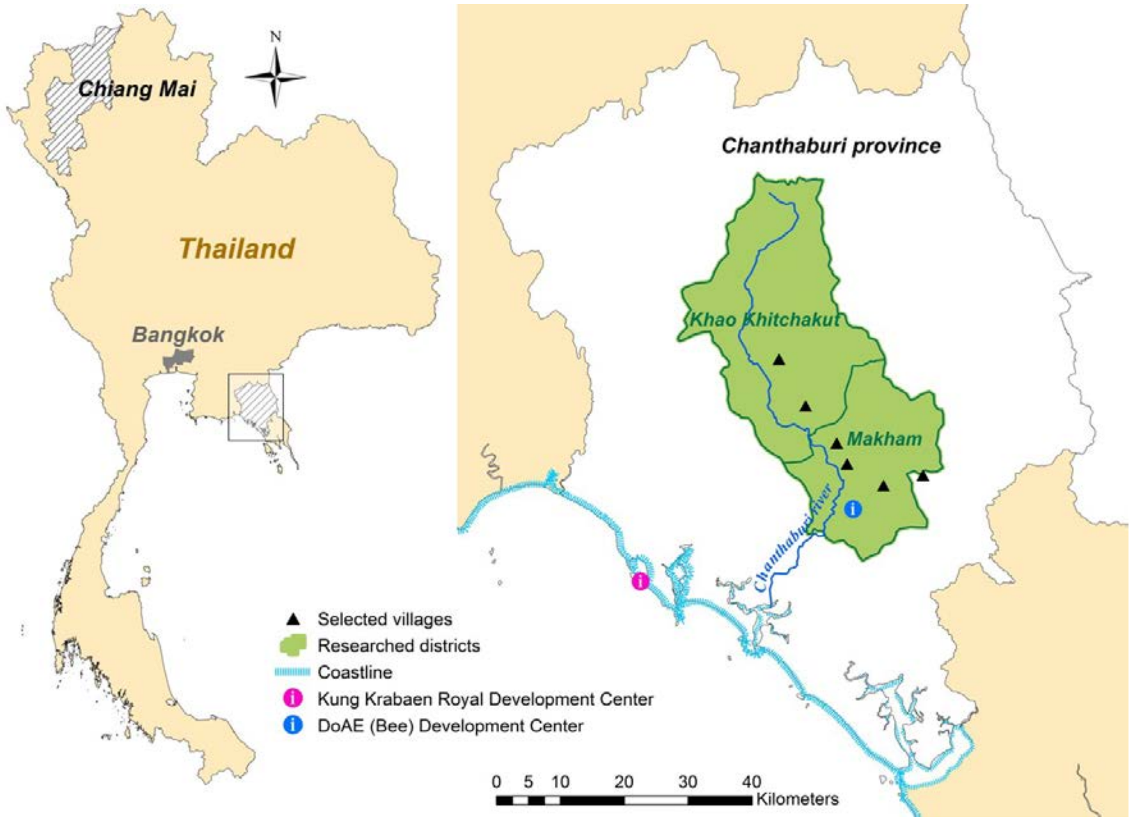

Figure 4.1. Research area in Chanthaburi. Source: Own representation using vector data from the DCW and GADM databases (DIVA-GIS, 2014).

In twelve different choice occasions, each respondent had to choose one of three alternative scenarios. Each alternative was described by five attributes, namely the three adaptive management techniques we selected from the IPI-POA toolkit, their potential impact on the local population of native bees, and a single advance contribution to cover their combined implementation costs (cf. Section 4.1.2).

Table 4.1. Choice alternative attributes, corresponding design levels and other variable definitions

Definition	*Levels*	*coding*	*variable name*
Bee conservation policy attributes (variables appearing in choice sets)			
Bee-friendly pest control	no a), yes	dummy	*PEST*
Improving native bee habitat	no a), yes	dummy	*HAB*
Native bee husbandry	no a), yes	dummy	*BEEKP*
Changes in native bee population (%)	−50 a), 0, +50	2 dummies	*NB_DEC* (−50%), *NB_INC* (+50%)
Policy implementation costs (THB) b)	0 a), 250, 500, 750	continuous	*COSTS*
Idiosyncratic variables			
Beekeeper (own bees)		dummy	*BEEKEEPER*
Keeps bees on her farm (own bees or someone else's)		dummy	*BEE_FARM*
Engages in at least one of the following activities: beekeeping, hunting for wild bee honey or charging a fee to allow someone else's bees to forage on her farm		dummy	*ECON_BEE*
Rated native bees' effect on her crop yields as good or excellent		dummy	*POS_BEE*
Believes he has experienced a native bee-pollination shortage		dummy	*POLL_DEC*
Farmer in Chanthaburi		dummy	*CHB*
Male respondent		dummy	*MALE*

a) Attributes fixed at these levels for the status quo alternative. b) The cost attribute represents a one-time fee that the farming households would pay to the local authorities for the implementation of the chosen policy alternative. €1 = 39.3048 Thai baht (THB), as of June 1, 2013.

Two of the alternatives, generically named "Policy A" and "Policy B", varied throughout the twelve choice sets by manifesting different attribute levels (Table 4.1), thus presenting different hypothetical scenarios of conservation policy implementation. These scenarios were associated with the full range of native bee population changes (i.e., −50, 0, +50) and were contrasted with an unvarying hypothetical status quo alternative, which described the absence of any conservation strategy (i.e., at zero cost) that would lead to a decline of half of the local native bees.

4.3 Results

4.3.1 Descriptive statistics

The descriptive data suggest important differences in the socio-demographics and farming practices of the Chiang Mai and Chanthaburi samples. Table 4.2 describes and compares these farming communities with selected variables, some of which we used to explain heterogeneity in part-worths and scale.

It becomes evident that agriculture plays a more important economic role among the surveyed households of Chanthaburi than it does in the longan farming population of Chiang Mai: with a 92% share derived from agriculture, Chanthaburi farmers earn 42% higher net incomes than the latter. Household incomes of Chiang Mai longan farms, on the other hand, consist mainly in off-farm earnings, i.e., 70% of net income.

Notably, the function of native bees as crop pollinators (POS_BEE) was more emphatically acknowledged by the respondents in Chanthaburi, who also present a significantly higher engagement in beekeeping activities (BEEKEEPER and BEE_FARM) than longan farmers in Chiang Mai. Moreover, being a Chanthaburi beekeeper is significantly related to POLL_DEC, the notion of a past crop pollination deficit that the farmer attributes to local native bee declines [$\chi^2(1) = 4.88$, p=0.027; tetrachoric correlation: ρ = 0.308, SE = 0.132, p<0.05]. Such association could not be determined for the Chiang Mai sample, where beekeeping is thus more likely to be practiced only for its direct benefits.

Table 4.2. Sample characteristics based on respondents' per-household values, 2013

	Chiang Mai $^{a)}$		*Chanthaburi* $^{b)}$	
Variable	*Mean*	*(SD)*	*Mean*	*(SD)*
Age (years)	55.76	(11.98)	44.89***$^{g)}$	(12.36)
Cultivated acreage (rai) $^{c)}$				
Longan	5.92	(6.09)	7.47	(5.82)
Rambutan	-		9.54	(9.97)
Durian	-		9.31	(9.65)
Total cultivated acreage (rai) $^{c)}$	7.15	(7.49)	25.42***$^{g)}$	(21.10)
Net annual agricultural income (THB) $^{d)}$	76,415	(96,822)	334,543***$^{g)}$	(361,809)
Net total annual income (THB) $^{d)}$	255,005	(654,567)	362,861	(367,551)
	Variable name	*Sample shares (%)*		
Male	MALE	58.59	49.61	
Main occupation: self-employed in agriculture	-	85.35	99.21***$^{h)}$	
Longan farmers $^{i)}$ [Total cultivated area (rai)]	-	100.00 [1172.7]	13.39 [127.0]	
Rambutan farmers $^{i)}$ [Total cultivated area (rai)]	-	0.00 [0.0]	70.87 [859.0]	
Durian farmers $^{i)}$ [Total cultivated area (rai)]	-	0.00 [0.0]	88.19 [1043.2]	
Keep bees on their farm $^{e)}$	BEE_FARM	38.89	62.99***$^{h)}$	
Their own	BEEKEEPER	15.66	59.84***$^{h)}$	
Someone else's	-	28.79	9.45***$^{h)}$	
Honey hunters $^{f)}$	-	20.71	29.13	
Engage in at least one of the above bee-related activities	ECON_BEE	50.00	72.44***$^{h)}$	
Completed only six years of primary school [no formal education]	- -	71.72 [5.56]	55.91***$^{h)}$ [1.57]	
Rated native bees' effect on their crop yields as good or excellent	POS_BEE	87.37	96.06**$^{h)}$	
Self-assessed knowledge regarding pollination before the survey: rated at least basic or [high]	- -	90.40 [7.58]	94.49 [24.41]***$^{h)}$	
Blame past yield declines on bee pollination deficits	POLL_DEC	38.38	47.24	

a) n = 198 respondents. b) n = 127 respondents. c) 1 rai = 0.16 ha. d) €1 = 39.3048 Thai baht (THB), as of June 1, 2013. e) *A. mellifera* or native bees (i.e., *A. cerana* and/or stingless bee spp.). f) Harvesting honey from wild bees in the forest. Significantly different from Chiang Mai sample with *p<0.05, ** p<0.01 and *** p<0.001. g) F-test. h) Chi-squared test of independence. i) Cultivates at least (not exclusively) 1 ngan (0.25 rai) of specified crop. Source: own calculation.

Also worth mentioning is the significant relationship between education and self-assessed knowledge regarding pollination [$\chi^2(10) = 21.62$, p<0.05] in the pooled dataset, where 171 respondents fell into the category of having acquired primary school education and having rated their pollination knowledge as basic. The independence of these categories could nevertheless not be rejected within the separate datasets, suggesting that their significant association in the pooled data results from a regional effect that may be explained with, e.g., the higher education level and higher pollination awareness in Chanthaburi. In fact, several initiatives supporting the research and development of native beekeeping have been hosted in Chanthaburi, including the Provincial Agricultural Occupation Promotion and Development Center

(Bee) of the DoAE, the Royal Development Study Center in Kung Krabaen bay (Figure 4.1) and a project sponsored by H.R.H. Princess Maha Chakri Sirindhorn in Makham district that trained local farmers on how to produce wooden stingless bee hive boxes (Chuttong et al. 2014; DoAE 2016; MoAC 2016; RDPB 2016).

4.3.2 Identifying sources of preference and overall scale heterogeneity

We used NLOGIT 5/LIMDEP 10 software to examine sources of choice heterogeneity with the ML and GMXL models. The GMXL I form is assumed in the analyses of this section to induce covariance in the vector of mean part-worths while allowing for independent standard deviations, thereby reducing the confounding[46] of scale and part-worth heterogeneities. We relied on the Bayesian information criterion (BIC)[47] to indicate which model offers a better fit and is thus preferred. Herewith, a lower BIC from an ML model would suggest that choice heterogeneity is better explained by randomness in the part-worths alone. Conversely, preferring a GMXL could indicate a non-negligible contribution of scale variance to choice heterogeneity. The Akaike information criterion (AIC), on the other hand, guided the choice between variations of models that resulted from explaining heterogeneity in the part-worth means with idiosyncratic variables; the large number of parameters that is necessary to capture complex choice behavior with these models (e.g., error correlations) would otherwise be heavily penalized by BIC (Fiebig et al. 2010).

Our first analysis consists of fitting a GMXL model to the Chiang Mai dataset to determine whether scale heterogeneity is present. To this end, we fitted several versions of this model approach, i.e., capturing scale and part-worth heterogeneity with different idiosyncratic characteristics. Table 4.3 reports the model results for the versions that minimized the BIC and AIC, which are to be compared with the ML models estimated by Narjes and Lippert (2016). For reference purposes, we also fitted a baseline ML without interaction terms in the mean part-worths to keep it at the minimum necessary number of parameters. In this way we produced a model with random part-worths (i.e., the full parameter vector) and correlated η_i (M1). Model M2 is a GMXL that captures unexplained scale and random part-worths heterogeneity, while Models M3 and M4 partly explain scale heterogeneity with either of two dummies that indicate whether

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[46] For a more detailed discussion of this issue refer to Hess and Rose (2012).

[47] For N choice observations and K parameters $\text{BIC} = -2\ln L + K\ln N$, where L is the maximized value of the likelihood function. For $N \geq 8$, BIC tends to choose models that are more parsimonious than those favored by the Akaike information criterion ($\text{AIC} = -2\ln L + 2K$), as it imposes a larger penalty on the added parameters (Neath and Cavanaugh 2012). Between two candidate models, a $\Delta\text{BIC} > 2$ is considered sufficient evidence to choose the model with the lower BIC (Kass and Raftery 1995). Given the few additional GMXL parameters, Fiebig et al. (2010) conclude that the BIC is the most reliable criterion to indicate whether scale heterogeneity is present or not.

the respondents *i)* believe they experienced a native bee-pollination shortage (POLL_DEC) or *ii)* keep their own or someone else's bees on their farm (BEE_FARM). Model M5 further explains heterogeneity in the part-worth means of BEEKP and NB_INC; the former with a dummy that indicates whether the farmer is a beekeeper (BEEKEEPER) and the latter with a dummy (ECON_BEE) indicating whether the household engages in at least one of three economic activities involving bees, i.e., beekeeping, hunting for wild bee honey and/or charging other beekeepers a fee to let their colonies forage longan nectar on their farms. This latter model resulted from a stepwise backward elimination derived from the "All-parameters-random" model reported by Narjes and Lippert (2016).

According to the obtained BICs, models M2, M4 and M5 are superior to M1 and to the ML model by Narjes and Lippert (2016), i.e., a significant amount of heterogeneity can be attributed to scale differences. Furthermore, the negative coefficient in POLL_DEC (M4 and M5) suggests a reduced scale (corresponding to a higher σ) among those farmers who stated that they had experienced a pollination deficit, which can be ascribed to the relatively higher contribution of unobserved factors to the utility they realized from their choices, i.e., a larger variance in residuals. Moreover, a reduced scale could also be significantly traced back to farmers keeping bees on their farms (M3) and to farmers' increasing age [model not reported: $\tau =$0.95487 (p<0.001) and $\delta = -0.00613$ (p<0.01); χ^2(27)=1645.20, p=0.00; BIC/N=1.326; AIC/N=1.256]. No important BIC difference between models M4 and M5 was found, yet AIC prefers the latter, indicating a model fit improvement from adding interaction terms that explain the means of BEEKP and NB_INC.

We also looked for sources of heterogeneity in the Chanthaburi dataset. The results are presented in Table 4.4 and hint at a significantly (yet not very) heterogeneous scale (M7) that a further model (M8) explained with a dummy indicating gender (MALE), according to which male respondents would have made fewer random choices. These results notwithstanding, M6 (the baseline ML model) was preferred by BIC (yet not by AIC), signaling that the heterogeneous choice behavior in Chanthaburi is better explained by solely capturing it with the random part-worths. Table 4.4 also reports a GMXL model (M9) that looks for heteroscedasticity that may result from pooling the Chiang Mai and Chanthaburi datasets. To this end, we defined a dummy indicative of Chanthaburi respondents (CHB) to capture possible scale effects resulting from differences in the underlying experimental design and/or in the regions. A further model, where CHB was specified to capture heterogeneity in both scale and the part-worth means, resulted in a non-significant effect of CHB on scale [model not reported: $\tau =$0.56502 (p<0.001) and $\delta = -0.02011$ (p=0.79); χ^2(30)=2438.00, p=0.00; BIC/N=1.339; AIC/N=1.288]. We thus compared M9 to a model in which CHB only has an effect on the part-worth means of the conservation policy attributes (M10), resulting in the latter being preferred by BIC and AIC in spite of its greater number of parameters.

We also tested whether POLL_DEC can capture part of the part-worth heterogeneity in the Chanthaburi dataset, an effect that for Chiang Mai had already been discarded following a stepwise approach by Narjes and Lippert (2016). A positive yet statistically (not quite) significant effect was detected for the POLL_DEC×BEEKP interaction [model not reported: β =0.798 (p<0.01) and Δ =0.484 (p=0.057); K=28; χ^2(26)=827.00, p=0.00; BIC/N=1.439; AIC/N=1.341]. This result suggests that those farmers in Chanthaburi who believe having experienced a bee-mediated pollination deficit value the beekeeping measure more than the others.

Table 4.3. Mixed and generalized mixed logit ($\boldsymbol{\gamma}$ fixed at 1) models fitted on Chiang Mai dataset

		M1: ML		*M2: GMXL*		*M3: GMXL*		*M4: GMXL*		*M5: GMXL*	
Variable		*Coefficient* [a)]	*(SE)*	*Coefficient* [a)]	*(SE)*	*Coefficient* [a)]	*(SE)*	*Coefficient* [a)]	*(SE)*	*Coefficient* [a)]	*(SE)*
PEST	*Mean*	1.29287***	(0.2122)	1.56450***	(0.2913)	1.56653***	(0.2925)	1.55832***	(0.2764)	1.45318***	(0.2402)
	SD	1.48343***	(0.2501)	1.79170***	(0.2714)	1.80074***	(0.2765)	1.66477***	(0.2613)	1.41726***	(0.2125)
HAB	*Mean*	1.28213***	(0.1566)	1.45823***	(0.2239)	1.46612***	(0.2256)	1.50250***	(0.2183)	1.47212***	(0.1909)
	SD	0.55993*	(0.2835)	1.35947***	(0.1709)	1.38437***	(0.1712)	1.03063***	(0.2061)	0.39712*	(0.1752)
BEEKP	*Mean*	0.84121***	(0.1453)	1.02148***	(0.1965)	1.03325***	(0.1972)	1.03743***	(0.1855)	0.85419***	(0.1870)
	SD	1.31470***	(0.1753)	1.67182***	(0.1721)	1.66406***	(0.1675)	1.68863***	(0.2391)	1.61024***	(0.1508)
NB_DEC	*Mean*	−5.23836***	(0.4294)	−7.30680***	(0.6523)	−7.34405***	(0.6569)	−7.02944***	(0.6309)	−6.39451***	(0.5266)
	SD	2.73449***	(0.4493)	1.89658***	(0.5036)	1.84724***	(0.5163)	1.80021***	(0.4363)	1.65209***	(0.3225)
NB_INC	*Mean*	3.26981***	(0.2945)	4.19878***	(0.4282)	4.24664***	(0.4324)	4.19584***	(0.4067)	3.39856***	(0.4108)
	SD	2.01934***	(0.3018)	2.58636***	(0.3661)	2.54772***	(0.3668)	2.14896***	(0.3657)	1.85950***	(0.2556)
COSTS (THB)	*Mean*	−0.00448***	(0.0004)	−0.00582***	(0.0006)	−0.00581***	(0.0006)	−0.00569***	(0.0006)	−0.00503***	(0.0006)
	SD	0.00320***	(0.0003)	0.00383***	(0.0005)	0.00388***	(0.0005)	0.00329***	(0.0007)	0.00383***	(0.0005)
Heterogeneity in random parameter mean (Δ)											
BEEKEEPER (z_i): BEEKP		-		-		-		-		1.03810**	(0.3474)
ECON_BEE (z_i): NB_INC		-		-		-		-		1.42583**	(0.4419)
Parameters in scale: (τ)		-		0.70201***	(0.0239)	0.75972***	(0.0345)	0.73007***	(0.0301)	0.64482***	(0.0296)
BEE_FARM (δ)		-		-		−0.22921***	(0.0570)	-		-	
POLL_DEC (δ)		-		-		-		−0.40490***	(0.0632)	−0.38335***	(0.0763)
Log-Likelihood (LL) [b)]		−1470.4834		−1462.9989		−1462.1765		−1454.0544		−1446.7320	
Parameters (K)		27		28		29		29		31	
BIC/N; [AIC/N] [b)]		1.326[1.261]		1.323[1.255]		1.326[1.255]		1.319[1.248]		1.319[1.244]	
Adjusted [McFadden] R^2 [d)]		0.353[0.357]		0.356[0.360]		0.356[0.360]		0.360[0.364]		0.363[0.367]	
LRT [c) d)] (*df*) χ^2		(25) 1629.4328***		(26) 1644.4018***		(27) 1646.0464***		(27) 1662.2907***		(29) 1676.9355***	

Refer to Table 4.1 for variable definitions. N = 2376 choice observations from 198 respondents. [a)] Significance levels: * $p < 0.05$, ** $p < 0.01$, *** $p < 0.001$. [b)] Compare to "all-parameters-random model" (Narjes and Lippert 2016; Chapter 3, Table 3.3): K = 32, LL = −1455.6685, BIC/N = 1.330, AIC/N =1.252. [c)] Likelihood ratio test. [d)] Based on the LL function of a restricted model with two intercepts only, i.e., choice probabilities set at each alternative's sample shares. Source: own calculation.

Table 4.4. Mixed (ML) and generalized mixed logit (GMXL; $\boldsymbol{\gamma}$ fixed at 1) models fitted on Chanthaburi dataset and GMXL fitted on pooled data

		M6: ML Chanthaburi d)		*M7: GMXL Chanthaburi* d)		*M8: GMXL Chanthaburi* d)		*M9: GMXL pooled data* e)		*M10: GMXL pooled data* e)	
Variable		*Coefficient* a)	*(SE)*	*Coefficient* a)	*(SE)*	*Coefficient* a)	*(SE)*	*Coefficient* a)	*(SE)*	*Coefficient* a)	*(SE)*
PEST	*Mean*	1.21990***	(0.1740)	1.06637***	(0.1717)	1.07457***	(0.1826)	1.19755***	(0.1607)	1.17471***	(0.1528)
	SD	1.65001***	(0.1960)	1.54174***	(0.1768)	1.55720***	(0.1810)	1.52034***	(0.1390)	1.58054***	(0.1436)
HAB	*Mean*	1.74206***	(0.2802)	1.69797***	(0.2787)	1.63901***	(0.2953)	1.25031***	(0.1483)	1.16678***	(0.1540)
	SD	1.49563***	(0.2689)	1.24242***	(0.3125)	1.21620***	(0.3013)	1.06475***	(0.1443)	1.04974***	(0.1507)
BEEKP	*Mean*	1.04907***	(0.2254)	0.98355***	(0.2208)	0.95039***	(0.2307)	0.84325***	(0.1346)	0.72991***	(0.1362)
	SD	1.35912***	(0.1892)	1.31533***	(0.2061)	1.25279***	(0.2064)	1.43884***	(0.1225)	1.39975***	(0.1180)
NB_DEC	*Mean*	−2.66051***	(0.3450)	−3.02800***	(0.4114)	−2.89768***	(0.3550)	−5.47557***	(0.4004)	−5.97960***	(0.4567)
	SD	1.42451***	(0.3303)	1.73858***	(0.4156)	0.90328***	(0.3458)	1.47275***	(0.4144)	1.42407***	(0.2626)
NB_INC	*Mean*	2.04788***	(0.2750)	2.17420***	(0.2884)	2.30901***	(0.3135)	3.30631***	(0.2490)	3.35288***	(0.2440)
	SD	1.49773***	(0.2535)	1.63267***	(0.3557)	1.48040***	(0.4000)	1.35985***	(0.3795)	1.72152***	(0.2851)
COSTS (THB)	*Mean*	−0.00232***	(0.0004)	−0.00241***	(0.0004)	−0.00253***	(0.0004)	−0.00413***	(0.0004)	−0.00411***	(0.0003)
	SD	0.00238***	(0.0004)	0.00260***	(0.0004)	0.00235***	(0.0005)	0.00291***	(0.0003)	0.00311***	(0.0003)
						Heterogeneity in random parameter mean (Δ)					
CHB (z_i): HAB		-		-		-		-		1.09402***	(0.2800)
CHB (z_i): BEEKP		-		-		-		-		0.79023***	(0.2257)
CHB (z_i): NB_DEC		-		-		-		-		2.46479***	(0.6103)
Parameters in scale: (τ)		-		0.15063***	(0.0224)	0.23764***	(0.0410)	0.53793***	(0.0214)	0.56236***	(0.0175)
MALE (δ)		-		-		0.52483***	(0.3321)	-		-	
CHB (δ)		-		-		-		0.23674***	(0.0574)	-	
Log-Likelihood (LL)		−995.5278		−996.2910		−993.1677		−2488.6325		−2478.7133	
Parameters (K)		27		28		29		29		31	
BIC/N; [AIC/N] d) e)		1.436[1.342]		1.442[1.344]		1.443[1.341]		1.338[1.291]		1.337[1.287]	
Adjusted [McFadden] R^2 c)		0.286[0.293]		0.286[0.292]		0.288[0.294]		0.327[0.325]		0.327[0.330]	
LRT b) c) (df) χ^2		(25) 823.4394***		(26) 821.9130***		(27) 828.1596***		(27) 2418.1554***		(29) 2437.9939***	

Refer to Table 4.1 for variable definitions. a) Significance levels: * $p < 0.05$, ** $p < 0.01$, *** $p < 0.001$. b) Likelihood ratio test. c) Based on the LL function of a restricted model with two intercepts only, i.e., choice probabilities set at each alternative's sample shares. d) Chanthaburi dataset: $N = 1524$ choice observations from 127 respondents. e) Pooled data (Chiang Mai + Chanthaburi): $N = 3900$ choice observations from 325 respondents. Source: own calculation.

Table 4.5. Willingness to pay (WTP) estimates in Thai Baht (THB) for Chanthaburi, Chiang Mai and pooled datasets (costs parameter fixed at 1 with std. dev=0)

		M11: Chanthaburi		*M12: Chanthaburi*		*M13: Chiang Mai* $^{g)}$		*M14: Chiang Mai* $^{g)}$		*M15: pooled data*	
Variable		*WTP* $^{a)\,d)\,e)}$	*(SE)*	*WTP* $^{a)\,d)\,f)}$	*(SE)*	*WTP* $^{a)\,d)\,f)}$	*(SE)*	*WTP* $^{a)\,d)\,f)}$	*(SE)*	*WTP* $^{a)\,d)\,f)}$	*(SE)*
PEST	*Mean*	529.865***	(95.000)	419.899***	(95.352)	133.412	(129.062)	290.413***	(64.659)	348.093***	(32.0257)
	SD	715.175***	(136.342)	711.365***	(110.390)	508.534***	(59.150)	577.934***	(66.669)	512.506***	(31.8897)
HAB	*Mean*	793.349***	(141.483)	775.677***	(152.726)	154.559	(81.246)	308.232***	(50.114)	355.655***	(41.8384)
	SD	988.039***	(199.768)	969.279***	(199.156)	333.378***	(37.925)	407.100***	(43.884)	450.952***	(31.8897)
BEEKP	*Mean*	461.883***	(106.975)	413.929***	(112.223)	177.437***	(40.384)	163.212***	(40.838)	216.880***	(37.3998)
	SD	903.766***	(174.903)	794.669***	(139.960)	318.904***	(42.389)	357.238***	(45.014)	452.149***	(33.7453)
NB_DEC	*Mean*	−1380.110***	(321.745)	−1295.380***	(295.193)	−1247.970***	(101.096)	−1445.340***	(118.190)	−1305.450***	(95.0837)
	SD	1017.920***	(232.051)	581.134**	(216.825)	515.354***	(101.177)	548.637***	(91.153)	629.691***	(72.5263)
NB_INC	*Mean*	939.196***	(122.856)	1004.630***	(120.128)	687.127***	(57.803)	705.825***	(78.973)	898.402***	(45.3727)
	SD	704.132***	(157.857)	752.330***	(191.536)	370.411***	(62.697)	474.329***	(81.126)	576.426***	(57.2815)
					Heterogeneity in random parameter mean (ΔWTP)						
BEEKEEPER (z_i): BEEKP		-		-		186.413*	(80.179)	211.108**	(81.798)	-	
ECON_BEE (z_i): NB_INC		-		-		225.422***	(66.393)	220.832*	(95.221)	-	
POS_BEE (z_i): PEST		-		-		204.683	(126.191)	-		-	
POS_BEE (z_i): HAB		-		-		178.478*	(73.040)	-		-	
CHB (z_i): HAB		-		-		-		-		356.052***	(62.3030)
CHB (z_i): BEEKP		-		-		-		-		265.419***	(54.2947)
CHB (z_i): NB_DEC		-		-		-		-		350.696**	(124.4221)
Parameters in scale: (τ)		-		0.25310	(0.316)	0.75980***	(0.127)	0.61819***	(0.115)	0.54113***	(0.1067)
MALE (δ)		-		0.66499	(1.173)	-		-		-	
POLL_DEC (δ)		-		-		−0.29629	(0.290)	−15.55560	(0.34×10^7)	-	
CHB (δ)		-		-		-		-		0.32619	(0.1999)
Log-Likelihood (LL)		−1012.8470		−1015.5205		−1500.6317		−1495.6878		−2558.9053	
Parameters (K); Obs. [N]		21[1524]		23[1524]		27[2376]		25[2376]		26[3900]	
BIC/N; [AIC/N]		1.430[1.357]		1.443[1.363]		1.351[1.286]		1.341[1.280]		1.367[1.326]	
Adjusted [McFadden] $R^{2\,c)}$		0.275[0.280]		0.273[0.278]		0.340[0.343]		0.342[0.346]		0.306[0.308]	
LRT $^{b)\,c)}$ (df) χ^2		(19) 788.8001***		(21) 783.4539***		(25) 1569.1360***		(23) 1579.0240***		(24) 2277.6098***	

Refer to Table 4.1 for variable definitions. $^{a)}$ Significance levels: * $p < 0.05$, ** $p < 0.01$, *** $p < 0.001$. $^{b)}$ Likelihood ratio test. $^{c)}$ Based on the LL function of a restricted model with two intercepts only, i.e., choice probabilities set at each alternative's sample shares. $^{d)}$ €1 = 39.3048 Thai baht (THB), as of June 1, 2013. $^{e)}$ WTP point estimates from ML with fixed costs (M6, Table 4.4). $^{f)}$ WTP space ($\boldsymbol{\gamma}$ fixed at 0) models. $^{g)}$ Compare to “fixed-costs model” (Narjes and Lippert 2016; Chapter 3, Table 3.3): $K = 25$, LL = −1513.8718, BIC/N = 1.356, AIC/N =1.295. Source: own calculation.

4.3.3 Willingness to pay estimation

The analyses in Section 4.3.2. helped us to disentangle the sources of heterogeneity in scale and random part-worths, thus guiding the WTP space specifications with γ fixed at 0 (i.e., GMXL II, according to Eq. 4.7) for the models M12 through M15 presented in Table 4.5. We first juxtaposed point estimates (M11, obtained from assuming a homogeneous COSTS part-worth in M6) with WTP space estimates (M12, guided by the part-worth and scale treatments of M8) for Chanthaburi to check for consistency in the signs and orders of magnitude. Similarly, we estimated two WTP space models for Chiang Mai to compare with the WTP point estimates model by Narjes and Lippert (2016): the first model (M13) was specified analogously to this WTP point estimates model, while the second (M14) was guided by the specification in M5. Finally, we tested for significant differences in the WTP of respondents from both regions by fitting a WTP space model (M14) on the pooled data, in which CHB explains heterogeneity in the conservation policy part-worths and in scale (the latter to rule out any possible regional effect on heteroscedasticity).

The estimates from M11 and M12 seem fairly robust, although PEST and BEEKP were 21% and 10.4% lower, respectively, in M12, which would thus render the WTP space estimates comparatively more conservative. Nevertheless, M11 was preferred by both information criteria, and the non-significant τ in M12 corroborates a negligible scale heterogeneity in the Chanthaburi dataset. A further model (with lower explanatory performance) suggested that, as in the case of Chiang Mai, Chanthaburi beekeepers have a higher preference for BEEKP than non-beekeepers, the mean WTP of the latter being not significantly different from zero [model not reported: ΔWTP_{BEEKP} =THB359, p<0.01; K=22; $\chi^2(20)$=785.73, p=0.00; BIC/N=1.437; AIC/N=1.360]. It is thus not surprising that the model without the BEEKEEPER×BEEKP term is preferred: the majority of respondents in Chanthaburi are beekeepers and therefore BEEKP suffices to capture the mean preference for the whole sample. Similarly, in another model, Chanthaburi farmers who present the POLL_DEC characteristic (regardless of being beekeepers or not) are WTP more than those who did not believe having experienced a native bee-pollination shortage [model not reported: $\overline{WTP}_{BEEKP}$ =THB251 (p=0.054) and ΔWTP_{BEEKP} =THB237 (p=0.07); K=22; $\chi^2(20)$=782.13, p=0.00; BIC/N=1.439; AIC/N=1.362].

Model M13 also yielded estimates slightly similar to those reported by Narjes and Lippert (2016), insofar as the mean WTP for PEST and HAB was not significantly different from zero. Additionally, the mean WTP for NB_INC and BEEKP, and the estimates for BEEKEEPER×BEEKP and ECON_BEE×NB_INC, were almost identical. The remaining estimates, nevertheless, differed greatly between the two models. On the other hand, M14 is preferred over both M13 and the point estimates model offered by Narjes and Lippert (2016) according to both information criteria. Moreover, the significant estimates for both of M14's interaction terms give further evidence of these terms' robustness.

The final comparison hints at significantly different preferences for conservation policy attributes between respondents of the two sampled locations. According to M15, Chiang Mai farmers were willing to pay ~THB356 for the implementation of the native bee habitat measure, while Chanthaburi farmers were WTP an *additional* THB356. This difference is slightly smaller than the ~THB485 difference between the estimates corresponding to M11 and M14. On the other hand, the additional THB265 that, according to M15, Chanthaburi farmers are WTP for BEEKP is a result almost identical to the corresponding difference between M11 and M14, given that Chiang Mai farmers are willing to pay THB196 for BEEKP in average (weighted by the 0.16 share of those keeping their own bees, presented in Table 4.2).

With regard to the remaining attributes (with the exception of NB_DEC), the estimates of M15 also resemble the differences between the estimates from M11 and M14, provided that the weights corresponding to the sample size and interaction term shares are correctly applied. These results also hint at the robustness of the WTP estimates from the selected models.

The idiosyncratic effects that were significant on the scale (i.e., δ) of the GMXL I models were not significant in all of the analogous WTP space (i.e., GMXL II) models. This may result from the fact that in GMXL II, λ_i not only weights the means of the random parameters (as in GMXL I), but also weights in equal proportion their corresponding standard deviations (cf. Eq. 4.4).

4.4 Discussion

We could not reject *H3.1* as there was significant part-worth heterogeneity for all attributes in all reported models. The sources of farmers' choice behavior heterogeneity are an important issue in this article, and thus it should be recalled that scale heteroscedasticity was especially relevant in the Chiang Mai dataset. The negative sign in the POLL_DEC coefficient (i.e., farmers who believe they experienced a bee-mediated crop pollination decline, place a relatively higher weight on unobserved attributes when making their choices) nevertheless came up as a surprise: we expected these farmers to place a higher weight on conservation policy attributes. We thus keep *H5.1* with an unanticipated negative sign. A possible interpretation of this counterintuitive result is that an important pollinator deficit has not yet been experienced (or perceived as such) among farmers in Chiang Mai and that POLL_DEC instead captured the random answers of respondents who did not fully understand the ecosystem service provided by the bees. In other words, farmers with a low understanding of and/or skepticism about the need for an intervention to conserve native pollinators may have introduced different subjective factors into their choices, thus contributing to an increased variability in the residuals; such farmers are also prone to misunderstanding the question captured by the dummy POLL_DEC, to which they may have randomly answered yes or no. Explaining scale differences may therefore point at a heterogeneous level of relevant knowledge (with respect to the importance of conserving the environmental good in question) or understanding of the DCE exercise on the part of the respondents. It may nevertheless also indicate that some attributes that were relevant to the choice decisions of a group of respondents were ignored by the researcher.

The supposition of a poor understanding of pollination services in Chiang Mai, relative to Chanthaburi, is supported by the descriptive statistics: the latter presents a significantly larger share of individuals that acknowledge the positive effect of native bees to crop pollination and with significantly higher self-assessed knowledge regarding this ecosystem service (Table 4.2). Although not directly providing evidence to test *H6*, a further indication of a lower awareness for the importance of bee-mediated pollination in Chiang Mai is given by the fact that there is a higher engagement in beekeeping in Chanthaburi, which only in this province correlates with the notion of having experienced bee-mediated pollination shortages. Between the two provinces, Chiang Mai farmers are thus more likely to perceive hive products as the only benefits they derive from bees.

Accounting for scale heterogeneity in the Chiang Mai dataset indeed resulted in a simpler model (with respect to the interactions in the part-worth means), suggesting that the models offered by Narjes and Lippert (2016) were over-parameterized in the part-worth vector and may have yielded slightly biased estimates. Farmers in Chanthaburi, on the other hand, seemed consistent in their choices (i.e., they placed a comparatively homogeneous weight on the policy attributes relative to the residual utility contributions), a behavior that could be attributed to a better understanding of the importance of conserving native pollinating bees. This reasoning is not farfetched considering the local evidence from *actual* past pollination deficits in Chanthaburi and the efforts that have been summoned to counteract such problems in this region.

The positive effect that CHB has on scale when modeling the pooled data (M9) may lead to the conclusion that Chanthaburi farmers place a greater weight than Chiang Mai farmers do on the overall conservation policy relative to ε_{ij}. This regional effect, nevertheless, is more dominant as an interaction term on the individual attributes (M10): Chanthaburi farmers placed a higher value on HAB and BEEKP than Chiang Mai farmers, while comparatively caring less about NB_DEC. The regional dummy thus affected the part-worth means individually, rather than acting proportionally over all attributes, i.e., we keep *H4.3*. A non-significant effect of CHB on scale implies that farmers in Chanthaburi and Chiang Mai have equal $E[\lambda_i]$ and therefore, on average, value conservation policy attributes in similar proportions to ε_{ij}. We thus reject *H5.2*. On the other hand, such a proportion is random for Chiang Mai farmers (i.e., individual-specific λ_i, given the significant τ estimate of the preferred model M5), whereas Chanthaburi farmers exhibit constant λ (see preferred Chanthaburi model M6), i.e., we keep *H3.2* for Chiang Mai, yet reject it for Chanthaburi.

As a consequence of discarding a regional effect on scale (according to the preferred model M10), we can also rule out any significant effect on scale from differences in the underlying experiments, which indeed were designed differently for both regions. Moreover, although an effect on scale could neither be rejected for BEE_FARM in Chiang Mai, nor for MALE in Chanthaburi (correspondingly Table 4.3 and Table 4.4, and hypotheses *H5.3* and *H5.4*), the addition of these parameters did not contribute to improvements in model performance.

The comparatively lower WTP of Chanthaburi farmers to avoid a 50% decline in native bee populations (M15) may result from the locally widespread adoption of traditional beekeeping (predominantly stingless bee spp.) that has endowed their farms with crop pollination independence, which otherwise is primarily mediated by wild bees. In contrast, traditional beekeeping in Chiang Mai mostly relies on *A. cerana* bees that are baited into man-made hives from which they eventually abscond back into their natural habitat (i.e., unlike stingless bees, which, once captured, can be kept indefinitely in box hives). At any rate, being a beekeeper plays an important role in forming preferences for the implementation of a native bee husbandry measure. This is also true for Chanthaburi, if we take into account that most farmers there are also beekeepers and that in an unreported model (cf. Section 4.3.3), the WTP for BEEKP only came out as significant for those individuals who keep bees. This leads us to keep *H4.1*. However, that model is less preferred than M11 probably because most farmers (60%) in Chanthaburi are also beekeepers. Furthermore, the notion of a possible native bee-pollination shortage (POLL_DEC) had a positive, yet ambiguous effect on Chanthaburi's farmers' preference for BEEKP (i.e., the beekeeping measure), which may result from POLL_DEC being confounded with BEEKEEPER (i.e., being a beekeeper) in this province. No confident decision could thus be taken regarding *H4.2*.

By all means, one should be careful with the interpretation of the WTP estimates for the percentage changes in the population of native bees, as these considerably exceed the highest implementation cost presented in the choice cards (i.e., THB750). One should also be wary not to infer about the entire population of targeted Chanthaburi farmers, considering that it was not possible to survey this population in a representative fashion.

We would like to close this section by situating this study's GMXL application in the context of following recent discussion: According to Hess and Train (2017), scale heterogeneity is a form of correlation in the part-worths that cannot be separately identified from other sources of heterogeneity. Furthermore, Hess and Rose (2012) already warned that models such as the GMXL, with which a number of authors try to disentangle scale heterogeneity from taste heterogeneity, maintain the scale/part-worth confounding and that the gain in model fit from those specifications results from allowing more flexible distributions.

In line with their criticisms, these authors suggest that a RPL specification is sufficient for capturing all sources of correlation, including scale heterogeneity (Hess and Rose 2012; Hess and Train 2017). In fact, Hess and Train (2017) argue that GMXL is a restricted form of RPL, unless the part-worths in the latter are assumed to be uncorrelated, in which case they consider the embedding is reversed. They further acknowledge that, if correlation is allowed in a full vector of random part-worths, the GMXL can accommodate scale heterogeneity. Nonetheless, they maintain that modelling scale imposes a restriction relative to RPL, arguing that in doing so, the covariance matrix is being captured by a single (scale) parameter, which draws on the argument that the heterogeneity from scale and from taste cannot be disentangled (ibid).

Indeed, the GMXL models we fitted (see Tables 4.3 and 4.4) do not separately identify scale heterogeneity. Instead, the specification proposed by Fiebig et al. (2010) assumes a functional form for the distribution of the scale parameter, such that its expected value equals unity. Thereby, one identifies the parameters that describe such distribution as the coefficients to be estimated, instead of estimating the scale parameter itself (see Section 4.2.2). Furthermore, the special case of GMXL that we applied (i.e., GMXL I, cf. Equation 4.4 and setting $\gamma = 1$) assigns heterogeneity to taste (through η_i, which captures independent unobserved deviations from the mean part-worth vector), separately from the heterogeneity that enters the model as correlation in the part-worth vector $[\lambda_i(\beta + \Delta z_i)]$, i.e., through the random scale parameter λ_i. Having additionally allowed all part-worths to be correlated, our GMXL model specifications produced parameter estimates that separately described the distributions assumed for the part-worth vector (i.e., multivariate normal) and for the scale parameter (i.e., lognormal with $E[\lambda_i] = 1$). The question nevertheless remains as to whether the improved goodness of fit in several of the estimated GMXL models (according to e.g., AIC and BIC) resulted from assuming a mixture of distributions that is more flexible than that assumed for the RPL models (see Tables 4.3 and 4.4).

As for the WTP space estimation (see Equation 4.7), Hess and Train (2017) affirm that such models allow for all sources of correlation as long as the price effect enters linearly the underlying utility specification, which is the case in our WTP analyses (see Table 4.5).

4.5 Conclusions

From the above analyses, we can dismiss the null hypotheses of homogeneous choice behavior between and within Chiang Mai and Chanthaburi farmers regarding alternative native bee conservation policies. We further conclude with confidence that such heterogeneous choices can be partly explained by the influence of observed idiosyncratic factors on the preference for the constituting policy attributes and on the unobserved residual variability.

Our results suggest that those farmers in Chiang Mai who answered *yes* to the question of whether they believed they had experienced a past bee-mediated pollination deficit derived less utility from the conservation policy measures, relative to other (unobserved) choice decision influences. This result is nevertheless challenged by the lack of evidence for an actual pollinator crisis in Chiang Mai. We therefore suggest that, instead, this question captured the poor understanding of a portion of farmers regarding the agricultural importance of bees, which further led them to base their choice decisions, to a larger degree, on unobserved factors. Similarly, Chiang Mai farmers who keep their own or someone else's bees on their farm may have incorporated relatively more unobserved factors into their decision process than farmers who do not keep bees. We presume that this effect results from longan farmers of Chiang Mai valuing bees, almost exclusively, for their direct economic benefits: most farmers keep bees that belong to beekeeping entrepreneurs who pay them for the right to forage longan nectar, and the few farmers who keep their own bees probably do it entirely for the hive products.

Engaging in activities that involve bees (which in Chiang Mai is likely to be mainly driven by their direct economic benefits) also has a significant effect on the preference for bee husbandry as part of a native bee conservation policy in this region. This finding, initially proposed in the study by Narjes and Lippert (2016), is substantiated by the GMXL estimates of the present analyses. Furthermore, this study preserves, to a considerable degree, the orders of magnitude and proportions of the willingness to pay (WTP) estimates for Chiang Mai from Narjes and Lippert (2016), thus indicating their robustness.

Perhaps the most important finding of this study is that the WTP for a native bee conservation policy was, in general, significantly higher in Chanthaburi than in Chiang Mai. The fact that the majority of Chanthaburi farmers (~60%) are also beekeepers (which in this region correlates with the notion of a past bee-mediated pollination deficit), makes it difficult to differentiate their value perceptions from those farmers who do not keep bees. Moreover, it suggests that the comparatively higher WTP of Chanthaburi farmers may result from the farmers' actual need to manage their own crops' pollination.

We further propose that an actual local pollinator decline may have made Chanthaburi farmers more aware of the importance of conserving native bees, while (paradoxically) making them more independent from the provision of wild pollination services, as they started managing crop pollination with stingless bees.

There are only a few species of domesticable honeybees and stingless bees in Thailand (Chuttong et al. 2015). Relying solely on bee husbandry for their conservation may pose the risk of neglecting the rest of the native pollinator fauna, which already contribute substantially to crop pollination and may serve as important insurance in the event of managed bee shortages (Kremen et al. 2002; Steffan-Dewenter 2005). A native bee conservation policy should thus integrate all three IPI-POA recommended adaptive management strategies (presented in this study) and further raise awareness of the importance of pollinators among the general public and special interest groups through the dissemination of high quality and easy-to-understand information (Byrne and Fitzpatrick 2009). Such a conservation policy should be seen as an investment, considering that the calculated costs of implementing the proposed conservation strategies would only amount to a fraction of the potential production losses that a bee-pollination deficit could entail (Narjes and Lippert 2016).

On average, the three proposed conservation measures were valued positively, yet all models coincide in their significantly wide-ranging standard deviations. Although expected (and partly explained in this study), this result poses a challenge for the implementation of these measures; to increase these policies' chances of success, policy makers could gain further insights from qualitative analyses that try to explain such part-worth heterogeneity.

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Chapter V

Discussion and conclusions

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The body of research of this thesis can be divided into two main parts. The first part consists of the (theoretical) microeconomic model presented in Chapter 2, which offers a generalized analytical framework that covers a broad range of farmer-beekeeper-nature interactions. The second part consists of the empirical analyses presented in Chapters 3 and 4, in which choice modelling is applied to assess the preferences and willingness to pay (WTP) of bee-dependent crop farmers for three different bee conservation measures and for the potential effect of a bee conservation policy on the population of native (wild) bees in Thailand.

This chapter discusses the main findings and contributions of both central parts of this thesis in Section 5.1, and their caveats and limitations in Section 5.2. It concludes in Section 5.3 by delineating recommendations for future research directions that may depart from this dissertation.

5.1 Contributions of this thesis

5.1.1 Conceptual model

This dissertation joins in the global efforts of mainstreaming the conservation of bees and pollination services into policy (Dias et al. 1999; Byrne and Fitzpatrick 2009), especially in the agricultural sector. Its contribution to this endeavour begins by giving a comprehensive picture of the role that bees, wild and managed, play in agricultural production. This is achieved in Chapter 2, insofar the basic theoretical model of Rucker et al. (2012) is generalized into an analytical framework that systematically describes a wider spectrum of the globally found diverse farmer-beekeeper interactions. Until now, the marginal productivity of wild bees has been neglected in the economic analyses concerned with the formation of pollination fees, while the relationship between farmers and beekeepers has mostly been depicted as the extreme case where wild pollination is being fully replaced by commercial pollination services. Additionally, the economic potential for on-farm beekeeping that resides in several bee species has been rather overlooked in the body of scientific literature, at least as regards cost-benefit analyses. Such potential is also assessed in Chapter 2, where the adoption of on-farm beekeeping is methodically compared to hiring commercial pollination services by juxtaposing the costs of these two alternatives against the total benefits each of them yields in terms of crop pollination and hive products. The proposed analytical framework thus also serves as a typology of farmer-beekeeper interactions and helps identifying the institutional settings and the economic and agro-ecological factors that are most likely to lead to a specific equilibrium supply of pollination. It thereby helps structuring the postulation of empirically testable hypotheses; this idea is elaborated with an example for potential future research in Section 5.3.

5.1.2 Elicitation format

The second part of this dissertation's body of research (i.e., Chapters 3 and 4) contributes to the efforts of assessing the economic value of wild pollinating bees. Its novelty consists in having conceived an elicitation format that captures at least part of the non-use value of bees by means of discrete choice experiments (DCEs) and choice modelling, which so far have been deemed unsuitable for this purpose. Mburu et al. (2006) for instance had validly argued that stated preference methods are limited by the knowledge that respondents may or may not possess regarding the quantitative contribution of pollinators to agricultural production. In response to that limitation, this thesis proposes eliciting the value that farmers place on different *measures* aimed at conserving native bees and the ecosystem services they deliver, instead of just asking them to state their preference for this environmental resource. After all, the value that farmers attach to the conservation of native bees conceivably transcends their direct use benefits to include non-use values, such as their cultural significance.

Under the proposed format, the relative value estimates may suggest potential loss aversion attitudes (as seemingly is the case in Chiang Mai province) and offer a ranking that hints at the expected commitment of farmers to the single bee conservation measures and at whether corrective or preventive measures are preferred. Furthermore, the total financial burden of implementing a comprehensive policy aimed at conserving native bees can be compared to the potential yield losses that would be attributed to their decline. Thus, eliciting the preference for bee conservation measures can be easily translated into policy recommendations.

5.1.3 Choice modelling applicability

A DCE dataset with enough choice observations lends itself to modelling preference heterogeneity, for which the random parameter (RPL) logit model has become increasingly popular among researchers and practitioners. The RPL can approximate any random utility model (McFadden and Train 2000) and is very flexible in that it allows for random taste variation, unrestricted substitution patterns and correlation in the unobserved factors, thus overcoming three key limitations of standard logit (Train 2009). Chapters 3 and 4 rely on RPL to estimate taste heterogeneity and to explain it by socio-demographic variables.

The capability of RPL to model heterogeneity is nevertheless limited to capturing variation in the sample's preference for the included DCE attributes and thus implicitly neglects the potential variation in the extent to which the choices of some (groups of) individuals have been influenced by unobserved factors, i.e.

heteroscedasticity. If such source of variation exists, the normalization of the error term translates into a scaling of the part-worth vector that is heterogeneous among individuals or subsamples. A heterogeneous scale is therefore another source of correlation, and one which the RPL model cannot disentangle from the correlation in the taste coefficients (Fiebig et al. 2010, but see Hess and Train 2017 and Section 5.2 of this dissertation). In order to deal with this issue, Fiebig et al. (2010) developed the generalized mixed logit (GMXL) model, which according to them solves the confounding of scale and taste heterogeneity.

This thesis does not contribute with novel econometric models, yet it explores the applicability of GMXL in controlling scale heterogeneity in order to compare DCE datasets that differ in their underlying designs and in the regions where the surveys were administered (see Figure 5.1). To this end, one special GMXL case that reduces the confounding of scale and taste heterogeneity was specified, while a dedicated selection procedure that penalizes the added parameters in a differentiated manner (i.e., select models with lowest BIC for scale heterogeneity and with lowest AIC for taste heterogeneity) was applied. This work is, to my knowledge, the first comparative study of this kind.

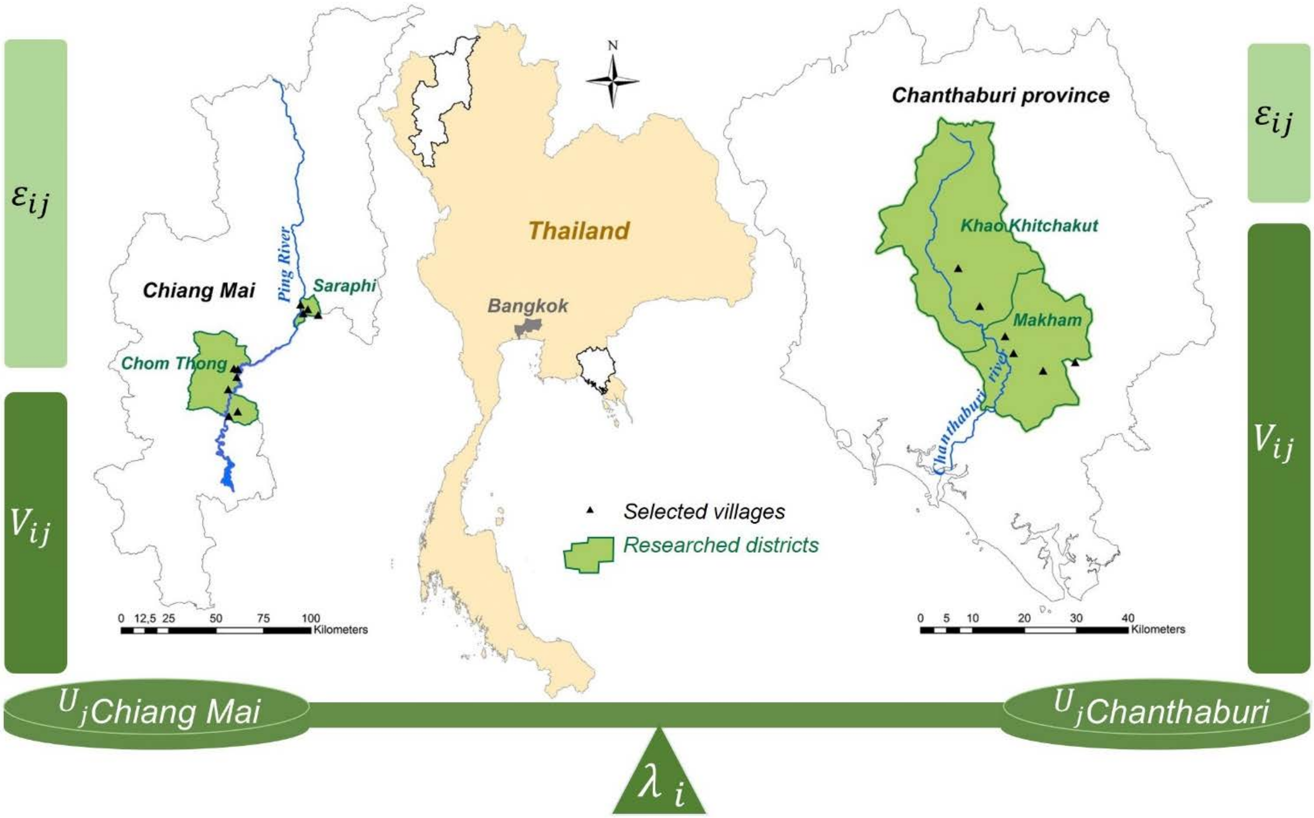

Figure 5.1 The scale parameter $\boldsymbol{\lambda_i}$ is the ratio of the representative utility $\boldsymbol{V_{ij}}$ to the variance of the residuals $\boldsymbol{\varepsilon_{ij}}$ that results from normalizing the latter (cf. Section 4.2.2). This proportion may differ between DCE datasets (as hypothesized for Chiang Mai and Chanthaburi, cf. Section 4.2.3) due to differences in e.g., the experimental designs and/or the extent to which unobserved factors may influence individuals' choices. Controlling for $\boldsymbol{\lambda_i}$ thus offers a common scale to meaningfully compare utility $\boldsymbol{U_{ij}}$ changes (i.e., from choosing an alternative $\boldsymbol{j}$) between groups of individuals $\boldsymbol{i}$, e.g., $\boldsymbol{dU_{jChiang\ Mai}}$ vs. $\boldsymbol{dU_{jChanthaburi}}$. Source: own representation with data from DIVA-GIS (2014).

5.1.4 Literature review

The literature on choice modelling has traditionally been produced by authors that are mostly affiliated to the fields of transportation and market research, which widely rely on (and often combine) both revealed and stated preference data (Agresti 2003; Train 2009); its language may thus often seem arcane to the outsider. It is only recently that other fields, including those of agricultural and environmental economics, started contributing with examples of DCE applications in the valuation of non-marketed goods. The growing scope of disciplines that rely on this method and the fast development of modelling capabilities (which have been made possible due to simulation advances in the wake of improved computational power) have enriched the choice modelling literature, yet they have also imparted it an inconsistent jargon that can easily lead to confusion. This became evident to me when I started consulting the relevant literature for this work. To remedy this problem, the second part of this dissertation's body of research contributes with the literature review that I prepared for the articles presented in Chapters 3 and 4. There, I attempt at breaking down the theory of choice modelling (especially with regards to RPL and GMXL) in a language that is consistent with and relatable to the econometrics vernacular used by agricultural and environmental economists.

5.1.5 Case studies

Finally, this work presents two case studies (see Chapters 3 and 4) that report anecdotal evidence on the state of pollination services as perceived by farmers from two regions that are arguably characterized by different agro-ecosystems. Notwithstanding the lack of measured evidence on the trends in diversity and distribution of native bee populations in Chiang Mai and Chanthaburi, these reports raise a red flag regarding the potential loss of local native bees due to the practice of intensive agriculture. Herewith, this thesis appeals to the international pollinator initiatives (Dias et al. 1999; Byrne and Fitzpatrick 2009; see Section 1.3) to address the possibility of such declines by monitoring the pollinator communities of Thailand and by promoting their conservation.

5.2 Caveats

The knowledge gained from this work is nevertheless not without its caveats. During my research it became apparent that its theoretical backdrop, assumptions and applied methods are subject to limitations that deserve being discussed. In this section I identify such limitations and warn about their potential implications on the interpretation of this dissertations' results.

First, the analytical framework presented in Chapter 2 is subject to following simplifications:

i) It models the optimal stocking density of bees based solely on their marginal productivity (i.e., per unit of land cultivated with honey plants) as regards honey and their contribution to fruit production. Here, their value (i.e., the equilibrium bee wage) is thus ultimately determined by the demand and supply of these marketed goods, which yield benefits that typify (direct and indirect) use values. Therefore, as it stands, this conceptual model is prone to underestimating the total value of bees (wild[48] and managed); it inherently neglects their non-use values and, even if one would know such values (e.g., from estimates such as those obtained in Chapters 3 and 4), one could hardly incorporate them into the model in order to obtain a corrected theoretical optimum: this would require additionally modelling, in three-dimensional space, the demand of farmers for non-use benefits from bees (i.e., per additional colony on a unit of land) on a utility scale that applies an equivalence factor between wild and managed bees.

ii) The long-run equilibria of the proposed model extensions display a clear cut inflection at the point where the marginal benefits from wild bees are replaced by those derived from managed bees (cf. Figure 2.5). Thereby, the model rules out potential pollination synergies between managed and wild bees, such as those that Brittain et al. (2013) describe for Apis and non-Apis bee species.

iii) It additionally fails at taking into consideration the dynamic processes that may play a role in long-run equilibria. Path-dependence and economies of scale could, for instance, have partly explained why a honey industry that is based on the costly business of keeping a non-native bee species has been successfully established in Chiang Mai province, disregarding that the necessary factors of production (i.e., land cultivated with honey plants and abundant domesticable native honeybees) are freely available to orchardists (cf. Section 2.2.2.3 and Chapter 3). Under different historical circumstances and in the absence of economies of scale, such setup could conceivably have favoured economies of scope to the advantage of farmers instead. The analytical framework can nevertheless accommodate such influences in a complementary qualitative assessment, such as that made for transaction costs in Sections 2.2.2.1 and 2.2.2.5.

Secondly, the estimated value of longan yield losses that were calculated for hypothetical scenarios of declines in the population of local native bees in Chiang Mai Province (see Chapter 3; Table 3.5) may be subject to the following underestimation:

iv) These yield losses were obtained by applying a bee-pollination dependence ratio of 60% [following the experiment results reported by Blanche et al. (2006) and Pham (2012)] to an average yield of 5.5 metric tons per hectare [as reported by the Thai Office of Agricultural Economics (2014) for 2013]. This reference production level nevertheless corresponds to an assumedly full (100%) delivery of crop pollination from the surrounding natural habitats. Should this not be the case, as local native bees had already undergone population declines before 2013, these calculations would result in an underestimation of the entailed economic losses.

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[48] An underestimation is of particular concern with respect to wild (native) bees, as these are more prone to (local) extinctions than managed bees.

Thirdly, the DCE elicitation format used to obtain the value estimates presented in Chapters 3 and 4 presents following drawbacks:

v) Although our DCE estimates capture non-use values, there is no way to know to which extent such values contribute to the (lower bound) total economic value (TEV) of (conserving) native bees, given that the use value that farmers perceived (for crop pollination and bee hive products) during the DCE is likely to significantly deviate from the actual (measurable) value of crop pollination deficits and reduced wild honey corresponding to losing half of the local population of wild bees.

vi) The experimental design underlying the DCE that we conducted in Chanthaburi was generated by Ngene software using the part-worth estimates from Chiang Mai as prior values. This was done in order to minimize the standard errors of the parameters estimated with the data from Chanthaburi, however resulting in both provinces being surveyed with differing experimental designs. Thus, the parameter estimate accuracy was gained at the expense of confounding a potential regional effect on the *scale of utility* with that from differences in the underlying designs.

vii) Preceding each DCE interview, the respondents were informed about the importance of bee-mediated pollination to their crops and about the current trends and consequences of pollinator declines worldwide. This constitutes an “information bias” that, although indispensable for the survey[49], may misrepresent the target population and further lead to the extrapolation of overestimated values. Such bias could have been accounted for by deliberately withholding aforementioned information from a large group of randomly selected respondents (preferably from half of the DCE sample), thus introducing it as a treatment effect in a DCE that was administered to individuals in Chanthaburi and Chiang Mai, following a randomized block design. Ideally, the experimental design underlying the DCEs would have also been the same in Chiang Mai and Chanthaburi, in order to avoid confounding a potential effect on utility scale from design differences with that from regional differences. At any rate, conducting a DCE in which the information bias is controlled would also have given some notion of the extent to which non-use values play a role in forming preferences regarding the conservation of native bees, provided that the subsample of treated respondents was given precise information about the (per unit of cultivated land) contribution of wild bees to their production. An effect of such information on utility would thus have considerably corresponded to the use value that farmers perceive for wild native bees. Accordingly, if determined significant, one could have subtracted such effect from the part-worths corresponding to changes in the local population of wild native bees in order to obtain an

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[49] DCEs are based on the assumption that the target population has already formed preferences, which is unlikely in the case of unfamiliarity with the good in question. In view of this, it was indispensable to assist the formation of preferences for the conservation of native bees by informing the DCE participants about the role that pollinators play in crop production.

approximate estimate of their non-use value. Nevertheless, controlling for an information bias would have entailed costs in terms of either sacrificing explanatory power (with regard to the hypotheses postulated in Sections 3.3.3 and 4.2.3) or investing additional resources for a larger survey (preferably twice as large), the latter of which was an unfeasible option due to a limited budget.

Lastly, the following caveat is concerned with the inherently variable nature of preferences:

viii) As a general rule, empirical findings are expected to be replicable for the published facts to be verified, although this is not necessarily the case in the social sciences [(Schmidt 2009), but cf. Camerer et al. (2016) for replicability of laboratory experiments in economics]. Given that preferences tend to change (conceivably faster than individual economic behaviour and incentives), DCE replications are especially unlikely to consistently deliver data that reproduces value estimates over time, even if the replications would involve the same subjects and the same experimental design of the original survey.

5.3 Outlook

The first part of this thesis' body of research (i.e., Chapter 2) offers an analytical framework that generalizes the theoretical model developed by Rucker et al. (2012), which describes the particular case of commercial beekeepers interacting with almond farmers in California, USA. Rucker et al. (2012) use their basic theoretical model to analyse the determinants of pollination fees and to deduce hypotheses regarding the statistical dependence between the stocking densities of rented honeybee colonies and a series of explanatory variables (i.e., crop acreage, the expected prices of honey and crops, and the manifestation of cost relevant honeybee ailments), which they tested using a panel of 180 stocking density observations. Similarly, our model extensions (see Section 2.2.2) could guide the deduction of hypotheses for a broader spectrum of beekeeper-farmer interactions. In the case of Chiang Mai and Chanthaburi, for instance, one could hypothesize that the magnitude and direction of the payments between farmers and beekeepers depends on the productivity and prices of crops and managed bees, and on the abundance of wild pollinating bees from surrounding habitats. Testing such hypotheses would involve the estimation of a spatial econometric model using relevant cross-sectional data and some indicator for the average supply of wild bees per unit of cultivated land. The former data could be obtained via surveys (during which additional information such as geospatial location of the farms, abundance of managed pollinators and of nesting sites and floral resources, among other biophysical variables, could be collected), while for the latter one could think of the "Pollinator Abundance: Crop Pollination" model of InVEST (Sharp et al. 2016). This model estimates an index for the spatial abundance of wild bees (i.e., for each cell on a landscape), based on the predicted availability of nest sites and floral resources within the flight ranges of bees. The InVEST pollination model is also capable of producing an index of the contribution of local wild bees to agricultural production. Herewith, determining a significant effect of changes in the population of wild bees (or of changes in the ecological factors that affect them) on the magnitude and direction of payments between beekeepers and farmers could, for instance, lay the foundation for modelling scenarios with different provision levels of crop pollination services. Such normative models could become a useful tool for the evaluation of different conservation policies. In any event, the postulation of hypotheses should also be assisted by a thorough qualitative research of each specific case of beekeeper-farmer interactions, in order to avoid the quantitative analysis to result in misguided generalizations.

As for the models presented in the second part of this thesis' body of research (i.e., Chapters 3 and 4), one obvious use would be the transfer of values estimated for Chiang Mai and/or Chanthaburi (i.e., the study sites) to policy sites with similar agroecosystems, and preferably also in similar socio-cultural contexts. In principle, discrete choice experiment estimates are well-suited for the value function approach to benefit transfer (Pascual et al. 2010). One could thus use the estimates from Chiang Mai to derive a demand function for a policy site with a relatively healthy wild bee population, while the estimates from Chanthaburi could be applied to a policy site where wild bee declines are thought to have already affected crop yields. Nevertheless, our estimates for changes in the population of native bees are given in percentages, which could be problematic considering that the reference population levels are believed to differ between Chiang Mai and Chanthaburi. Transferring our DCE estimates would thus require transforming the value estimates for changes in the population of wild bees to their absolute value equivalents; this could for example be achieved by estimating the population of wild bees in Chiang Mai and Chanthaburi with the InVEST pollination model (Sharp et al. 2016), provided that the reference levels assumed by the respondents during the DCE have not dramatically changed since the time when the surveys were conducted. In any respect, one should be wary about the proneness of preferences to change over time whenever applying the benefit transfer approach (see Section 5.2).

Finally, from our analyses in Northern Thailand (see Section 3.4), we learned that, although longan farmers underestimated the true use value of crop pollination, their willingness to pay for a comprehensive bee conservation project by far exceeded the implementation costs that each faming household would have to incur for its realization. From a social perspective in Northern Thailand, it would therefore be clearly economically suboptimal to abstain from implementing such a pollinator conservation program. Also, in view of the globally diverse beekeeper-farmer-nature interactions (see Section 2.2.2), it is quite likely that many regions of the world can still benefit from an optimization of their agro-ecological conditions by means of the implementation of pollinator conservation programs. In this regard and against the theoretical framework developed in this thesis, one should assess the economic viability of such programs for ecologically critical cases, by assessing the net benefits resulting from their implementation vis-à-vis the costs associated with the risk of declines in local pollinator populations. This way, as demonstrated for the case in Chiang Mai, the (social) profitability of pollinator protection measures can become an argument for the promotion of nature conservation and may support political efforts to prevent ongoing biodiversity erosion.

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SUMMARY

During the last few decades, worldwide reports of declines in the population of wild and managed bees (especially in Europe and North America) have raised the alarm about the harmful effect of intensive agriculture on pollinators (especially in connection with the misuse of synthetic pesticides) and about the consequent reduction in pollinator-dependent crop outputs. Although most reports concern the European honeybee (*Apis mellifera*) and bumblebees, these species often act as proxies for the overall health of the ecosystems they share with other pollinators. These concerns have prompted the international community to coordinate global pollinator conservation efforts and to call for research into the economic value of bees and into the economic impact of declines in the population of pollinators. As Thailand's agriculture is also rapidly converting to the production of cash crops under intensive farming systems, this development, in combination with the loss, fragmentation and degradation of natural habitats, threatens its rich bee fauna, which includes eight indigenous *Apis* species and a great diversity of stingless bees, especially from the genus *Trigona*. In fact, according to accounts from farmers growing pollinator-dependent crops in Eastern Thailand (a region characterized by intensive fruit farming), pollinator declines have already negatively affected their yields, which they first mitigated by renting honeybee hives and later by managing stingless bees themselves. In contrast, we discovered that in Northern Thailand (a region with a comparatively less degraded forest cover), beekeepers tend to pay longan farmers a fee for the exclusive right to bee forage on their farms; longan (*Dimocarpus longan*) is a fruit tree that produces valuable honey.

The *first article* of this cumulative thesis addresses the diversity of beekeeper-farmer interactions that, until now, could not be accommodated within the existing microeconomic models concerned with this issue. The most prominent of such theoretical models is dedicated to describing the determinants of colony stocking densities and of equilibrium wages that, for decades, have been paid by farmers to commercial beekeepers in the Californian almond monocultures, which are highly reliant on bee-mediated pollination. We generalized this basic model by taking into account the marginal productivity of a given agro-ecosystem's wild bees and the opportunity costs that farmers incur when assigning labor time to beekeeping. In that regard, we assessed the economic potential of on-farm beekeeping (which can involve several bee species) by juxtaposing this activity's net benefits from crop pollination and hive products against those from hiring commercial pollination services. In addition to serving as a classification tool for a plurality of farmer-beekeeper-nature interactions and related optimization problems, the resulting analytical framework helps identifying the institutional settings that are most likely to lead to a specific bioeconomic equilibrium supply of pollination. What is more, it illustrates the interplay of the pertinent economic and agro-ecological factors, thus assisting the postulation of empirically testable hypotheses.

We also conducted two separate discrete choice experiments (DCEs) with orchardists from the provinces of Chiang Mai (N = 198 respondents) and Chanthaburi (N = 127), in order to elicit their preferences for

changes in the population of local wild bees that would hypothetically result from a conservation policy consisting (along with a per-household implementation fee) of at least one of the following three measures: *(i)* offering farmers bee-friendly alternatives to conventional agro-chemicals, *(ii)* enabling the protection and/or rehabilitation of natural bee habitats near cropland, and *(iii)* fostering the husbandry of native bee species by transferring technical knowledge on the practice of on-farm beekeeping.

In this context, for the *second article* we fitted random parameter logit (RPL) models on the Chiang Mai DCE dataset. They yielded a significant willingness to pay (WTP) for the presented conservation measures and suggested that the disutility the respondents perceived for a 50% decline in the local population of native bees was greater than the utility they would derive from experiencing a bee population increase of the same magnitude, thus hinting at *loss aversion*. Moreover, comparing our aggregated WTP estimates to the expected production losses (as calculated with a realistic 60% dependence on bee pollination), showed that longan farmers underestimated the true use value of pollination. On the other hand, the average WTP for all conservation measures combined by far exceeded the costs that, according to our calculations, each household would incur for such a project to be implemented. Our RPL models also indicated a significant preference heterogeneity in the sampled population, which we could partly explain with idiosyncratic variables such as gender and the respondents' attitudes towards native bees and beekeeping.

Finally, in the *third article* we further examined the sources of randomness in the observed choice behavior, by modelling the unknown choice decision-relevant influences that could not be captured during the DCEs. To that end, we fitted generalized mixed logit (GMXL) models on the pooled datasets, which allowed comparing, on a common utility scale, the part-worth (value) estimates from Chiang Mai and Chanthaburi, where different experimental designs were applied. Our results reveal that farmers in Chanthaburi, who reported having experienced crop declines that they attributed to insufficient pollination, introduced less subjective factors into their choices than their Chiang Mai counterparts, who may have been less familiar with the importance of conserving bees. Moreover, the GMXL results also suggest that Chanthaburi farmers placed a significantly higher value on the above-mentioned measures *(i)* and *(ii),* while caring comparatively less about a 50% decline in local wild bee colonies. One can thus hypothesize that an actual local pollinator decline may have made Chanthaburi farmers more aware of the importance of conserving native bees, while paradoxically making them more independent from the provision of wild pollination services, as they started managing crop pollination with stingless bees.

As depicted in our analytical framework and econometrically exemplified in two regions of Thailand, many regions of the world may draw net benefits from optimizing their agro-ecological conditions with regard to crop pollination. In that respect, an assessment of the social profitability of conserving wild bees in ecologically critical cases is advisable and expected to result in conservation arguments for the protection of wild pollinators, their habitats and related ecosystems services, and of the biodiversity that supports them.

ZUSAMMENFASSUNG

Berichte über einen weltweiten Bestandsrückgang bei wildlebenden und domestizierten Bienen, insbesondere in Europa und Nordamerika, sowie die einhergehenden schädlichen Effekte einer intensiven Landwirtschaft (vor allem in Verbindung mit der übermäßigen Nutzung synthetischer Pestizide) und die resultierenden Ertragseinbußen, haben in den letzten Jahrzenten zunehmend Anlass zur Sorge gegeben. Obwohl die meisten Berichte die Europäische Honigbiene (*Apis mellifera*) und Hummeln betreffen, handelt es sich bei diesen Arten oft um Indikatoren für den allgemeinen Gesundheitszustand der Lebensräume, die sie mit anderen Bestäubern teilen. Aus diesem Grund hat die internationale Gemeinschaft angeregt, globale Anstrengungen zum Schutz bestäubender Insekten zu koordinieren und zum ökonomischen Wert von Bienen sowie den wirtschaftlichen Auswirkungen eines Verlustes an Bestäubungsleistungen zu forschen.

Da Thailands Landwirtschaft sich in einem raschen Wandel hin zur vermehrten Erzeugung sogenannter “cash crops” in intensiven Bewirtschaftungssystemen befindet, ist davon auszugehen, dass diese Entwicklung, in Kombination mit dem Verlust, der Fragmentierung und Schädigung natürlicher Habitate, die artenreiche Bienenfauna, die acht einheimische Honigbienenarten und zahlreiche stachellose Bienen (vor allem aus der Gattung *Trigona*) umfasst, massiv bedroht. So berichten Landwirte aus Ostthailand – eine Region die durch intensiven Obstbau gekennzeichnet ist –, dass Bestäuberverluste bereits zu Ertragsrückgängen bei bestäubungsabhängigen Kulturen geführt hätten. Dieses Problem wurde von den Betroffenen zunächst durch die Anmietung von Bienenstöcken, später dann durch die eigene Haltung stachelloser Bienen abzumildern versucht. Im Gegensatz hierzu konnten wir für Nordthailand – eine Region mit vergleichsweise großen und zusammenhängenden Waldgebieten – feststellen, dass dort die Landwirte dazu neigen, von den Imkern eine Gebühr für das Aufstellen von Bienenstöcken in Longan-Plantagen (*Dimocarpus longan*) zu verlangen. Dabei ist zu bedenken, dass Longan-Nektar einen sehr wertvollen Honigertrag erbringt.

Der *erste Artikel* dieser kumulativen Dissertation behandelt mögliche Imker-Landwirte-Interaktionen, die bisher nur zum Teil mit den hierfür entwickelten mikroökonomischen Modellen analysiert werden konnten. Das bekannteste dieser theoretischen Modelle bezieht sich auf die hochgradig von der Bienenbestäubung abhängigen kalifornischen Mandel-Monokulturen, in denen seit Jahrzenten kommerzielle Imker für Bestäubungsleistungen entlohnt werden. Das Modell analysiert die Bestimmungsgründe für die Besatzdichten an Honigbienen sowie für die Gleichgewichtslöhne der Bestäubungsimker. In dieser kumulativen Dissertation wird dieses Modell erweitert, indem zusätzlich *(1)* die Grenzproduktivität von Wildbienen in einem Agrarökosystem sowie *(2)* die Opportunitätskosten derjenigen Landwirte, die sich selbst der Bienenhaltung widmen, berücksichtigt werden. In diesem Zusammenhang wurde das ökonomische Potenzial, das sich für Landwirte aus der eigenen Haltung (u. U.

von verschiedenen Arten) von Bienen ergibt, explizit berücksichtigt und dabei den Nettonutzen aus den Bestäubungsleistungen und den anfallenden Imkereiprodukten dem Nettonutzen aus „zugekauften“ Bestäubungsleistungen gegenübergestellt. Der von uns entwickelte analytische Rahmen dient einerseits zur Klassifizierung der vielfältigen Beziehungen zwischen Landwirtschaft, Imkerei und Natur sowie den entsprechenden Optimierungsproblemen und hilft andererseits bei der Identifizierung institutioneller Lösungen, die jeweils geeignet erscheinen, ein bioökonomisches Gleichgewicht für ein optimiertes Angebot an Bestäubungsleistungen herbeizuführen. Darüber hinaus stellt er das Zusammenspiel der relevanten agrarökologischen und ökonomischen Einflussfaktoren dar und ermöglicht damit, empirisch überprüfbare Hypothesen abzuleiten.

Neben diesen theoretischen Analysen haben wir zwei voneinander unabhängige sogenannte “Discrete Choice”-Eperimente (DCE) mit Obstbauern in den thailändischen Provinzen Chiang Mai (*N* = 198 Teilnehmer) und Chanthaburi (*N* = 127) durchgeführt, um deren Präferenzen hinsichtlich der Veränderungen der örtlichen Wildbienenpopulation zu ermitteln. Diese Veränderungen können sich jeweils aus hypothetischen Kombinationen von Teilnahmegebühren und den folgenden Naturschutzmaßnahmen ergeben: *(i)* Angebot von bienenschonenden Alternativen zum konventionellen chemisch-synthetischen Pflanzenschutz, *(ii)* Ermöglichung von Schutz und/oder Wiederherstellung natürlicher Bienenhabitate auf den Plantagen sowie den angrenzenden Flächen, und *(iii)* Förderung der Haltung einheimischer Bienenarten auf den eigenen landwirtschaftlichen Flächen durch Wissenstransfer zur Bienenhaltung.

In diesem Zusammenhang, haben wir im *zweiten Artikel* “Random Parameter Logit (RPL)”-Modelle für das in Chiang Mai durchgeführte DCE geschätzt. Diese ergaben signifikante Zahlungsbereitschaften (WTP) der Befragten für die oben genannten Maßnahmen sowie einen dem 50%-igen *Rückgang* der Bienenpopulation beigemessenen negativen Nutzen, der dem Betrag nach größer war als der mit einer entsprechenden Populations*zunahme* einhergehende Nutzen, was auf *Verlust-Aversion* hindeutet. Ein Vergleich der aggregierten WTP-Schätzung mit den realistischerweise zu erwartenden Ertragseinbußen, die anhand einer 60%-igen Abhängigkeit des Ertrags von der Bienenbestäubung errechnet wurden, zeigte zudem, dass der wahre Wert des Bestäubungsnutzens von den befragten Longan-Anbauern unterschätzt wird. Außerdem zeigten auf den Modellergebnissen beruhende Berechnungen, dass die durchschnittliche Zahlungsbereitschaft für das Bündel aller drei Schutzmaßnahmen die für den einzelnen Haushalt kalkulierten Teilnahmekosten im Fall einer Projektumsetzung bei weitem übertreffen. Unsere RPL-Modelle haben darüber hinaus eine signifikante Präferenzheterogenität bei den Befragungsteilnehmern ergeben, die wir teilweise mit idiosynkratischen Variablen, wie z. B. dem Geschlecht und der Einstellung der Befragten zu einheimischen Bienen und zur Imkerei, erklären konnten.

Im *dritten Artikel* sind wir schließlich den möglichen Ursachen der Zufallskomponente des beobachteten Wahlverhaltens nachgegangen, indem wir die unbekannten entscheidungsrelevanten Einflüsse, die wir in unseren beiden DCE nicht abbilden konnten, modelliert haben. Zu diesem Zweck wurden sogenannte “Generalized Mixed Logit (GMXL)”-Modelle für den aus beiden Regionen zusammengefügten (“pooled”) Datensatz geschätzt. Dies ermöglichte den Vergleich der Teilnutzenschätzer für Chiang Mai mit denen für Chanthaburi auf einer gemeinsamen Nutzenskala, obwohl in Chiang Mai ein anderes experimentelles Design verwendet wurde. Unsere Ergebnisse zeigen, dass die Landwirte in Chanthaburi, die bereits von Ertragseinbußen wegen (nach ihrer Ansicht) unzureichender Bestäubung berichteten, weniger subjektive Faktoren in ihre Wahlentscheidungen einfließen ließen als ihre Kollegen in Chiang Mai. Letztere scheinen insgesamt weniger mit der Bedeutung des Bienenschutzes vertraut gewesen zu sein. Außerdem legen die GMXL-Ergebnisse nahe, dass die Landwirte in Chanthaburi einen signifikant höheren Wert auf die obigen Maßnahmen *(i)* und *(ii)* legen, während sie dem 50%-igen Rückgang der örtlichen Wildbienenpopulation einen relativ geringeren Wert beimessen. Daher kann man die Hypothese vertreten, dass es dort in der Vergangenheit bereits zu einem Rückgang lokaler Bestäuber gekommen ist und die Landwirte in Chanthaburi sich deshalb der Bedeutung des Schutzes einheimischer Bienen stärker bewusst sind. Die in diesem Zusammenhang paradox anmutende geringere Gewichtung eines Populationsrückgangs mag sich dadurch erklären, dass die Landwirte bereits damit begonnen haben, die Bestäubung mit Hilfe stachelloser Bienen zu bewerkstelligen, was sie von den Bestäubungsleistungen wilder Bienen unabhängiger macht.

Es ist davon auszugehen, dass weltweit viele weitere Regionen hinsichtlich der Kulturpflanzenbestäubung von einer Optimierung der agrarökologischen Verhältnisse profitieren können, ähnlich wie wir dies zunächst anhand des entwickelten Analyserahmens und schließlich durch ökonometrische Studien im Detail für zwei thailändische Regionen zeigen konnten. In diesem Kontext ist die systematische Erhebung und Bewertung des volkswirtschaftlichen Nutzens von Wildbienen insbesondere in ökologisch problematischen Fällen zu empfehlen. Dies dürfte in vielen Fällen überzeugende Argumente für den Schutz wilder bestäubender Insekten, ihrer Habitate und der einhergehenden Ökosystemleistungen erbringen und damit allgemein den Erhalt der Biodiversität befördern.

AUTHOR’S DECLARATION

Affidavit

pursuant to Sec. 8(2) of the University of Hohenheim’s doctoral degree regulations for Dr.sc.agr.

1. I hereby declare that I independently completed the doctoral thesis submitted on the topic

” The value of private and public goods from agro-forest ecosystems and native pollinators: the case of selected rural communities in Northern and Eastern Thailand”.

2. I only used the sources and aids documented and only made use of permissible assistance by third parties. In particular, I properly documented any contents which I used - either by directly quoting or paraphasing - from other works.

3. I did not accept any assistance from a commercial doctoral agency or consulting firm.

4. I am aware of the meaning of this affidavit and the criminal penalties of an incorrect or incomplete affidavit.

I hereby confirm the correctness of the above declaration. I hereby affirm in lieu of oath that I have, to the best of my knowledge, declared nothing but the truth and have not omitted any information.

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(Stuttgart-Hohenheim, April 4, 2018)