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Doctoral dissertation submitted to obtain the degree of
Doctor of Business Economics, to be defended by

Silvie Daniels

DOCTORAL DISSERTATION

A methodological framework for the valuation of biodiversity, based on the ecological function of species in the ecosystem.

Promoter: Prof. Dr Ir Steven Van Passel
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Dr Nele Witters

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A Methodological Framework for the Valuation of Biodiversity based on the Ecological Function of Species in the Ecosystem

PhD dissertation for the degree of doctor in applied economics

Silvie Daniels

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Center for Environmental Sciences (CMK)

Faculty of Business Economics

Research group Environmental Economics

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prof. dr. Robert Malina

dr. Dries Maes

dr. Hilde Eggermont

Ignace Schops

"*Humankind still has a lot to learn about the value of nature and the nature of value*"

(The Economics of Ecosystems and Biodiversity, 2010)

Dankwoord

During the course of my final master year at Ghent University, it became very clear to me that I gained enormous satisfaction out of applying economic theories for ecological purposes, in the interest of people, society and conservation. Everyone, the scientific community, world leaders and the media, have picked up upon the issue with regards to climate change that we are faced with today. However, the current rate of biodiversity loss as the cornerstone of the provisioning of essential and vital ecosystem services, all in the interest of human well being, has been proclaimed by scientists worldwide as an even bigger threat than climate change. Why has climate change managed to capture the attention to a much greater degree than biodiversity conservation? The difference between that climate change science has been able to convince political leaders and the general public that first of all, human actions did cause climate change, and second, that climate change will negatively affect human welfare in a fundamental way. For biodiversity loss, many scientists worldwide are working on establishing these links and providing irrefutable evidence so as to reverse the current trend of rapid and unforeseen losses. As I have been for the past six (or maybe even seven) years, so thank you to the university, the faculty of Business Economics, Jaco and Steven for providing this opportunity.

It is for all these reasons and more that I have absolutely loved working on a PhD that matters, at Hasselt university, an organization that has shown to genuinely believe in their employees. At a time in my life, when it mattered most to me and Benny, the university showed that it really cared, in more ways than one. It made me feel respected and made me want to work even harder when I continued my PhD after having been absent for a year.

The university is also the employer of many wonderful colleagues, old and new. So first and foremost, dankjewel Steven. Ik heb het genoegen gehad om ondertussen zes jaar met je samen te werken. Je geeft leiding op een natuurlijke ongedwongen manier die veel respect afdwingt bij de mensen rondom je. Je bent er als er problemen zijn, reikt oplossingen aan waar nodig en laat iedereen zijn eigen weg zoeken op een zodanige manier dat je verantwoordelijke en zelfstandige medewerkers creëert. En dit alles met inzicht en kennis van erg uiteenlopende onderwerpen. Heel

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Introduction

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CHAPTER 1: introduction

1.1 Abstract

An important issue in biodiversity valuation is gaining a better understanding of how biodiversity conservation contributes to economic activities and human welfare. However, quantifying the economic benefits of biodiversity for human wellbeing is not straightforward. In this chapter, first, the importance of biodiversity for human welfare is exemplified. The changing policy agenda and the multitude of initiatives taken at the global level to conserve biodiversity indicate the global sense of urgency.

Next, the use of Cost Benefit Analysis for the inclusion of environmental and social aspects into decision-making is analyzed. Since many effects of interventions or policies are outside the market, the need for monetary valuation techniques of non-marketed goods is established.

Finally, the current valuation techniques for biodiversity are identified and evaluated and the main problems for monetary valuation of biodiversity are assessed. This provides the context and rationale for the introduction of a methodological framework for the valuation of biodiversity based on the ecological role of the species, as captured in the central research question and sub questions.

1.2 Biodiversity and the global policy agenda

The term "biodiversity" or "biological diversity" is **defined** as "*the variety of life on Earth, including all organisms, species, and populations; the genetic variation among these; and their complex assemblages of communities and ecosystems*" (United Nations Environment Programme, 2010). Three levels of biodiversity are distinguished: (i) genetic diversity is all the different genes contained in all the living species, including individual plants, animals, fungi, and microorganisms, (ii) species diversity is all the different species, as well as the differences within and between different species and (iii) ecosystem diversity is all the different habitats, biological communities and ecological processes, as well as variation within individual ecosystems.

Biodiversity plays a key role in ecological processes and the delivery of ecosystem services, and its importance has been widely recognized (MA, 2005). The Millennium Ecosystem Assessment (MA) assessed the consequences of ecosystem change for human well being, providing a state-of-the-art scientific appraisal of the condition and trends in the world's ecosystems and the services they provide, as well as the scientific basis for action to conserve and use them sustainably. UNEP (2010) stated that: " ***biodiversity conservation provides substantial benefits to meet immediate human needs****, such as clean, consistent water flows, protection from floods and storms and a stable climate. The loss of biodiversity is dangerous and its consequences are immediate"* (United Nations Environment Programme, 2010).

With the adoption of the Sustainable Development Goals (SDGs) on September 25th 2015, **biodiversity is recognized as one of the key aspects for the achievement of sustainable development**. In SDG14 (life below water), the protection of marine habitats is acknowledged as an important factor in poverty reduction by its contribution to people's income through increasing fish catches and improving health. Not only do they provide resources such as food and medicine but they also drive breakdown and removal of pollution, protect coastal communities from storm damage and mitigate climate change[1].

SDG15 (Life on land) aims to sustainably manage forests, combat desertification, halt and reverse land degradation and halt biodiversity loss. Globally, around 1.6 billion people depend on forests for their livelihood. Forests produce the oxygen we breathe, purify the water we drink and provide an important risk reduction strategy to combat climate change, thereby increasing the resilience of people to the impacts of climate change. They also serve as important areas for recreation and mental well-being[2]**.**

Recognition was given to the contribution of biodiversity for the successful implementation of the SDGs. In it's technical note "Biodiversity and the 2030 Agenda for sustainable development", the Convention on Biological Diversity (CBD) intends to expand the knowledge on the contribution of biodiversity to achieving the SDGs and states that "the SDGs and the Strategic Plan for biodiversity are mutually supportive and reinforcing, and therefore the implementation of one contributes to the achievement of the other" (CBD, 2016). The note highlights the contribution of the Aichi

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[1] http://www.un.org/sustainabledevelopment/oceans/ (last accessed September 30, 2017)
[2] http://www.un.org/sustainabledevelopment/biodiversity/ (last accessed September 30, 2017)

Biodiversity Targets to all SDGs and offers a summary of the linkages. "Biodiversity and healthy ecosystems provide the essential resources and ecosystem services that directly support a range of economic activities, such as agriculture, forestry, fisheries and tourism" and in doing so "…provide livelihoods for many of the world's rural poor". Also, "biodiversity is a key factor for the achievement of food security and improved nutrition. All food systems depend on biodiversity and a broad range of ecosystem services that support agricultural productivity, soil fertility, and water quality and supply. Furthermore, at least one-third of the world's agricultural crops depend upon pollinators." Biodiversity contributes directly and indirectly to healthier lives by reducing environmental risk. Healthy ecosystems contribute to clean drinking water, by underpinning the delivery of water supplies, water quality, and guard against water-related hazards and disasters.

The CBD that gave rise to the **Aichi Biodiversity Targets** as the strategic plan for the period 2011-2020, was erected in order to conserve the Earth's biological resources and address threats to species and ecosystems. In November 1988, the United Nations Environment Programme (UNEP) started exploring the need for an international convention on biological diversity (CBD) June 5th 1992 marked the date on which 168 parties at the United Nations Conference on Environment and Development (the Rio "Earth Summit") signed the convention[3]. As early as 1995, the CBD adopted the ecosystem approach as the primary framework for addressing the objectives of the Convention, being: the conservation of biodiversity, the sustainable utilization of component species; and the fair and equitable sharing of genetic resources and resultant benefits. In 2002, world leaders committed through the Convention on Biological Diversity (CBD) to halt the rate of biodiversity loss by 2010, and declaring 2010 the International Year of Biodiversity. On 22 December 2010, the United Nations declared the period 2011-2020 as the UN Decade on Biodiversity. The commitment expressed in the CBD in 2002 was further extended in the Aichi Biodiversity Targets with the mission **to *"take effective and urgent action to halt the loss of biodiversity in order to ensure that by 2020 ecosystems are resilient and continue to provide essential services***, *thereby securing the planet's variety of life, and contributing to human well-being, and poverty eradication. To ensure this, pressures on biodiversity are reduced,*

[3] www.cbd.int (last accessed May 27 2017).

ecosystems are restored, biological resources are sustainably used and benefits arising out of utilization of genetic resources are shared in a fair and equitable manner, adequate financial resources are provided, capacities are enhanced, biodiversity issues and values mainstreamed, appropriate policies are effectively implemented, and decision-making is based on sound science and the precautionary approach." Two targets are of particular importance for the analyses described here:

- target 1 (strategic goal A) states that: *"By 2020, at the latest, people are aware of the values of biodiversity and the steps they can take to conserve and use it sustainably. Understanding, awareness and appreciation of the diverse values of biodiversity help to underpin the willingness of individuals to make such changes. Public awareness also underpins the political will for governments to act. Meeting this target requires that people are aware not only of the values of biodiversity in an abstract way, but know the concrete contributions of biodiversity to their lives".*

- target 19 (strategic goal E) states that: *"By 2020, knowledge, the science base and technologies relating to biodiversity, its values, functioning, status and trends, and the consequences of its loss, are improved, widely shared and transferred, and applied."*

The importance of biodiversity for underpinning the delivery of ecosystems services comes through the provisioning of ecosystem functions. For example, natural predators perform important biological pest control services, thereby reducing crop damages and indirectly contributing to farmer's income (Daniels et al., 2017) or aquatic invertebrates low on the aquatic food chain providing indirect use values to the fishing industry, that depends on selling the fish that eat them (Benjamin and Bellmore, 2016). The first paper to value the world's ecosystem services was by Costanza et al. (1997), with values -most of which are outside the market- estimated to be in the range of US$16-54 trillion per year (Costanza et al., 1997).

Soil biota for example also play a large role in the regulation of many of the processes occurring in soils such as decomposition of organic matter, nutrient cycling, bioturbation, and suppression of soil borne diseases and pests (Wall and Nielsen, 2012). Current estimates of the contribution of soil biota to ecosystem services provided by soils globally range from 1.5 to 13 trillion US Dollars annually (Van der Putten, 2004).

Many authors recognize that biodiversity is crucial for ecosystem stability and long-term resilience of the ecosystem functions and services that they underpin (Cleland, 2011; Oliver et al., 2015; Perz et al., 2013; Sundstrom et al., 2012). Cleland et al. (2011) exemplified the importance of species diversity for primary production and ecosystem stability as follows:

"Species play essential roles in ecosystems, so local and global species losses could threaten the stability of the ecosystem services on which humans depend (McCann 2000). For example, plant species harness the energy of the sun to fix carbon through photosynthesis, and this essential biological process provides the base of the food chain for myriad animal consumers. At the ecosystem level, the total growth of all plant species is termed primary production, and — as we'll see in this PhD — communities composed of different numbers and combinations of plant species can have very different rates of primary production. This fundamental metric of ecosystem function has relevance for global food supply and for rates of climate change because primary production reflects the rate at which carbon dioxide (a greenhouse gas) is removed from the atmosphere. There is currently great concern about the stability of both natural and human-managed ecosystems, particularly given the myriad global changes already occurring. Stability can be defined in several ways, but the most intuitive definition of a stable system is one having low variability (i.e., little deviation from its average state) despite shifting environmental conditions. This is often termed the resistance of a system" (Cleland, 2011).

Resilience can be defined as the ability of an ecosystem to return to its original state following a disturbance or other perturbation. The insurance hypothesis explains that with increasing biological diversity, chances of the presence of species having traits that enable them to adapt to a changing environment increase (Yachi and Loreau, 1999) In this situation, species identity — and particular species traits — are the driving force stabilizing the system rather than species richness per se (Cleland, 2011).

In spite of the global and undisputed importance of biodiversity conservation for every aspect of human well-being, evidence shows that the general trend is negative but that progress is under way.

In 2009, **Rockström et al.** identified the loss of biodiversity as the most important threat to the resilience of our ecosystems and human welfare. Species are becoming extinct at a rate that has not been seen since the last global mass-extinction event (Rockström et al., 2009).

Also, in 2010, **Butchart et al.** compiled 31 indicators to measure progress on the state of biodiversity (e.g. species' population trends, extinction risk, habitat extent/condition, and community composition) and concluded that, in spite of global actions, biodiversity is declining at an alarming rate. Their results show declines with no significant recent reductions in rate, whereas the indicators of pressures on biodiversity (e.g. resource consumption, invasive alien species, nitrogen pollution, over-exploitation, and climate change impacts) showed increases (Butchart et al., 2010).

In 2014, the **Global Biodiversity Outlook** identified that progress had been made towards meeting some components of the Aichi Biodiversity Targets. "*Some target components, such as conserving at least 17 per cent of terrestrial and inland water areas, are on track to be met.*" However, the report equally confirmed the results from Butchart et al. (2010) in that "…*extrapolations for a range of indicators suggest that based on current trends, pressures on biodiversity will continue to increase at least until 2020, and that the status of biodiversity will continue to decline. Despite the fact that society's responses to the loss of biodiversity are increasing dramatically, and based on national plans and commitments are expected to continue to increase for the remainder of this decade*". Therefore, "…*in most cases the progress will not be sufficient to achieve the targets set for 2020, and additional action is required to keep the Strategic Plan for Biodiversity 2011–2020 on course."* (CBD, 2014).

The **2017 report of the SDGS** puts forward a general message of decreasing biodiversity and a growing extinction risk. However, there is substantial variation between different species groups. The IUCN (International Union for the Conservation of Nature) Red List Index paints a more positive picture for birds and mammals as the result of effective conservation actions. In the same positive trend, between 2000 and 2017, the percentage of Key Biodiversity Areas that are protected has risen significantly – from 35 to 47% for terrestrial protected areas with percentages varying widely around the world (UN, 2017).

Many more initiatives have been initiated to counter the negative trend of biodiversity loss. In 2012, the **Intergovernmental science-policy Platform on Biodiversity and Ecosystem Services (IPBES)** was established as an independent body in a collaborative partnership with four UN agencies: UN Environment, United Nations Development Programme (UNDP), United Nations Educational Scientific and Cultural Organization (UNESCO) and the Food and Agricultural Organization (FAO), to "*assess available knowledge from multiple disciplines to better inform decision-making in response to requests from member States".* IPBES also claims to become *"the leading scientific body for assessing the state of the planet's biodiversity and ecosystems, as well as the essential contributions they make to people"*[4].

In January 2015, the launch of four regional assessments on knowledge of biodiversity and ecosystem services was approved. In the **regional assessments** that will be presented to the 6th session of the IPBES Plenary in March 2018, **"…***the status and trends regarding biodiversity, ecosystem functions and ecosystem services and their interlinkages, the impact of biodiversity, ecosystem functions and ecosystem services and threats to them on good quality of life, and the effectiveness of responses, including the Strategic Plan for Biodiversity 2011–2020 and its Aichi Biodiversity Targets, the Sustainable Development Goals, and the National Biodiversity Strategies and Action Plans developed under the Convention on Biological Diversity"* are assessed*. "The assessments will address terrestrial, freshwater, coastal and marine biodiversity, ecosystem functions and ecosystem services. The overall objective of the regional assessments is to strengthen the science-policy interface on biodiversity, ecosystem functions and ecosystem services at the regional and sub regional level."* [5]

Also, IPBES is to perform **assessments at the global level**. "*The overall scope of the assessment is to assess the status and trends with regard to biodiversity and ecosystem services, the impact of*

[4] www.ipbes.net (last accessed May 5, 2017)

[5] https://www.ipbes.net/deliverables/2b-regional-assessments (last accessed October 23, 2017)

biodiversity and ecosystem services on human well-being and the effectiveness of responses, including the Strategic Plan and its Aichi Biodiversity Targets. It is anticipated that this deliverable will contribute to the process for the evaluation and renewal of the Strategic Plan for Biodiversity and its Aichi Biodiversity Targets."[6]

Within the context of biodiversity protection and the improvement of sustainable livelihoods, **nature-based solutions (NBS)** have been put forward as an alternative approach to technological innovation in managing socio-ecological systems. The term refers to the sustainable use of nature in solving societal challenges in which ecosystems can provide solutions for the benefit of biodiversity, human well-being and society at large. It is considered an umbrella concept to include concepts such as e.g. natural solutions, ecosystem-based approaches, green infrastructure and ecological engineering (Eggermont, 2015; IUCN, 2016). In 2015, the term was adopted by the European Commission (EC) as "solutions that aim to help societies address a variety of environmental, social and economic challenges in sustainable ways" and are "… inspired by, supported by or copied from nature." The goals of the research and innovation agenda on NBS include: (i) enhancing sustainable urbanization, (ii) restoring degraded ecosystems, (iii) developing climate change adaptation and mitigation and (iv improving risk management and resilience. Since 2016, the EU is supporting science-policy-business-society dialogue to promote the co-design, testing and deployment of improved and innovative NBS to promote the market uptake of NBS (Nikolaidis et al., 2017). As an example, Europe's ThinkNature[7] project aims to promote NBS across research, policy, non-governmental organizations and business.

As described here, many initiatives have been introduced over the years in order to protect biodiversity and the benefits associated with it. At the policy level, decisions on interventions or policies are based on the appraisal of the costs and benefits associated with the project or policy. A universally accepted tool is Cost Benefit Analysis (CBA) outweighing the costs and benefits of an intervention or policy. In the next paragraph, the use of CBA to include environmental and social aspects into decision-making is addressed.

[6] https://www.ipbes.net/deliverables/2c-global-assessment (last accessed October 23, 2017)
[7] https://www.think-nature.eu/ (last accessed October 26, 2017)

1.3 Cost Benefit Analysis for Natural resource Management

In order to appraise an investment decision and assess the welfare changes attributable to it, a **Cost Benefit Analysis (CBA)** can be employed. A CBA is a methodology which facilitates the selection of projects and policies which are efficient in terms of resource use and aims to demonstrate the convenience for society of a particular intervention rather than possible alternatives (EU, 2015). As such, it is a systematic process for identifying, valuing and comparing the costs and benefits of an intervention. In welfare economics, the aim is to improve the Pareto efficiency of the economy, meaning that the project or intervention makes people better off, without making others worse off. In practice, almost all projects have winners and losers, and therefore, a variation to the Pareto efficiency criterion is the Kaldor-Hicks compensation principle, which asks whether those that gain from an intervention could compensate the losers and still remain better off after the compensation. For a description of the different steps in a CBA, we refer to the *guide to cost-benefit analysis of investment projects* (EC, 2015).

Although CBA is widely accepted and practiced, Pearce (2006) identified several issues with CBA such as the distribution of costs of benefits amongst the population, the uncertainty over discounting future values and the lack of a sustainability criterion. In the context of this analysis, the focus in the next paragraphs will be placed on the uncertainty with which the costs and benefits of the projects can be estimated and the accuracy and acceptability of monetary valuations.

An impact arising from a project or an intervention is included in the CBA if either utility or production levels are affected by the impact. All the positive and negative impacts of a proposed project or policy are then valued in monetary terms. In addition to the difficulties of forecasting all cost and benefit flows over the lifespan of the project, an additional difficulty arises from the environmental and social impacts that are not traded and therefore have no explicit market price (Hanley, 1992).

"Not taking into account environmental impacts will result in an over- or underestimation of the social benefits of the project and will lead to bad economic decisions. In other words, the economic evaluation of the environment helps decision-makers to integrate into the decision-making process the value of environmental services provided by ecosystems. Direct and external environmental

effects must be expressed in monetary terms in order to integrate them into the calculation of homogenous aggregate CBA indicators of net benefits." (EC, 2015)

In practice, the fact that money is chosen as a value scale in CBA, means that those impacts that cannot easily be monetized are difficult to include in the CBA. Many of the social and environmental impacts arising from a project are not traded and therefore have no explicit market price.

Atkinson & Mourato (2015) describe the recent developments in environmental CBA since the publication of the OECD volume on this topic by Pearce et al. (2006). One noticeable development is the maturity of environmental valuation techniques. On the downside, this maturation is accompanied by fewer groundbreaking contributions in the area of economic valuation methodologies (Atkinson and Mourato, 2015).

With regards to the inclusion of economic values of biodiversity in CBA, Hanley et al. (1995) described that the values derived for biodiversity by means of stated preference techniques are influenced by the definition of biodiversity and the terminology and are therefore lexicographic rather than utilitarian. This results in decreased validation of CBA as a means of decision making for biodiversity protection, since lexicographic preferences are incompatible with the Kaldor-Hicks Compensation Test (Hanley et al., 1995).

In another example, Lehtonen et al. (2003) estimated the existence values of forest biodiversity, and used the benefit estimated based on a mail survey to value the forest conservation in southern Finland. The preliminary analysis produced fairly high willingness to pay (WTP) measures for increased conservation (Lehtonen et al., 2003). The results, however, indicated that due to preference uncertainty and respondents' willingness to support forest conservation even at a high level of personal costs, traditional welfare measures used in the preliminary analysis might differ from the actual willingness to pay.

The inclusion of environmental effects in CBA requires placing a monetary value on a change in supply of non-market goods such as clean air, clean water or biodiversity. Although methodologies to cope with such estimation requirements have recently seen improvements and gained wider acceptance, in practice, the inclusion of the economic values of biodiversity in CBA analysis is not universally applied and remains difficult due to the limitations of the current valuation methodologies for valuing complex concepts such as biodiversity. In the next paragraph, the limitations of the current valuation methodologies for the economic valuation of biodiversity are analyzed.

1.4 Economic valuation of biodiversity

When environmental service markets are available, the easiest way to measure economic value is to use the actual related market price. When there is no market, the price can be derived through non-market evaluation procedures. This is the case, for example, for biodiversity. Whilst the costs for biodiversity conservation incentives are generally well known, the quantification of economic benefits is less straightforward (Christie et al., 2006). The role of economic valuation is increasingly being recognized as an indispensible tool to target biodiversity protection with scarce budgets and in determining damages for losses of biodiversity in liability regimes (Christie et al., 2006; OECD, 2001). In their *Guide to Corporate Ecosystem Evaluation*, the WBCSD forthrightly declaim the promise of taking nature into account as a capital good like any other (Maier and Feest, 2016).

While the motivation for increased knowledge of the economic effects of biodiversity losses is clear, assessing the role of biodiversity on ecosystem services is not straightforward (TEEB, 2010a) and several key challenges predominate the scientific rhetoric: (i) the elicitation of values for biodiversity remains at best unclear, (ii) no established framework has been agreed upon that effectively assesses biodiversity losses for their effects on economic performances, (iii) ecological uncertainty remains on the link between species diversity and ecosystem services provisioning and (iv) biodiversity is a multi-dimensional concept and requires multiple proxies for measuring them.

1.4.1 The elicitation of values for biodiversity remains at best unclear

It is safe to assume that biodiversity has a large **indirect use value** to humans when it is considered as an input in a production function, thereby generating products or services that are used directly by humans. Many identify the need for direct market valuation techniques that can capture indirect use values through the use of a production function and a market price, whereby the contribution of biodiversity to certain ecosystem services and the production of marketable goods is quantified (Bertram and Rehdanz, 2013; Farnsworth et al., 2015; Laurila-Pant et al., 2015).

In their review of biodiversity valuation studies, Bartkowski et al. (2015) reveal that more than 80% of biodiversity valuation studies use **stated preference methods** and none of the studies consider alternatives to public preference valuations since they are rare in valuing biodiversity (Farnsworth et al., 2015). The values of goods and services exchanged on markets reveal an individual's willingness to pay (WTP) for their direct use. Ecosystem services are used indirectly by society and have no exchange markets to reveal their values and therefore the perceived economic values of ecosystem services are vastly subjective and context specific (Tallis and Kareiva, 2005). Nevertheless, they provide useful information for economic and environmental decision-making and inclusion in CBA. Very often, valuation studies take 'biodiversity conservation' or 'nature' as the object of valuation, rather than biodiversity in itself (Farnsworth et al., 2015).

The elicitation of values for biodiversity with the aid of stated preference methods suffers from the generally low level of awareness and understanding of what biodiversity means on the part of the general public (Bräuer, 2003; Christie et al., 2006). People may be poorly informed about the meaning of biodiversity, complicating the use of contingent valuation as a means of measuring preservation benefits. Moreover, willingness to pay for biodiversity protection increases with the level of information provided (Hanley et al., 1995). Similarly, Lehtonen (2003) showed that the estimation method and assumptions have significant effects on the WTP estimates and may therefore produce unrealistic results. The willingness-to-pay (WTP) for species that are unfamiliar or undesired by the general public could yield extremely low values despite the fact that these species could be performing indispensible ecological services and thereby indirectly contribute to the generation of welfare (Daniels et al., 2017).

This, combined with the complexity of biodiversity (Feest et al., 2010), might just overstretch the capacity of the usual stated preference valuation techniques for the valuation of biodiversity (Bartkowski et al., 2015). Economic valuation of biodiversity as defined in natural science - the quantification of the total difference between a biological system's part in terms of phylogenetic, structural and functional differences- is to date unfulfilled (Farnsworth et al., 2015). The use of this definition of biodiversity is rejected for being 'incomprehensible to the general public', and renders "*valuation by stated-preference methods, at best, very difficult*" (Farnsworth et al., 2015).

1.4.2 Lack of an established framework for the assessment of biodiversity losses and their effects on economic performance

The concept of ecosystem services has become an important model for linking the functioning of ecosystems to human well-being (Fisher et al., 2009). The **ecosystem approach** adopted as the primary framework for addressing the objectives of the CBD states that "*The ecosystem approach is a strategy for the integrated management of land, water and living resources that promotes conservation and sustainable use in an equitable way. Based on the application of appropriate scientific methodologies focused on levels of biological organization, it encompasses the essential structure, processes, functions and interactions among organisms and their environment. This approach recognizes that humans, with their cultural diversity, are an integral component of many ecosystems.*" And, "*The ecosystem approach requires adaptive management to deal with the complex and dynamic nature of ecosystems and the absence of complete knowledge or understanding of their functioning. Management must be adaptive in order to be able to respond to such uncertainties and contain elements of "learning-by-doing" or research feedback. Measures may need to be taken even when some cause-and-effect relationships are not yet fully established scientifically.*"[8]

Also, The Natural Capital Approach (NCA) promoted by the International Institute for Sustainable Development is a means for identifying and quantifying natural resources and associated ecosystem goods and services… "*that can help integrate ecosystem-oriented management with economic decision-making and development. By integrating economic and environmental imperatives, NCA operationalizes the ecosystem approach and facilitates policy-making for sustainable development. Born out of theoretical advancements in ecological economics, the NC concept is gaining considerable interest for devising policies that reconcile economic and environmental imperatives. Integrating the concept within economic and environmental*

[8] www.cbd.int/ecosystem/description.shtml (last accessed: May 27th 2017)

management systems is best achieved by treating the natural environment similarly to other forms of valued capital and adopting the ecosystem approach which is compatible with a wide range of contexts.” (Voora and Venema, 2008)

The **Ecosystem Services Cascade** introduced by Potschin and Haines-Young (2011), provides a cascade of consequent events leading to monetary valuation. The cascade starts from (i) Ecosystem Properties (EP) leading to (ii) Ecosystem Functions (EF), (iii) Ecosystem Services (ES), (iv) Benefits (B) and (v) Values (V) (Potschin and Haines-Young, 2011). Ecosystem Properties (EPs) are defined as the biophysical structure of the ecosystem, Ecosystem Functions (EFs) are ‘any change or reaction that occurs in an ecosystem (biophysical, chemical or biological)’, Ecosystem Services (ES) are the ‘contributions of ecosystems to human well-being’, Benefits (B) are ‘positive changes in well-being from the fulfillment of needs and wants’ and Value (V) is defined as the ‘economic worth of the change in well-being’ (TEEB, 2010).

According to Farnsworth (2015), the functional value of biodiversity can be found in four steps: the first quantifies the relation between biodiversity and the function, the second quantifies the contribution of the function to providing a service, the third determines the benefits experienced by certain stakeholders, the fourth and final step quantifies the value as expressed by the beneficiaries (Farnsworth et al., 2015).

Methodologies that provide a strong link between economic theory and ecological research (i.e **production function analogy** or cost-based methods) remain largely unexplored (Bartkowski et al., 2015; Farnsworth et al., 2015). Farnsworth et al. (2015) emphasize an urgent refocusing of economists for the economic valuation of biodiversity towards cost-based or production based methods. Furthermore, “*a biophysical method does not assume that value is determined by individual preferences, but rather attempts a more ‘objective’ assessment of ecosystem contributions to human welfare*” (Sagoff, 2011).

Strengthening a production-based method could be achieved by stressing the **functionality of biodiversity** in valuation studies, which constitutes a major research gap and the recent biodiversity valuation literature emphasizes that the ecological role biodiversity plays in human well-being should be at the center of valuation studies (Bakhtiari et al., 2014; Bartkowski et al., 2015; Daniels et al., 2017; Farnsworth et al., 2015). A loss of biodiversity may, both directly and indirectly, affect ecosystem function, service and human welfare (Chapin et al., 2000). As functional groups of species provide a link between species diversity and ecosystem function (Grimm, 1995; Bengtsson, 1998; McCann, 2000), functional groups are the main units to investigate the consequences of global environmental change on ecosystem function and the services delivered (Steffen et al., 1996; Diaz and Cabido, 1997; Woodward et al., 1997; Grime et al., 2000). Valuation methodologies taking into account the functional role of biodiversity are supported by consistent findings of meta-analyses and valuation studies confirming that indirect use values constitute the largest source of total economic value in biodiversity valuation (Costanza et al., 1997; de Groot et al., 2002; Farnsworth et al., 2015).

1.4.3 Uncertainty on the link between species diversity and ecosystem services provision

Biodiversity contributes directly (through provisioning, regulating, and cultural ecosystem services) and indirectly (through supporting ecosystem services) to many constituents of human well-being, including security, basic material for a good life, health, good social relations, and freedom of choice and actions (MEA, 2005). Wall & Nielsen (2012) explore the relationship between soil biodiversity and ecosystem services, and discuss why biodiversity might influence the rate and stability of ecosystem service provision. One of the key questions for maintaining continued provision of ecosystem services provided by soils and their biota is whether functioning depends on the number of species present (i.e., biodiversity), on key species, species traits (i.e., functional group, life-cycle and history, stress tolerance, etc.) or on the composition of the communities (Wall and Nielsen, 2012).

Theoretical models suggest that there could be multiple relationships between diversity and ecosystem stability, depending on how we define stability: "*Recent advances indicate that diversity can be expected, on average, to give rise to ecosystem stability. The evidence also indicates that diversity is not the driver of this relationship; rather, ecosystem stability depends on the ability for communities to contain species, or functional groups, that are capable of differential responses*" (Ives and Carpenter, 2007).

The positive link between biodiversity and ecosystem services may be represented by three relationships: (i) a linear relationship would occur if the addition of any new species enhances functioning, (ii) the redundancy relationship occurs if multiple species have the same influence on functioning and the addition of a new species only has a positive influence on functioning if it possesses a trait not already found in the community, and (iii) an idiosyncratic relationship indicates a system where species differ in their ability to enhance functioning, or where biotic interactions enhance (e.g. facilitation), or inhibit (e.g. competition), functioning. In this case the inclusion of a single rare species has a disproportionally large negative or positive impact on functioning and the overall community composition is therefore more important for functioning than species richness *per se* (Wall and Nielsen, 2012).

1.4.4 Biodiversity measurements require multiple proxies

Since biodiversity refers to diversity at multiple scales of biological organization (genes, populations, species, and ecosystems) and can be considered at any geographic scale (local, regional, or global), it is generally important to specify the level of organization and scale of concern (MEA, 2005) or "*More specifically, the diversity of genes, species, or ecosystems per se is often confused with a particular component of that diversity… The consequences of changes in biodiversity for people can stem both from a change in the diversity per se and a change in a particular component of biodiversity"*.

In their review of economic valuation studies of biodiversity, Bartkowski et al. (2015) propose that the selection of biodiversity proxies, as a consequence of its complexity, should not reduce biodiversity to one single aspect. " *A proper proxy should cover as many aspects and dimensions of biodiversity as possible, given the data, resources and other constraints*." A single component will not do the job: "no single component, whether genes, species or ecosystems is consistently a good indicator of overall biodiversity, as the components can vary independently" (MEA, 2005)

1.5 Central research question and sub questions

The biodiversity valuation literature currently experiences a number of gaps: (i) there is a lack of studies that use multiple indicators to represent biodiversity (ii) there is no agreed framework for the valuation of biodiversity, (iii) there is a lack of methodologies that effectively capture the ecological role of biodiversity on the delivery on ecosystem functions and services, and (iv) there is a lack of studies using market-based approaches for valuing biodiversity. Therefore, in this analysis, I would like to contribute to the construction of a methodological framework that effectively integrates the ecological role of species in an ecosystem, by (i) using a multi-attribute approach to characterize biodiversity (meaning that more than one attribute is used to describe biodiversity), (ii) integrating a dynamic ecological model with an economic model and (iii) integrating a production function technique with a market based valuation technique. In doing so, the development of such a framework could (i) show potential to contribute to the strategic goals as set out by the Aichi Biodiversity Targets (CBD, 2014), and (ii) provide support for objective policy making outweighing the costs and benefits of biodiversity conservation for inclusion in CBA. The ultimate aim is to provide quantifiable and objective measurements for the inclusion of biodiversity in policymaking and CBA.

Therefore, the **central research question** of this PhD is:

"Can a dynamic, multi-attribute methodological framework for the valuation of biodiversity be constructed, based on the ecological role of species in the ecosystem to reveal the indirect use value of biodiversity?"

In chapter 1, the importance of biological diversity is defined and the recent development of the global biodiversity policy agenda is examined. Also, the contribution of this research to important international targets (Aichi Biodiversity Targets) is framed. Chapter 1 examines the inclusion of biodiversity values in CBA and explores the state-of-the-art with regards to the methodologies currently employed for biodiversity valuation. It examines the obstacles with regards to valuation and assesses the need for a methodological framework based on the ecological role of species in the ecosystem. Therefore, **sub question 1** addressed in chapter 1 is: "**Which are the main challenges and motivations for the development of a methodological framework for the valuation of biodiversity, based on the ecological function of species in the ecosystem?**"

Chapter 2 sets out to build a methodological framework, based on existing frameworks and valuation methodologies, taking into account the recommendations from other authors. **Subquestion 2** being answered here is: **"Can a generic methodological framework be introduced that quantifies the indirect use value of changes in biodiversity?".** The methodological framework proposed quantifies the effects of changes in non–marketable species diversity for their impact on economic activities through the delivery of ecosystem services and attaches an indirect use value to species diversity. It integrates (i) a dynamic ecological model simulating interactions between species with (ii) an economic model assessing the effect of changes in species diversity for net revenues. The methodological framework both (i) quantifies the contribution of species diversity to net revenues through the use of a production function technique, and (ii) attributes an objective monetary value to species diversity by employing a direct market-based technique based on the changes in the provisioning of a marketable good in order to provide information for the inclusion of biodiversity into policy making.

Chapter 3 applies the framework through elaboration of a case study valuing the presence of natural predators for the biological pest control of pest insects in pear production in Flanders (BE). **Sub question 3** being answered here is: **" What is the indirect use value of natural predators for pear production in Flanders?".** The methodological framework is applied for the ecological role of a limited number of species and builds an integrated ecological-economic model to derive the indirect use value of changes in biodiversity.

Chapter 4 assesses whether the methodological framework proposed in chapter 2 can be used in different circumstances as compared to chapter 3: (i) for a larger number of species, (ii) for another ecosystem (freshwater river systems instead of an agricultural production system), and (iii) for another ecosystem service (salmon production instead of biological pest control). Chapter 4 also accounts for the contribution of the individual effects of changes in species richness, species composition and species abundance to determine the indirect use value of biodiversity.

Sub question 4 addressed in chapter 4 is: **" What is the indirect use value of aquatic macro-invertebrates for salmon production in the US North West?".**

Last, **chapter 5** starts by providing a summary of the objectives for this analysis. Next, the development of the methodological framework and the dynamic ecological model is reviewed critically and the potential for the framework to include the effect of management practices is discussed. Also, the use of market-based valuation techniques is discussed for its potential to value other functional groups. Finally, suggestions to improve the overall applicability and ease of implementation of the framework are discussed.

CHAPTER 2

Quantification of the Indirect Use Value of Functional Group Diversity based on the Ecological Function of Species in the Ecosystem

<u>This chapter is under review in:</u>

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Schröder, P.; Beckers, B.; Daniels, S.; Gnädinger, F.; Maestri, E.; Mench, M.; Millan, R.; Obermeier, M.; Oustriere, N.; Persson, T.; Poschenrieder, C.; Rineau, F.; Rutkowska, B.; Schmid-Sutter, T.; Szulc, W.; Witters, N.; Sæbø, A. Intensify production, transform biomass to energy and novel goods and protect soils in Europe – a vision how to mobilize marginal lands.

CHAPTER 2: Quantification of the Indirect Use Value of Functional Group Diversity based on the Ecological Function of Species in the Ecosystem

2.1 Abstract

An important issue in biodiversity valuation is gaining a better understanding of how biodiversity conservation contributes to economic activities and human welfare. However, quantifying the economic benefits of biodiversity for human well being is not straightforward. Here, we expand the ecosystem service cascade by (i) adding a stepwise methodological framework to the cascade to assess the effects of changes in functional group diversity on economic activities; (ii) including multiple attributes for defining functional diversity and (iii) integrating a dynamic ecological model simulating complex interactions and feedbacks between species with an economic model assessing the effects of changes in functional group diversity for gross revenues. The stepwise methodological framework integrates a production function approach with a market price-based approach in order to investigate the indirect use value of functional group diversity based on the ecological role of species in the ecosystem.

2.2 Introduction

Biodiversity plays a key role in ecological processes and the delivery of ecosystem services, and its importance has been widely recognized (MA, 2005). Most of the central issues facing conservation involve understanding the effects of economic activity on biodiversity and ecosystems, whilst finding solutions to conservation problems requires demonstrating the benefits of conservation for the wellbeing of people (Polasky, 2009). Whilst the costs for biodiversity conservation measures are generally well known, the quantification of economic benefits is less straightforward (Christie et al., 2006). Economic valuation is increasingly being recognized as an indispensible tool to target biodiversity protection (Polasky, 2009) with scarce budgets and to determine damages for losses of biodiversity in liability regimes (Christie et al., 2006; OECD, 2001) and it is believed to be a suitable means to facilitate their recognition, demonstration and consideration in decision making (Lienhoop et al., 2015).

At the same time, economic valuation is heavily criticized due to the lack of the inclusion of the respondent’s motives or the lack of social embeddedness or social formation of preferences, and therefore the use of more deliberative approaches to valuation have been advocated (Lienhoop et al., 2015).

While the motivation for increased knowledge on the economic impact of biodiversity losses on human welfare is clear (Polasky, 2009), assessments of the role of biodiversity for the generation of human welfare remain unclear (Barbier, 2012; TEEB, 2010).

Several key challenges predominate the scientific discourse: (i) The plurality and multiplicity of valuation languages (Cardoso, 2018) as well as the ambiguity on the definition of biodiversity and the object of valuation (Bartkowski, 2017; Bartkowski et al., 2015) weakens the credibility of the use of economic values of non-marketed goods for decision-making purposes, (ii) no established framework has been agreed upon that effectively assesses biodiversity losses for their effects on economic performances (Farnsworth et al., 2015; Nijkamp et al., 2008), (iii) ecological uncertainty (Tilman et al., 2014) and ambiguity (Jax and Heink, 2015) exist on the relationship between species diversity and ecosystem services and (iv) biodiversity is a multi-dimensional concept and requires multiple proxies for quantifying it (Bartkowski et al., 2015; Nijkamp et al., 2008).

In the literature, many sources of value derived from biodiversity have been identified. They can be found in the use value and existence value of individual species (Mace et al., 2012; Polasky et al., 2005), as a source of bio prospecting revenues or knowledge values (Heal, 2000; Polasky et al., 2005), as an integral part of the provision of ecosystem services via its contribution to ecosystem functions (Cardinale et al., 2012; Polasky et al., 2005), as a source of indirect use value (Costanza et al., 1997; de Groot et al., 2002; Farnsworth et al., 2015), insurance value (Baumgärtner, 2007; Heal, 2000; Henselek et al., 2016) or intrinsic value (Sandler, 2012). Sometimes, ecosystem resilience is considered an asset in itself for valuation (Walker et al., 2010). Recently, it has been argued that biodiversity has an economic value extending beyond these values, including an option value and a spill-over value (Bartkowski, 2017).

Hamilton (2013) recognized that, as a consequence of the fact that the majority of ecosystem services are provided to the economy as externalities, these values are already capitalized in other values such as farmland or as the economic benefits to their owners who benefit from the supply of (costless) environmental services. In this respect, adding up the values of the ecosystem services and including them as separate values in a national balance sheet would be considered double-counting (Hamilton, 2013).

Economic valuation of biodiversity as defined in natural sciences is yet unfulfilled and methodologies that provide a strong link between economic theory and ecological research (i.e. a production function analogy that describes how ecosystems generate services and expressing the relationships between the quantities of production factors used and the amount of goods or services produced) remain largely unexplored (Bartkowski et al., 2015; Daily et al., 2000; Farnsworth et al., 2015). Farnsworth et al. (2015) emphasize a refocusing of economists for the economic valuation of biodiversity towards production-based methods whereby biodiversity is considered as an input in a production function, thereby generating products or services that are used directly by humans. The production function estimates the contribution of biodiversity for the production of marketable goods or services (Bertram and Rehdanz, 2013). The use of a production function approach therefore recognizes functional group diversity as an essential production factor so that changes in functional group diversity indirectly affect the production of a marketable good. *"A production function approach generally uses scientific knowledge on cause-effect relationships between the ecosystem service(s) being valued and the output level of marketed commodities. It relates to objective measurements of biophysical parameters."* The ecological functions are considered "*emergent properties of a system and inherent to it, on different system levels, not artifacts or social constructs made and controlled by humans*. *They are relevant for the ecosystem, its functioning and development, regardless of any human recognition or valuation and in this sense objectively valuable for the ecosystem…" (Spangenberg et al., 2014). "Production function approaches estimate how much a given ecosystem service contributes to the delivery of another service or commodity which is traded on an existing market… The PF approach generally uses scientific knowledge on cause-effect relationships between the ecosystem service(s) being valued.*" (TEEB, 2010). In this respect, production functions have the advantage that they rely on objective measurements of biophysical parameters and can therefore quantify the physical effects of changes in a biological resource on an economic activity (Barbier, 1994, 2012; TEEB, 2010). Diversity might increase output by supporting landscape-level ecosystem functions that help to enhance productivity (Omer et al., 2007; Pascual et al., 2013; Tscharntke et al., 2005). The impact of these changes is valued in terms of the corresponding change in marketed output (Barbier, 1994; TEEB, 2010). For example, natural predators have been shown to perform important biological pest control services, thereby reducing crop damages and indirectly contributing to farmer's income (Daniels et al., 2017). Similarly, through consumer-resource interactions, the diversity of insects and other invertebrates in streams and rivers support the production of economically valuable fishes (Bellmore et al., 2017), which in turn, supports fishing industries and local economies.

The strength of production functions as a viable methodology for policy analysis (Barbier, 2007) stems from the potential to relate objective measurements of cause-effect relationships to changes in economic activities. In doing so, they provide justification when making a case for environmental protection by providing supporting scientific information on the effects of changes in biological resources for human welfare (Polasky, 2009).

Strengthening a production-based method could be achieved by stressing the functionality of biodiversity in valuation studies. The recent biodiversity valuation literature emphasizes that the ecological and broader biological role that biodiversity plays in human well-being should be at the centre of valuation studies (Bakhtiari et al., 2014; Bartkowski et al., 2015; Daniels et al., 2017; Farnsworth et al., 2015). Meta-analyses have shown that ecosystem eervices - the benefits humans receive from ecosystems – are tightly linked to the performance of ecosystem functions and the level of biodiversity (Wall and Nielsen, 2012). A loss of biodiversity may directly and indirectly affect ecosystem functions and services, as well as human welfare (Chapin Iii et al., 2000; Hooper et al., 2005). Functional groups of species provide a link between species diversity and ecosystem function (Cleland, 2011), and are the main units to investigate the consequences of global environmental change on ecosystem function and the delivered services (Carmona et al., 2016). It has been estimated that indirect use values may constitute the largest source of total economic value in biodiversity valuation (Costanza et al., 1997; de Groot et al., 2002; Farnsworth et al., 2015). The indirect use value can be derived from the regulation services provided by species and ecosystems (TEEB, 2010) or can be defined as the support and protection provided to economic activity by regulatory environmental services (Barbier, 1994). It is safe to assume that biodiversity has a large indirect use value to humans when it is considered as an input in a production function, thereby influencing the provision of products or services that are used directly by humans.

Bartkowski et al. (2015) presented an overview of needs for the proper valuation of biodiversity among which the need to formulate a coherent framework for the valuation of biodiversity based on the functional roles it plays. Studies on biodiversity require a pluridisciplinary approach (Nijkamp et al., 2008), requiring integrated valuation methodologies combining disciplines (ecology and economics) and methods (production function approach and market-based technique) and aiming at assessing real life impact (Jacobs et al., 2016). The Ecosystem Services Cascade introduced by Potschin and Haines-Young (2010) provides a cascade of consequent events leading to monetary valuation. The cascade starts from (i) Ecosystem Properties (EP) leading to (ii) Ecosystem Functions (EF), (iii) Ecosystem Services (ES), (iv) Benefits (B) and (v) Values (V) (see figure 1) (Boerema et al., 2017; Haines-Young and Potschin, 2010; Potschin and Haines-Young, 2011). Recently, it has been argued that i) a cascade is both "*an oversimplification of a complex reality*" as well as "*an unnecessary complication*" since "*ecosystem services equal benefits by definition*" and a better representation was suggested, including complex interactions and feedbacks among built, social, and natural capital in order to produce ecosystem services (Costanza et al., 2017).

Here, the use of production functions is explored whereby the flow of benefits provided by a functional diversity can be conceived as the result of a 'natural production function'. Functional diversity (quantified by multiple attributes) is the input to the production function, resulting in marginal changes in the flow of benefits (Barbier, 2012; Hamilton, 2013). In line with Costanza et al. (2017), we here expand the ecosystem service cascade by (i) adding a stepwise methodological framework to the cascade to assess the effects of changes in functional group diversity on economic activities; (ii) including multiple attributes for defining functional diversity and (iii) integrating a dynamic ecological model simulating complex interactions and feedbacks between species with an economic model assessing the effects of changes in functional group diversity for gross revenues. The stepwise methodological framework integrates a production function approach with a market price-based approach in order to investigate the indirect use value of functional group diversity based on the ecological role of species in the ecosystem. It serves to quantify the effects of changes in non–marketable species diversity for their impact on economic activities through the delivery of a selected set of ecosystem services and as such attaches an indirect use value to functional group diversity. The methodological framework both (a) quantifies the contribution of functional group diversity to gross revenues through the use of a production function, and (b) attributes an indirect use value to functional group diversity by employing a direct market based technique based on the changes in the provision of a marketable good.

2.3. A methodological framework for the valuation of functional diversity based on the ecological role of species in the ecosystem

In the following, a stepwise methodological framework is investigated as an extension of the ecosystem service cascade. An overview of the methodological framework is represented in figure 2.1.

STEP 1: Defining the Ecosystem Services Cascade narrative to determine scope:						
Select Functional group: the set of species that play equivalent roles in ecosystems and have similar effects on major ecosystem processes	➡	Ecosystem properties (EP): the biophysical structure of the ecosystem	Ecosystem Function (EF): The activities of organisms and their effects on the physical and chemical conditions of their environment	Ecosystem Services (ES): the contributions of ecosystems to human well-being	Benefits (B): positive changes in well-being from the fulfillment of needs and wants	Values (V): economic worth of the change in well-being

STEP 2: Dynamic Ecological Model Development

With the help of model software such as iThink or Vensim, a dynamic model allows for (i) continuous spatial and intertemporal variations, (ii) interactions between species, (iii) the effects of these interactions and variations on the ecosystem functions, services and values, (iv) a valuation of *all* species in the functional group.

STEP 3: Alternative Scenario Development

The number of alternative scenarios that can potentially be developed depends on the number of species i in the reference scenario R_r and equals 2^i-1 alternative scenarios.

STEP 4: Quantifying Ecosystem Function

The change in total function T between the baseline T and each of the alternative scenarios T' is given by $\Delta T=T'-T$ and results in a range of ΔT for all species richness levels, depending on the identity and abundance of the species removed.

STEP 5: Quantifying Ecosystem Services

Extrapolating the results of the Biodiversity – Ecosystem Function relationship to ecosystem services with $ES=f(T)$, requires adding seasonal variability and therefore depends on ecosystem function at a specific time of year.

STEP 6: Specification of an Ecological-economic Linking Function to determine Benefits

The benefits (B) derived from the change in ecosystem services $\Delta ES=ES-ES'$ are related to the actual use of the services. The ecological economic linking function therefore links the provisioning of ecosystem services to the benefits delivered to humans.

Step 7: Separating the Effects of Species Richness, Composition and Abundance

The differences in T and ES arise from the cumulative effect of changes in species richness, abundance, and composition. In order to separate the effects of richness from composition and abundance changes, the change in total services delivered is adjusted from the Price equation (Fox, 2006; Fox and Harpole, 2008; Fox and Kerr, 2012; Winfree et al., 2015).

Step 8: Assessing the Economic Value of Human Benefits

An economic value is attributed to the delivered benefits and assesses the costs/benefits of a change in abundance/richness and/or composition by analyzing the effects of the benefits at the income level (*i.e.* net farm income for changing yields).

Step 9: Determining the Indirect Use Value of the Functional group of Species

The changing levels of gross or net revenue correspond to a specific combination of species richness; composition or abundance levels and represent the indirect use values of the species under analysis.

Figure 2.1: Overview of the VABES framework to quantify the effects of changes in non–marketable species diversity for their impact on economic activities through the delivery of ecosystem services, attaching an indirect use value to species diversity.

Step 1: Definition of the Ecosystem Services Cascade narrative to determine scope

Describing the five key concepts of the Ecosystem Services Cascade sets the scope and boundaries for the analysis. Filling out the cascade starts by identifying the functional group to be valued. A functional group is defined as a set of species that have similar effects on major ecosystem processes (Blondel, 2003). Daniels et al. (2017) examined the effect of natural predators (i.e., functional group), which act as biological pest control of pear psylla (*Cacopsylla pyri L.*), on pear quality and net farm income. Other examples are the contribution of plankton to the (regional or global) commercial fishing industry or the contribution of wild pollinators to changes in net farm income from fruit or crop production. The analysis can contain one or more functional groups that will be valued, whereby each functional group can consist of an unlimited number of species performing a similar function. For example, to determine the consequences of a reduction of bacterial diversity on soil functions and bioremediation, functional groups of bacteria were identified (i.e. denitrifying or nitrifying bacteria, photosynthetic bacteria and organic carbon degraders) (Jung et al., 2016). The functional group does not have to be geographically located in the same area, but they do have to contribute to the production of the same marketable output.

Next, the endpoint is identified which directly or indirectly depends on the services delivered by the functional group of species. This endpoint is a marketable good or service, defined as a good or service that is sold and has a market value. Last, the ecosystem function (EF), ecosystem service (ES) and benefits (B) are the different cascade components linking the marketable good identified to the functional group of species to be valued and are explained in detail in step 3, 4 and 5 (see figure 1).

Step 2: Dynamic ecological model development

A dynamic ecological model starts from a multi-attribute approach taking into account the complexity, abstractness and multidimensionality of biodiversity. Since biodiversity is a multi-dimensional concept '*spanning genes and species, functional forms, adaptations, habitats and ecosystems, as well as the variability within and between them*' (Laurila-Pant, 2015), biodiversity proxies should not reduce biodiversity to one single aspect, should not cover more than biodiversity, and the connection between the proxy and the contribution of biodiversity to human well-being should be clear. Therefore, it is suggested to use a multi-attribute approach (Bartkowski et al., 2015), meaning that multiple variables are required to describe and quantify biodiversity. By choosing a multi-attribute approach to account for complexity, the choice of variables representing biodiversity should encompass at least the species richness, species composition and species abundance (or biomass) (Bartkowski et al., 2015). Added to the variables describing biodiversity are the population dynamic parameters, expressed on a continuous scale. A dynamic model allows for (i) continuous spatial and intertemporal variations, (ii) interactions between species, (iii) the effects of these interactions and variations on the ecosystem functions, services and values and therefore (iv) comparison of realistic alternative scenarios of species richness, composition and abundance (Letourneau et al., 2015), and (v) a valuation of *all* species in the functional group. System dynamic software packages such as Stella (iThink) or Vensim can provide valuable tools for building dynamic ecological models (Ford, 2009).

Step 3: Alternative scenario development

Once biodiversity is incorporated into a dynamic modeling framework, the model can be used to test alternative scenarios that evaluate the implication of species loss (or species replacement). The number of alternative scenarios that can potentially be developed depends on the number of species *i* in the reference scenario R_r and equals $2^i - 1$ alternative scenarios. The alternative scenarios all differ in species richness and/or species composition from the baseline scenario: some scenarios may have the same species richness (number of species) but may differ in the composition (identity) of the species present. For each alternative scenario, one or more species can be removed from the system in order to assess the individual and cumulative effects of removal. An example of the different scenarios for a functional group consisting of *i* species is represented in table 4.1.

Species (s=i)	R_1	R_2	R_3	**…**	R_2^{i}
SPECIES 1	x	0	x	…	0
SPECIES 2	x	x	0	…	0
SPECIES 3	x	x	x	…	0
…	…	…	…	…	…
SPECIES i	x	x	x	…	0

Table 4.1: Schematic overview of the reference scenario (R_r = R_1) and potential alternative different scenarios (R_2, R_3,…, R_{2^i-1}) to be developed, indicating the presence (x) or absence (0) of a species *i*. The reference scenario R_r = R_1 includes all species *i,* each of the alternative scenarios reduce species richness or share the same species richness but a different species composition and the last scenario R_{2^i} contains no species.

Step 4: Quantifying ecosystem function

The ecosystem function was defined as "*all the activities of plants, animals and bacteria and their effects on the physical and chemical conditions of their environment"* (Tilman et al., 2014).

The reference scenario contains s species (species richness) with species composition $i = 1,2 \dots, s$ and species abundance a_i. The alternative scenarios contain s' species with species composition *j*=1,2,…s' and species abundance a'_j. The number of species that both the reference scenario and the alternative scenarios have in common is denoted as s_c. The functional contribution of species i is denoted as z_i with $\sum_{i=1}^{s} z_i = s\bar{z} = T$ for the reference scenario and $\sum_{j=1}^{s'} z_j = s'\overline{z'} = T'$ where T and *T'* represent the total function (EF) for all species $s \; or \; s'$.

The change in total function T between the baseline *T* and each of the alternative scenarios *T'* is then given by $\Delta T = T' - T$ with the number of $\Delta T_{2^i-2} = 1,2,\dots,2^i-3,2^i-2$. This results in a range of ΔT for all species richness levels and depends on the identity of the species removed (since alternative scenarios can have the same species richness but can differ with regards to the identities of the species involved and therefore can also differ with regards to total function T).

Step 5: Quantifying ecosystem services

After a quantification of the total ecosystem function T, the ecosystem services delivered can be quantified. As an example, the increase in pollen grain deposition by pollinators (EF), for example, will be closely linked to an increase in pollination (ES) and hence yields (B). Therefore, extrapolating the results of the Biodiversity – Ecosystem Function relationships (step 4) to ES is an essential step in the valuation process that requires establishing a quantitative relationship between EF and ES. The relationship $ES = f(T)$ is highly specific and depends on the nature of the Ecosystem Function *T* and the ES. The nature of the EF-ES relationship can be determined by seasonal variability and hence depends on ecosystem function at a specific time of year. For example, for pollination different types of relationships have been found between June pollen concentrations and the yields of dryland cereals on the one hand and between mean cereal yields and mean annual pollination on the other (Muñoz et al., 2000). A time-specific effect is also encountered for biological pest control services when pest populations at a crucial stage of the growth process are more significant than at other periods of the year (Daniels et al., 2017). Carbon sequestration has also been shown to exhibit seasonal variation patterns (Zhao et al., 2016). For the dynamic approach explained here, the existence of a specific time-dependent relationship does not pose any issues for establishing the relationship, due to the continuous nature of the model outputs.

Step 6: Specification of an ecological-economic linking function to determine benefits

The economic-ecological linking function links the ecosystem services quantified for each scenario to the benefits delivered to humans. It links the ecological model (from EPs to ES) to the economic model (from B to V). The benefits (B) derived from the change in ecosystem services $\Delta ES = ES - ES'$ are related to the actual use of the services *(i.e.* pollination is linked to yields (kgha^{-1}yr^{-1}; numbersha^{-1}yr^{-1}), gross energy (GJha^{-1}) or food consumed (ton household^{-1}yr^{-1})(Boerema et al., 2017). If the economic-ecological linking function is not known, a number of functions can be simulated, resulting in a range of benefits (Daniels et al., 2017) (*i.e.* linear, logistic, logarithmic, exponential,…).

Step 7: Separating the effects of species richness, composition and abundance

One of the key questions for maintaining continued provision of ecosystem services is whether the EF and ES depend on the number of species present, on key species, species traits or on composition of the communities (Wall and Nielsen, 2012). The difference in T and ES as a consequence of the changes in biodiversity arises from the cumulative effect of changes in species richness, abundance, and composition. Therefore, the change in total services delivered (ES) is not only due to losses in species richness, but also depends on the number of individuals lost and the composition of the species that remain. In order to separate the effects of richness from composition and abundance changes, the change in total services delivered stems from the Price equation (Fox, 2006; Fox and Harpole, 2008; Fox and Kerr, 2012; Winfree et al., 2015) according to:

$$\Delta ES = ES - ES' = (s_c - s)\bar{x} + (s' - s_c)\overline{x'} + \mathrm{Sp}(w_{\cdot}^{I}, x) + \left[\text{-}\mathrm{Sp}\left(w_{j}^{\cdot}, x'\right)\right] + \sum_{i=1}^{s} \sum_{j=1}^{s'} w_j^i (x'_i - x_i) \qquad \text{(eq.2)}$$

where Sp denotes the sum of products operator, $w_{\cdot}^{I} = \sum_{j=1}^{s'} w_j^i$, and $w_j^{\cdot} = \sum_{i=1}^{s} w_j^i$. The variable $w_{\cdot}^{I} = 1$ if species *i* is present at both sites and 0 otherwise, while the variable $w_j^{\cdot} = 1$ if species j is present at both sites and 0 otherwise. The variable w_j^i indicates whether a species is present at both sites, so that $w_j^i = 1$ if $i = j \leq s_c$ and 0 otherwise so that:

$$\Delta ES = ES - ES' = (\text{RICH-L}) + (\text{RICH-G}) + (\text{COMP-L}) + (\text{COMP-G}) + (\text{ABUN}) \qquad \text{(eq.3)}$$

The first term of equation 2 is a measure of the fraction of change in total ES due to species losses from the baseline site (RICH-L) and is analogue to the second term, which represents the fraction of change in ES due to increased species richness (RICH-G). The third term (COMP-L) captures the effects that depend on the identity of the species lost. If species with a low functional contribution are absent from alternative scenario, this will increase average functional contribution. Similarly, the fourth term (COMP-G) captures the effects that depend on the identity of the species gained. The last term (ABUN) captures differences in abundance effects for the species common to both sites. Depending on the ultimate goal of the analysis, the effect of changes in species richness, composition, abundance or functionality can be singled out as a proportion of ΔES delivered and the corresponding values calculated for each component of biodiversity. (*i.e.* if the goal is to analyse the effect of the loss of species: the ΔES in function of RICH-L (ΔES_{RICH-L}) can be singled out and the benefits and values calculated based on them. For mathematical details or proof of the additionality of the five terms, we refer to appendix S1 in Winfree et al. (2015).

Step 8: Assessing the economic value of human benefits

Through employment of the Ecosystem Services Cascade (see figure 1), the relationship between a functional group of species and a marketable good can be established. In order to have a realistic representation of the contribution of diversity to changes in welfare, an economic value is attributed to the benefits delivered whereby not just the changes in gross revenues but also the changes in net revenues are to be analyzed. The economic model assesses the costs/benefits of a change in abundance/richness and/or composition by analyzing the effects of the benefits at the income level (*i.e*. net farm income for changing yields).

The net income for each scenario is defined as $I_N = I_G - TC$ with I_G the gross revenue (price x quantity) and TC the total costs (equaling the sum of all variable and fixed costs). A sensitivity analysis takes into account uncertainty in the data. This results in confidence intervals for the net income for each scenario, which is a function of changes in species richness, composition and/or abundance changes.

Step 9: Determining the indirect use value of the functional group of species

Throughout the framework and for each step of the Ecosystem Services Cascade, the contribution of more or less biodiversity for the delivery of economic value can be traced back to changing levels of species richness, composition, and abundance. The changing levels of gross or net revenue therefore correspond to a specific combination of richness, composition or abundance levels and make up the indirect use values of the species under analysis.

2.4 Discussion

In line with recent recommendations by Costanza et al. (2017), the ecosystem services cascade was expanded to include complex interactions and feedbacks found in ecosystems to represent the complexity of consumer-resource interactions. Here, we (i) added a stepwise methodological framework to the cascade to assess the effects of changes in functional group diversity on economic activities; (ii) included multiple attributes for defining functional diversity and (iii) integrated a dynamic ecological model simulating complex interactions and feedbacks between species with an economic model assessing the effects of changes in functional group diversity for gross revenues. The stepwise methodological framework integrates a production function approach with a market price-based approach in order to investigate the indirect use value of functional group diversity based on the ecological role of species in the ecosystem.

As opposed to the dynamic approach suggested here, an empirical approach, based on a fixed number of field experiments and following the same steps (except for step 2: building a dynamic ecological model and step 3: scenario development) could also result in an indirect use value of biodiversity. With an empirical approach the ecological function is measured or observed in terms of the diversity present, instead of modeled. The selection of an empirical or a dynamic approach has important consequences for the number of species to which an indirect use value can be attributed. The consequences of both approaches differ considerably with regards to the valuation of biodiversity, since an empirical approach will only be able to deliver a valuation for a fixed set of species in the functional group, being that set of species that is absent in the alternative scenario(s). This approach is also limited in expressing spatial and temporal variations since data availability is expressed at a limited number of points in time/spatial locations. Both approaches can however account for complexity and abstractness of biodiversity by choosing a multi-attribute approach in the choice of variables representing biodiversity: species richness, species composition and species abundance (Bartkowski et al., 2015).

Finally, the methodological framework could be extended to include the effect of management practices. Steps 1 to 9 of the framework did not specify any cause(s) for changing biodiversity levels. However, the framework can be extended to include the valuation of management practices by examination of their effects on biodiversity and hence of their effects on the marketed goods or ecosystem services.

With the proposed methodological framework, we hope to facilitate and encourage further research on the effect of changes in biodiversity for the economy and human well-being that effectively take into account the importance of species diversity for ecological function, with the ultimate aim of assessing the effects of ecosystem management for the well functioning of ecosystems.

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CHAPTER 3

Monetary Valuation of Natural Predators for Biological Pest Control in Pear Production

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Chapter 3: Monetary Valuation of Natural Predators for Biological Pest Control in Pear Production

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3.1 Abstract

In spite of global actions, biodiversity is declining at an alarming rate. Despite the need for objectively comparable monetary standards to include biodiversity arguments in policymaking, research on the relationship between species diversity and its valuation from a societal perspective is still scarce.

In this paper, a methodological framework for the valuation of natural predators based on their ecological role in the agroecosystem is introduced. The framework integrates a dynamic ecological model simulating interactions between species with an economic model, thereby quantifying the effect of reduced numbers of natural predators on the net farm income. The model attributes an objective monetary value to increased species diversity through the changes in the provisioning of a marketable good.

Results indicate that the loss of three predators could decrease net farm income with 88.86 €ha^{-1} to 2186.5 €ha^{-1}. For the pear production sector in Flanders in 2011, this constitutes to an indirect use value of 0,68 million € for one predator and 16.63 million € for the presence of three predators. The aim is to provide a justification for the argument for biodiversity conservation, based on the ecological function of species, through the delivery of comparable monetary standards.

Keywords: monetary valuation, ecological function, biodiversity loss, biological pest control, ecological-economic modeling

3.2 Introduction

In spite of global actions, biodiversity is declining at an alarming rate (Butchart et al., 2010). The transformation of natural landscapes to agricultural systems, the abandonment of farmland with high natural values, and the intensification and changing scale of agricultural operations are the key processes driving low ecosystem quality and biodiversity losses in agro-ecosystems (Liu et al., 2013; Reidsma et al., 2006; Smith et al., 2013). Available evidence strongly indicates the importance of agro-ecosystem restoration for environmental benefits and acknowledges the potential to simultaneously minimize biodiversity harm at the local level and increase farm yields (Barral et al., 2015; Cunningham et al., 2013).

Although measurements of biodiversity have often been investigated, analyses at the farm scale and specific studies providing insights into factors driving agro-ecosystem community structure are scarce (Birrer et al., 2014; Farnsworth et al., 2015; Turtureanu et al., 2014). Furthermore, habitat and increased numbers of natural predators facilitate the provisioning of important ecosystem services such as maintaining agricultural pest control, and may increase efficiency in controlling pests. However, the relationship between natural predators and pest reduction potential is not well established (Chaplin-Kramer et al., 2013; Letourneau et al., 2015). More specifically, the control of pests and diseases by biological control agents contributes positively to the provisioning of agricultural products of a better quality or in higher quantities, however the relationship between the presence of natural predators and pear production in particular has not been investigated yet. Mathematical models for biological pest control have proposed the use of linear feedback control strategies to indicate how natural enemies should be introduced into the environment (Rafikov and de Holanda Limeira, 2011).

Farmers are in need of supporting evidence of biodiversity benefits outweighing the opportunity costs incurred in order to strengthen the argument for biodiversity conservation at the farm level.

Moreover, without economic valuation of the environment, policy decisions that contradict economic rationality could be supported. In spite of the need for objectively comparable monetary standards, empirical literature investigating the relationship between species diversity and its valuation from a farmer's perspective is still scarce (Finger and Buchmann, 2015). The elicitation of values for biodiversity with the aid of stated preference methods suffers from the generally low level of awareness and understanding of what biodiversity means on the part of the general public (Bräuer, 2003; Christie et al., 2006). Furthermore, the willingness-to-pay (WTP) for species that are unfamiliar or undesired by the general public could yield extremely low values despite the fact that these species could be performing indispensible ecological services and thereby contribute indirectly to the farmers' income. This, combined with the complexity of biodiversity (Feest et al., 2010), might just overstretch the capacity of the usual stated preference valuation techniques for the valuation of biodiversity (Bartkowski et al., 2015). Revealed preference techniques have the advantage that they rely on the observation of peoples' actions in markets. However, the majority of species do not have a market price. Letourneau et al. (2015) value the changes in natural enemy diversity by studying changes in producer and consumer surplus. They estimate that losses in natural enemy species richness in squash and cucumber fields in Georgia and South Carolina could cost society between $1.5 and $12 million in social surplus every year.

In this paper we provide a complementary approach and overcome some of the limitations mentioned by Letourneau et al. (2015) by (i) including an ecological model that allows for spatial and temporal variation in the ecosystem service potential of natural enemies, their interactions with pests and the effect of those interactions on pest control cost savings, (ii) providing an alternative approach when the relationship between natural enemies and crop damage is not known, as is true for the majority of cases, (iii) confirming the results of Letourneau et al. (2015) that values are case specific and providing these values for a different crop in a different climatic zone, with a different pest insect and natural enemies and (iv) including the comparison of realistic alternative scenarios of species richness and measure economically meaningful data in a field setting that comes close to the conditions that prevail on actual farms.

This paper values the biological pest control provided by three natural predators of pear psylla (*Cacopsylla pyri L.*) (Homoptera: Psyllidae) in organic pear orchards in Flanders (Belgium). Three main research hypotheses are investigated:

H_1: a decrease in natural predators' species richness causes a decrease in pest suppression

H_2: a reduction in species richness of natural predators reduces marketable agricultural production, thereby decreasing farm revenues

H_3: an alternative valuation method for natural predators based on their ecological function in the ecosystem can be identified

The first hypothesis is quantified through the development of an ecological simulation model; the second hypothesis is supported by the use of production functions and a direct market valuation technique and the third hypothesis integrates all three research tools: an ecological simulation model with a production function approach and a direct market valuation technique.

The approach results in a monetary value for marginal changes of biodiversity losses (here: reduced number of natural predators) whereby the functional role of the species in the ecosystem (here: pest control) is the key mechanism for affecting the provisioning of a marketable good (here: agricultural production). The aim is to provide support for the decision making process so that not only the costs of biodiversity conservation can be taken into account but also the monetary benefits.

3.3 Case study description: biological pest control of pear psylla

Apple and pear production in Flanders accounted for 13764 hectares in 2011 and increased to 14285 ha in 2013, comprising 3% of all farmland. Since 2005, pear production comprised just over half the hectarage with 7607 ha in 2011 and 7995 ha in 2013. The province of Limburg accounts for 85% of the total apple and pear production in Flanders. In 2011, an average farm possessed 12,0 hectares of pear plantations and 14,4 hectares in 2013. Organic production accounts for only a small fraction but production areas increased by 224% over the period 2002 – 2012 from 25,09 ha to 58,07 ha. Average yields were 36031 kg per ha in 2011 and 38681 kg per ha in 2013, with a maximum of 44751 kg per ha in 2014 (Van der Straeten, 2016). Yearly sales volumes of pears amounted to almost 340 million kg in 2014 (NIS, 2015). Annual sales revenues ranged between 15133 €ha^{-1} in 2011 and 20114 €ha^{-1} in 2013 (Van der Straeten, 2016). Yearly average selling prices for the period 2009-2013 were 0.57 €kg^{-1} for first-class pears, 0.39 €kg^{-1} for second-class pears and 0.88 €kg^{-1} for organic pears (personal communication Regional Auction Borgloon). Assuming that annual sales volumes would consist of second class pears only, 55.68% of gross revenues would be lost since if harvests consisted of only second class pears and gross revenues would amount to 11736 €ha^{-1} as compared to 26481 €ha^{-1} for harvests consisting of only first class pears (Van der Straeten, 2016). The sector is characterized by a decrease in the number of farms and an increase in the average size. Sales volumes and revenues remain extremely volatile due to changing environmental and market conditions (Platteau et al., 2014).

A major threat for the pear production industry is pear psylla (*Cacopsylla pyri*). The adults cause damage both directly by extracting nutrients from the meristem tissue, and indirectly by causing russet and roughness on pear skin. Pear psylla's status as a major pest is based on its damage potential and its ability to develop resistance to insecticides. Through the production of honeydew, the growth of black, sooty fungi, causing so-called "black pears" is facilitated. It russets the pear skin and causes the fruit to be downgraded, thereby decreasing its market value (Erler, 2004). Literature quantifying the relationship between pest insect density levels and the occurrence of fruit russet is however scarce (Brouwer, 2008). Research revealed the failure of conventional chemical control agents against the pear tree psyllid, stressing the need for alternative strategies such as enhancing natural arthropod enemies (Daugherty et al., 2007; Erler, 2004; Rieux et al., 1999). Pear psylla are commonly attacked by several different natural enemies (e.g. *Anthocoris nemoralis* (Heteroptera: Anthocoridae), *Allothrombidium fuliginosum* (Acari: Trombidiidae) and *Heterotoma planicornis* (Hemiptera: Miridae)), of which *A. nemoralis* is the most common predator. Data collection is comprised of two independently executed field tests. The first field test comprises field data collected on 7 plots in organic *Conférence* pear orchards in Hesbaye (Belgium) for two years from 2013 until 2014. Each field test sampled pear psylla eggs and nymphs on multiple days with an interval of 2-3 weeks (See ANNEX A.1 for data sampling method and pooled results). The second dataset was obtained from field tests performed every two weeks for the period 2010-2011 on 7 different organic plots in Hageland (Belgium) and Gelderland and Limburg (NL). The same techniques were used to assess mean egg numbers and larvae numbers (visual scouting and the beating tray method) (see ANNEX A.3).

Counts for the presence of beneficial insects were performed between February and October of 2013 and 2014 in organic *conférence* pear orchards (see ANNEX A.2 for data sampling methods and pooled counts).

3.4 Methodology

3.4.1 Ecological model construction

The ecological model simulates predator-prey dynamics between the pest insect and three of its main natural enemies to analyze the effect on pear psylla (Pp) abundance in case of a reduction in species diversity and abundance of natural predators. The main criterion for selection of the natural enemies is the importance of a species as main pear psylla antagonist and has been verified through expert opinion and literature review. With the use of STELLA 10.0.6 (Stella; available at http://www.iseesystems.com) (Costanza and Gottlieb, 1998; Costanza and Voinov, 2001), the biodemographics of a pest insect *Cacopsylla pyri* (Pp) and the interaction with (i) *Anthocoris nemoralis* (An), (ii) *Allothrombidium fuliginosum* (Af) and (iii) *Heterotoma planicornis* (Hp) (Erler, 2004) are simulated over a period of one year whereby:

$$dn_{Pp}/dt = f\left(n_{An}, n_{Af}, n_{Hp}, n_{other}\right) \qquad \text{(eq. 1)}$$

with n the species abundance and n_{other} the effects of other predators not explicitly included in the model.

Initial model parameter values are allowed to vary on a daily basis and can be found in ANNEX B. The food fractions (the fraction that Pp makes up in a daily diet of a natural predator) were set at 0.8 for specialists (An) and 0.2 for generalists (Af and Hp) (Piechnik et al., 2008). The number of Ppe (eggs) and Ppn (nymphs) preyed upon per day are variable and depend on prey density according to a logistic dependency. The higher the density of Pp, the more Pp will be subject to predation as opposed to a linear dependency approach. Natural mortalities for all species are represented as a time-dependent variable longevity. Both Oviposition and longevity are non-constant parameters, depending on the time of the year and the adult generation cycle. The carrying capacity for Pp has been determined by excluding predation under the assumption that resource use did not pose constraints. The growth function is modeled as a logistic growth curve, followed by a decline of the population.

In the model, the effects of omitted species in the agro-ecosystem have been taken into account in various ways:

(i) An, Af and Hp are themselves subjected to predation from omitted species at higher trophic levels and this effect has been taken into account by the inclusion of a predation fraction for An, Af and Hp of 0.6. All natural predators are continuously exposed to this predation fraction, on top of the longevity variable. The natural predators, as well as the pest insect, therefore disappear from the model either by natural death or due to predation by omitted species.

(ii) An, Af and Hp have multiple food sources besides Pp which is represented in the model by varying the An, Af and Hp food fractions between 0 and 1. The predation fractions therefore allow the predation of omitted species.

Other predators besides the three natural predators included in the model prey on Cacopsylla pyri. This effect is not included in the model, since the main aim of the model is to assess the specific effect of the loss of three specific natural predators on pest insect dynamics.

Despite the potential for beneficial effects for other natural predators upon removal of one natural predator, no such interspecies competition has been taken into account due to various reasons:

(i) different pest stages are attacked by different predators. Each species is modelled throughout their different life stages (egg, nymph, adult) and it is only that specific stage which is under predation from that natural predator.

(ii) there is an overlap in timing of occurrence for the three natural predators but their peak times differ considerably, thereby reducing the potential for competitive effects.

(iii) they differ in their nature (generalists/specialists) and generalists have the ability to switch to other food sources.

(iv) the pest insect is abundant and there is no lack of food resources for all predators.

Biodiversity loss is then quantified by the loss in species richness of natural predators which is defined as the loss in the total number of species present, and assessed for its effect on the species abundance of the pest insect, both expressed in absolute numbers per hectare. A total of eight model scenarios (S1 – S8) were developed with S1 containing all species, S2 - S4 extinction of one natural predator, S5 - S7 extinction of two predators and S8 no natural predators.

Predator species	**Scenarios**							
	S1	**S2**	**S3**	**S4**	**S5**	**S6**	**S7**	**S8**
PREDATOR 1: *Anthocoris nemoralis* (An)	x	x	0	x	0	x	0	0
PREDATOR 2: *Allothrombidium fuliginosum* (Af)	x	x	x	0	x	0	0	0
PREDATOR 3: *Heterotoma planicornis* (Hp)	x	0	x	x	0	0	x	0

Table 2.1: Schematic overview of the eight predator loss scenarios developed, indicating the presence (x) or absence (0) of a natural predator for 8 scenarios (S1-S8). Scenario 1 (S1) contains the pest insect and three natural predators, scenario 2 to 4 (S2 - S4) contains the pest insect and two predators, scenario 5 to 7 (S5 - S7) contains the pest insect and one natural predator and scenario S8 represents the scenario without predators.

The effect of a loss of species richness of natural predators is modeled for a one-year period whereby the effect on pest suppression results in the absolute biological pest control loss BPC_{loss} composed as the sum of (i) an increase in pest insect abundance (Pp_I) and (ii) a decrease in predation (C_{loss}) with

$$BPC_{loss} = \sum(C_{loss}, Pp_I) > 0 \quad \text{(eq.2)}$$

with $Pp_I = \sum(Ppe(S1) + Ppn(S1)) - \sum(Ppe(Sx) + Ppn(Sx)) < 0$ (eq.3)

and $C_{loss} = C(S1) - C(Sx) > 0$ (eq.4)

Since eggs and nymphs are the main target for predation by predators, Pp_I calculates the difference between S1 and each of the other scenarios (Sx) for the sum of all eggs Ppe and nymphs Ppn appearing per year.

The relative loss in biological pest control $RBPC_{loss}$ for S2-S8 compared to S1 is then

$$\frac{BPC_{loss(Sx)}}{BPC_{loss\,(S1)}} \qquad \text{(eq.5)}$$

As eggs and nymphs are the main target for predation by predators, $RBPC_{loss}$ is described in terms of numbers for pest insect eggs and nymphs. These losses result in exponential increases of numbers of adults over multiple generations per year. The latter numbers are then linked to the occurrence of black pears through the identification of an ecological-economic linking function.

3.4.2 Identification of ecological-economic linking function

Linking biological pest control losses, which result from the ecological simulation model, with the economic model (section 3.3) is established by identifying a damage threshold function that links the maximum pest density level ∂_{Ppa} (adults $ha^{-1}y^{-1}$) over all eight scenarios with the yield quality decrease (black pear occurrence) γ (%). It is assumed that the maximum ∂_{Ppa} at any given time throughout the growing season will affect fruit russeting. Experimental fruit research institutions recommend action to avoid 'detectable damage' when monitoring reveals pest insect densities ∂_{Ppa} > 1000 adults per 10 beatings ($\partial_{ETL} = 386*10^6$ adults ha^{-1})[9]. They then define the Economic Treshold Level (ETL) as the percentage of black pears that is encountered at ∂_{ETL}.

Since the shape of the damage threshold function is not known, two sets of four hypothesized relationships are constructed to simulate the correlation between Pp_a density levels δ_{Ppa} ($ha^{-1}y^{-1}$) and black pear occurrence γ (%) for the two assumptions made:

(i) Linear: $\gamma_{lin} = \alpha\, \partial_{Ppa}$ (eq. 6)

(ii) Logistic: $\gamma_S = \frac{k}{(1+(k-\partial_0/\partial_0)} * exp^{r\partial_{Ppa}}$ (eq. 7)

(iii) Logarithm: $\gamma_{log} = 1 - exp^{-\partial_{Ppa}}$ (eq. 8)

[9] ∂_{Ppa}>1000 (adults per 3 shoots)*20 (assume 5% caught)*40 (shoots per tree)* 1450 (trees per ha) = $386*10^6$ (adults per ha)

(iv) Exponential: $\gamma_{exp} = exp^{\partial_{Ppa}}$ (eq. 9)

For the two sets of relationships, this results in a lower (γ_l) and upper (γ_u) percentage of black pears for each scenario S1-S8 with:

$\gamma_l = \min(\gamma_{lin}, \gamma_S, \gamma_{log}, \gamma_{exp})$ and $\gamma_u = \max(\gamma_{lin}, \gamma_S, \gamma_{log}, \gamma_{exp})$ (eq. 10)

The first set of four hypothesized relationships assumes that the maximum ∂_{Ppa} in the no-predator scenario (S8) results in 100% black pears. This results in an ETL of 0,28% and 32,02% black pears (figure 3.1 left vertical axis).

The second set of four hypothesized relationships assumes that the ETL for ∂_{Ppa} equal to $386*10^6$ adults ha^{-1} equals 1% of black pears. This results in a potential maximum amount of black pears of 12.90% at maximum ∂_{Ppa} [10] (figure 3.1 right vertical axis).

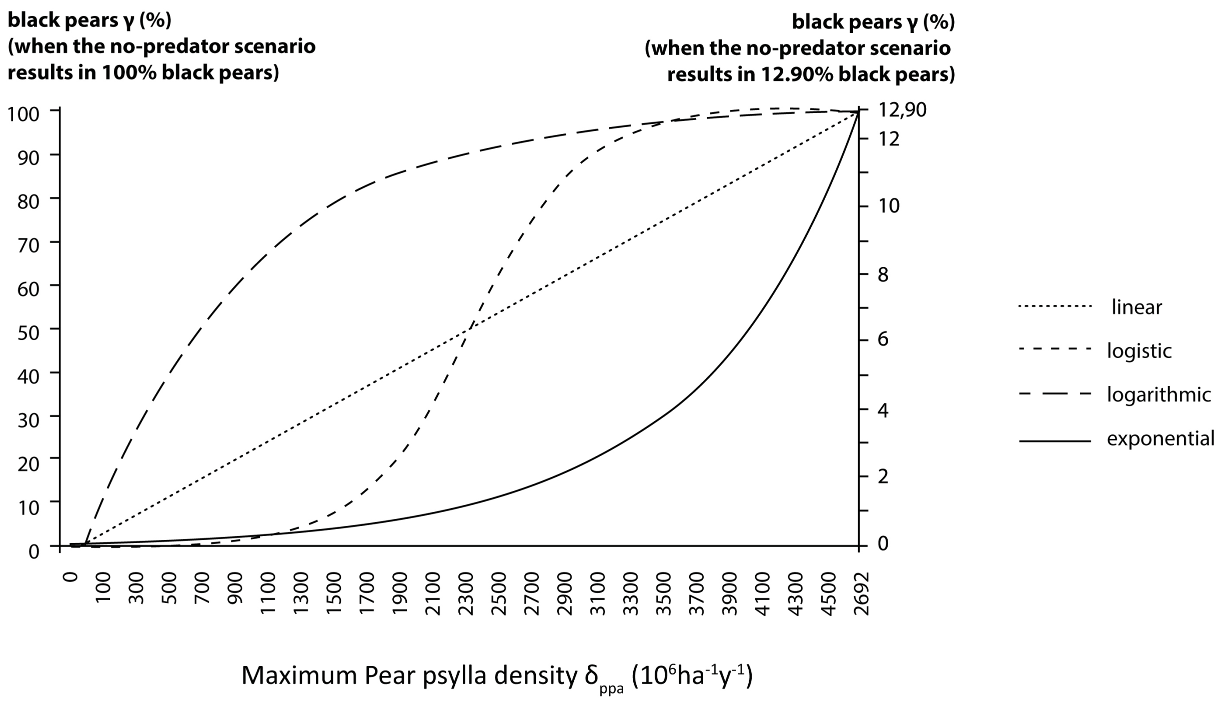

Figure 3.1: shows the four hypothesized relationships γ_{lin}, γ_S, γ_{log}, γ_{exp} that can exist between the maximum pest density level δ_{ppa} ($10^6 ha^{-1} y^{-1}$) and the occurrence of black pears γ (%). For each scenario, changing natural predator species results in changing pest density levels. The damage

[10] It is assumed that 'detectable damage' for the farmer equals 1% black pears.

threshold function then assesses the lower (γ_l) and upper (γ_u) percentage of black pears encountered at the maximum pest density level δ_{ppa} ($10^6 ha^{-1} y^{-1}$). For the first set of hypothesized relationships (left vertical axis), the maximum ∂_{Ppa} in the no-predator scenario (S8) results in 100% black pears (and therefore the ETL ranges between 0,28% and 32,02% black pears). The second set of hypothesized relationships (right vertical axis) assumes that the ETL equals 1% of black pears, resulting in a maximum potential percentage of black pears of 12.90%.

3.4.3 Economic model construction

The economic model assesses the costs of a decrease in abundance and richness of natural predators by analyzing the effects on yield quality decreases at farm scale calculating the impact on (i) gross revenue and (ii) net income.

The gross revenue I_G for each scenario is defined as $I_G = \sum(I_b, I_f)$ with b black pears and f first class pears where I_b (respectively I_f) represents the gross revenue with $I_b = P_b * Q_b$ (respectively $I_f = P_f * Q_f$), with P_b (respectively P_f) the price and Q_b (respectively Q_f) the quantity. The farm net income for each scenario is defined as $I_F = I_G - TC$ with TC the total costs, C_v the sum of all variable costs and C_f the sum of all fixed costs.

Annual accounting data on yields (kg ha^{-1}), revenues (€ ha^{-1}), variable costs (€ ha-1) and fixed costs (€) for organic production and non-organic production (ANNEX C) were used from the Agricultural Monitoring Network (LMN) data (Van der Straeten, 2016), which are conform FADN[11] data collection procedures. The LMN dataset contains 53 non-organic pear farmers (accounting for 662 hectares) and provides annual accounting data for the period 2009-2014 (Van der Straeten, 2016). Some numbers needed adjustment to represent organic production taking into account the following assumptions: (1) yields ($kgha^{-1}$) are 80% of non-organic production with μ = 30092,27 $kgha^{-1}$ and s = 3652,28[12], (2) organic management requires 30 % more full-time equivalents (FTEs) with μ = 4118,33 €ha^{-1} and s = 352,15 for non-organic production and μ = 5353,83 €ha^{-1} and s = 457,79 for organic production (EC, 2013).

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[11] Farm Accounting Data Network

[12] With μ the average and s the standard deviation

The parameters for which differences exist between organic and non-organic production are discussed here, for all other parameters we refer to ANNEX C. The yearly average selling price for 2009-2013 for all pear classes was μ = 0.57 €kg^{-1} (s = 0,16) (Van der Straeten, 2016) (with μ = 0.55 €kg^{-1} and s = 0,16 for first class non-organic pears, μ = 0.88 €kg^{-1} (s = 0,17) for organic pears and μ = 0.39 €kg^{-1} (s = 0,12) for black pears (personal communication Regional Auction Borgloon))."

The Department of Agriculture and Fisheries[13] states that organic farmers receive 50% higher subsidies (μ = 140 €ha^{-1} (s = 55) for non-organic and μ = 210 €ha^{-1} (s = 55) for organic production). Costs for crop protection account for 1579,83 €ha^{-1} (s = 100,12) for non-organic production and no costs are taken into account for organic production (Van der Straeten, 2016).

Yields of black pears for each scenario were calculated based on the percentages of black pears encountered in the two sets of hypothesized relationships (section 3.2) and hence differ for all scenarios under analysis. For reasons of simplicity, other production factors (*e.g.* conservation costs, maintenance, packaging) are assumed equal for non-organic and organic production. The accounting data are imported into the risk analysis tool Aramis (@risk) and all economic parameters are stochastic variables to calculate a confidence interval for the gross revenues and the farm net income for each scenario S1-S8. Results from the risk analysis show the difference in gross revenues and the farm net income for a 95% confidence intervals for S1 to S7 for the two sets of relationships and are linked to yield quality decreases (black pear increases) that result directly from species richness losses.

3.4.4 Model calibration

We calibrated the dynamic simulation model for pest suppression in organic agriculture based on field data from one year for which most data points were available (2010). The units of field measurements (mean eggs/10 shoots) were transformed to yield model parameter units (absolute egg numbers per hectare), based on 33,84 shoots/tree on average, 5% of the eggs captured and

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13 http://lv.vlaanderen.be/nl/bio/subsidies/hectaresteun-biologische-productiemethode-pdpo-iii (last visited: 08-08-2016)

1714 trees per hectare (Van der Straeten, 2016). The reference model (S1) predicts both the peak density as well as the timing of the peaks relatively well (see ANNEX D).

3.5 Results

3.5.1 Losses of natural predators result in significant decreases for biological pest control $RBPC_{loss}$

The effect of a loss of species richness of natural predators on pest insect suppression revealed an increase in pest insect abundance (Pp_I) (see eq.3) with decreasing predator numbers depending on the generalist/specialist nature of predation. For the reference scenario (S1), containing the 3 natural predators under investigation, the peak density of the sum of pest insect eggs and nymphs equaled 1237*10^6ha^{-1}. S7 simulated the absence of An and Af revealing an increase to maximum peak density of 23888 (10^6ha^{-1}) or an increase rate of 19.31. S2 (respectively S3; S4; S5; S6) simulates the absence of Hp (respectively $An; Af; An\,\&\,Hp;\ Af\,\&\,Hp; An\,\&\,Af$) resulting in a peak density increase rate of 6.57 (respectively 10.21; 8.82; 12.94; 19.31) revealing increases in eggs and nymphs absolute numbers to 2551 (respectively 12633; 8130; 10905; 16005) (10^6ha^{-1}).

Furthermore, for S1, 133 (10^6ha^{-1}) of the total eggs and nymphs (see section 4.1) are consumed in absolute terms (eq. 4). For S2 (respectively S4; S5; S6; S7) predation decreased to 113 (respectively 88; 78; 27; 4) (10^6ha^{-1}) equal to a reduction of 14.45 % (respectively 33.71%; 96.98%; 79.61%; 41.43%) compared to predation in S1. For S3 an increase in predation to 290 (10^6ha^{-1}) was observed. This can be explained by the sharp increase in absolute numbers but when comparing relative numbers predation decreased from 10.72% in S1 to 2.30% for S3.

Summing the (i) increase in pest insects density and (ii) the decrease in predation resulted in an estimate for the biological pest control provided by differing combinations of natural predators (eq. 2). For S1, 10.72% of the total eggs and nymphs are consumed. For S2 to S7 the relative biological pest control $RBPC_{loss}$ reduced gradually to 4.45%, 2.30%, 1.08%, 0.71%, 0.17% and 0.02%.

Predator losses resulted in exponential increases of numbers of pest insect adults over multiple generations per year, and the maximum peak densities for pest insect adults δ_{ppa} (10^6ha^{-1}y^{-1})

increased from 146.92 for S1 to 379.77 (respectively 386.00; 1331.68; 1815.20; 2134.83; 2714.97; 4036.55) for S2 (respectively S3; S4; S5; S6; S7). The no predator scenario (S8) resulted in adult pear psylla densities of 4692.23 $10^6 ha^{-1} y^{-1}$. Biological pest control losses of eggs and nymphs therefore induced adult pest insect increases as compared to S1 of 258% for S2, 263% for S3, 1236% for S4, 1453% for S5, 1847% for S6, 2747% for S7 and 3193% for S8, thereby strongly supporting Hypothesis 1.

Next, the decrease in biological pest control, particularly the increase in adult pest insect densities, was investigated for its potential to decrease pear quality in terms of % black pears observed.

3.5.2 Correlation between maximum pest insect density δ_{ppa} and black pear occurrence γ

For each scenario, the maximum pest density δ_{ppa} ($10^6 ha^{-1} y^{-1}$) resulting in a lower (γ_l) and upper (γ_u) percentage of black pears for the two sets of four hypothesized relationships γ_{lin}, γ_S, γ_{log}, γ_{exp} was obtained. The results are presented in table 2.2.

(1)	(2)	(3)	(4)	(5)	(6)
Scenario	Max pest insect density δ_{ppa} ($10^6 ha^{-1} y^{-1}$)	Loss of three predators causes 100% black pears		Loss of three predators causes 12.90% black pears	
		Lower % black pears (γ_l)	Upper % black pears (γ_u)	Lower % black pears (γ_l)	Upper % black pears (γ_u)
S1	146.92	0.14	13.66	0.01	1.08
S2	379.77	0.27	31.60	0.03	2.25
S3	1331.68	3.79	73.60	0.31	6.32
S4	1815.20	6.14	83.72	1.01	7.75
S5	2134.83	8.46	88.17	2.08	8.53
S6	2714.97	15.10	93.38	4.39	9.66
S7	4036.55	56.63	99.38	9.02	11.28
S8	4692.23	100.00	100.00	12.90	12.90

Table 3.2: the lower (γ_l) and upper (γ_u) percentage of black pears that can be encountered for the scenarios under investigation (S1-S8). Column (2) represents the maximum adult pest insect densities δ_{ppa} that are expected for each scenario. Column (3) and (4) represent the lower (γ_l) and upper (γ_u) percentage of black pears under the assumption that the overall maximum ∂_{Ppa} in the no-predator scenario S8 results in 100% black pears. Column (5) and (6) represent the lower (γ_l) and upper (γ_u) percentage of black pears under the assumption that the ETL equals 1% of black pears, corresponding to a potential maximum of black pears of 12.90%.

3.5.3 Economic impact of natural predator losses

The economic impact of a loss of natural predators is first discussed for the first set of hypothesized relationships, which assumed that the loss of three predators could result in 100% black pears.

The gross revenues for S1 ranged between 12856 €ha^{-1} and 23835 €ha^{-1} with a mean of 18261 €ha^{-1}. The reduction in mean gross revenues for S2 (respectively S3-S8) constituted 2.9% (respectively 18.41%, 27.49%, 33.69%, 45.10%, 79,34% and 86.98%) resulting in an average I_G of 217731€ha^{-1} (respectively 14899 €ha^{-1}, 13241 €ha^{-1}, 12109 €ha^{-1}, 10026 €ha^{-1}, 3773 €ha^{-1} and 2377 €ha^{-1}). Hence, for the loss of the three predators, the average gross revenues decreased from 18261 €ha^{-1} for S1 to 2377 €ha^{-1} for S8. The net farm income (figure 3.2) also reveals large losses under the assumption that the loss of three predators can yield 100% black pears. The mean farm income I_F for S1 with three natural predators (n) was 11921 €ha^{-1} and decreased to -3962 €ha^{-1} for S8 with the loss of three predators (n-3).

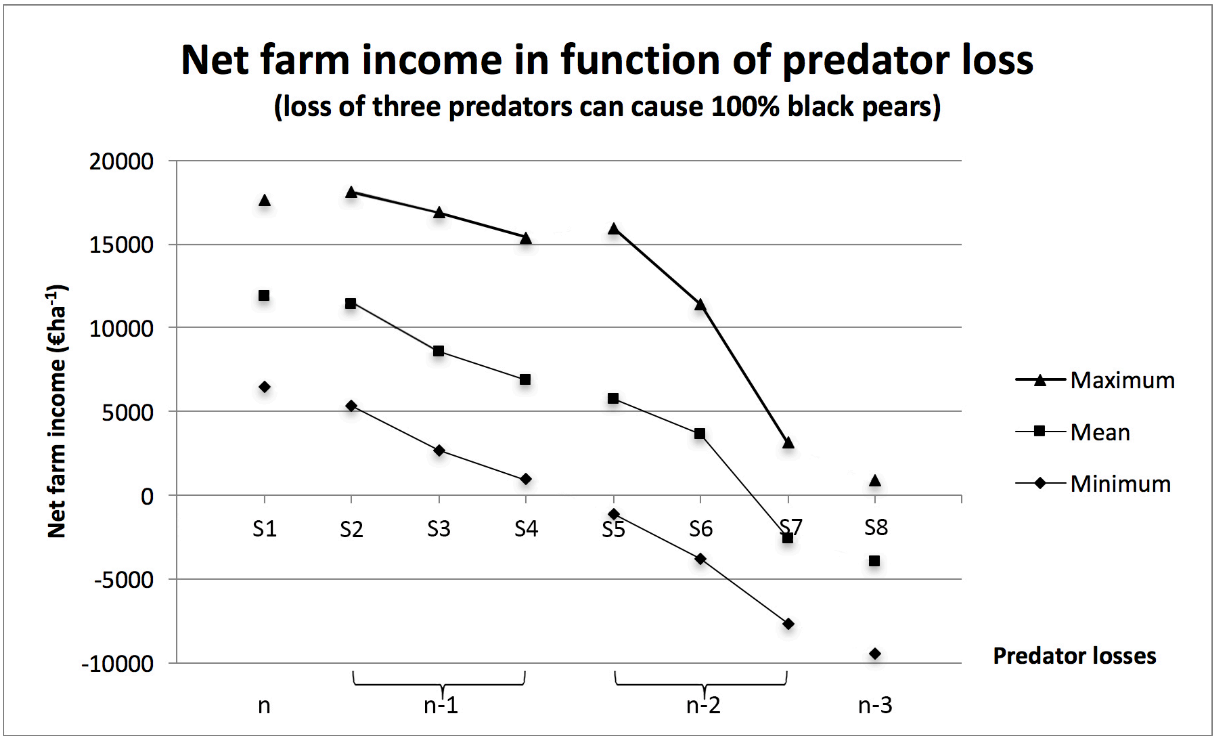

Figure 3.2 represents the effect of a loss of one or more natural predator on the net farm income I_F (€ha^{-1}) under the assumption that the loss of all three predators can result in 100% black pears (with n all predators present for S1; n-1 the loss of one predator for S2, S3 and S4; n-2 the loss of two predators for S5, S6 and S7; and n-3 the loss of all three predators for S8). The 95% confidence intervals are represented as the minimum and the maximum and are plotted together with the mean for each scenario. The graph shows that for the loss of all three predators, the mean net farm income for S1 reduces from 11921 €ha^{-1} to -3962 €ha^{-1} for S8.

Next, the economic impact of a loss of natural predators is discussed for the second set of hypothesized relationships, which assumed that the loss of three predators could result in an overall maximum of 12.90% black pears.

Under this assumption, the mean gross revenues I_G for S1 reduce from 18500 €ha^{-1} to 16313 €ha^{-1} for S8, constituting a loss of 2187 €ha^{-1} or 11,82 % for the loss of all three predators. The mean net farm income I_F (figure 2.3) reduces from 12161 €ha^{-1} for S1 to 9974 €ha^{-1} for S8, also constituting a loss of 2187 or 17,98 % for the loss of all three predators. The losses on a per hectare basis vary between 1941 €ha^{-1} and 2531 €ha^{-1} for S1 compared to S8. All the results for the gross revenues and the net farm income are presented in table 3.3.

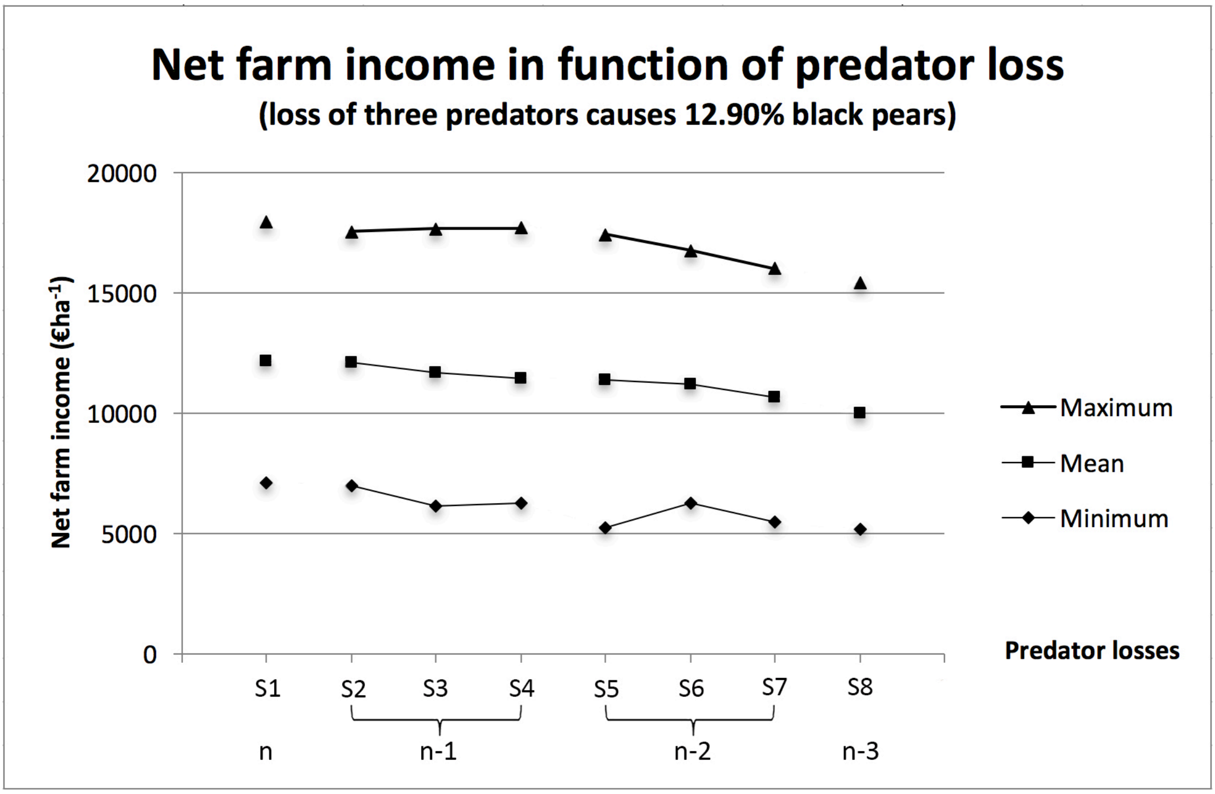

Figure 3.3 represents the effect of a loss of one or more natural predator on the net farm income I_F (€ha^{-1}) under the assumption that the ETL equals 1% black pears (with n all predators present for S1; n-1 the loss of one predator for S2, S3 and S4; n-2 the loss of two predators for S5, S6 and S7; and n-3 the loss of all three predators for S8). The 95% confidence intervals are represented as the minimum and the maximum and are plotted together with the mean for each scenario. The graph shows that for the loss of all three predators, the mean net farm income for S1 reduces from 12161 €ha^{-1} for S1 to 9974 €ha^{-1} for S8.

Scenario	Loss of three predators causes 100% black pears				Loss of three predators causes 12.90% black pears			
	min	max	mean	stdev	min	max	mean	stdev
	GROSS REVENUES (€ha^{-1})							
S1	12856,3	23834,94	18260,68	1944,92	13227,04	24280,28	18499,78	2028,19
S2	11739,73	24203,07	17730,51	2043,76	13207,21	23877,41	18410,92	1997,01

S3	9234,34	23200,83	14898,57	2329,98	12476,74	24158,11	18040,56	1921,93
S4	7410,81	21788,05	13241,45	2487,25	12788,47	23938,64	17789,06	1963,86
S5	5075,61	22270,21	12108,94	2512,07	11812,83	23620,97	17735,32	1960,43
S6	2692,53	17836,26	10025,62	2565,14	12567,21	22959,54	17516,96	1910,06
S7	-1095,99	9653,07	3773,27	1749,26	11806,73	22142,97	16994,41	1868,49
S8	-3128,91	7227,23	2377,36	1778,3	11591	21634,32	16313,27	1840,14
	NET FARM INCOME (€ha^{-1})							
S1	6440,26	17621,08	11921,49	1956,64	7082,07	17908,47	12160,6	2032,66
S2	5384,04	18080,43	11391,35	2053,67	6957,19	17537,69	12071,74	2001,95
S3	2688,18	16904,73	8559,41	2332,45	6120,66	17660,34	11701,39	1935,03
S4	945,09	15384,3	6902,27	2487,09	6272,24	17685,12	11449,9	1977,06
S5	-1096,02	15937,79	5769,77	2505,61	5250,49	17396,57	11396,15	1971,96
S6	-3753,8	11385,11	3686,44	2567,32	6247,29	16741,57	11177,8	1912,34
S7	-7651,83	3138,49	-2565,92	1751,27	5460,22	15988,82	10665,26	1868,96
S8	-9443,79	878,18	-3961,8	1784,15	5141,26	15377,25	9974,1	1836,61

Table 3.3: shows the minimum, maximum, mean and standard deviation for the gross revenues (€ha^{-1}) and the net farm income (€ha^{-1}) for scenario S1 to S8 under the assumption that the loss of three predators causes 100% of black pears, and under the assumption that the loss of three predators causes a maximum of 12.90% of black pears.

For both sets of hypothesized relationships, the net farm income reduces when natural predators are lost, thereby supporting Hypothesis 2.

3.5.4 An indirect use value for the presence of natural predators

The losses with respect to the gross revenue show results very similar to the losses with respect to the net farm income but differ greatly between the two sets of hypothesized relationships. Under the assumption that the overall maximum ∂_{Ppa} in the no-predator scenario S8 results in 100% black pears, gross revenue for the removal of one predator indicate a loss of I_G between 530.17 €ha^{-1} and 5019.23 €ha^{-1}. A loss of two natural predators would result in I_G losses between 6151.74 €ha^{-1} and 14487.41 €ha^{-1} and the removal of all predators caused a loss of 15883.32 €ha^{-1}. With regards to the net farm income I_F, results are in the same order of magnitude with the loss of one natural predator resulting in a loss of I_F between 530.14 and 5019.22 (€ha^{-1}). A loss of two natural predators would result in I_F losses between 6151.72 €ha^{-1} and 14487.41 €ha^{-1} and the removal of all predators caused a loss of 15883.29 €ha^{-1}.

Under the assumption that the loss of natural predators can cause a maximum of 12.90% black pears, gross revenue reductions for the removal of one predator indicate a loss of I_G between 88.86 €ha^{-1} and 710.72 €ha^{-1}. A loss of two natural predators would result in I_G losses between 764.46 €ha^{-1} and 1505.37 €ha^{-1} and the removal of all predators caused a loss of 2186.51 €ha^{-1}. With regards to the farm income I_F, results are again in the same order of magnitude with the loss of one natural predator resulting in a loss of I_F between 88.86 €ha^{-1} and 710.70 €ha^{-1}. A loss of two natural predators would result in I_F losses between 764.46 €ha^{-1} and 1495.34 €ha^{-1} and the removal of all predators caused a loss of 2186.50 €ha^{-1}. The net farm income losses for both hypotheses are presented in table 3.4.

Scenario	Loss of three predators causes 100% black pears	Loss of three predators causes 12.90% black pears
	Net farm income losses (€ha^{-1})	Net farm income losses (€ha^{-1})
S2	530.14	88.86
S3	3362.08	459.21
S4	5019.22	710.70
S5	6151.72	764.45
S6	8235.05	982.80
S7	14487.41	1495.34

S8	15883.29	2186.50

Table 3.4: shows the losses to the net farm income (€ha^{-1}) for all scenarios S1 – S8 under the assumption that a loss of three predators can cause 100% black pears and under the assumption that the loss of three predators causes 12.90% black pears.

3.6 Discussion

The results support Hypothesis 1 that a decrease in natural predators causes a significant decrease in the provisioning of the ecosystem service biological pest control from 10.72% for S1 to a minimum of 1.08% for the loss of one predator, further reducing to 0.02% for the loss of three predators, or equal to a total potential reduction with a factor 536 for the loss of two species. Also, the analysis showed that a reduction in natural predators could considerably reduce the quality of marketable agricultural production and that this depends highly on the hypotheses used. The first set of hypothesized relationships assumed that the total yield could consist of black pears only if all three predators would no longer occur in the agro-ecosystem. The second set of hypothesized relationships assumed that the Economic Threshold Level (ETL) equaled 1% of black pears, fixing the maximum potential of black pears upon losing the three predators at 12.90%. The economic results for the first set revealed losses of up to 15883 €ha^{-1} for the loss of three predators, making pear production financially unviable. The results for the second set reveal losses of up to 2186 €ha^{-1} when losing all three predators. Considering the fact that pear psylla has other natural predators (e.g. *Theridion spp*., *Philodromus spp*., members of the Araneidae and the seven-spot ladybird) (Erler, 2004)), it seems likely that the combined effect of all predators keeps pest densities within economic threshold levels, thereby supporting Hypothesis 2 that the three predators under analysis could induce a maximum of 12.90% of lower quality pears. On a per hectare basis, the occurrence of lower quality yields could therefore decrease gross revenues or net farm income with 88.86 € to 2186.5 €. For the pear production sector in Flanders in 2011, this would mean an indirect use value of 0,68 million € for one predator and 16.63 million euros for three predators. Considering that the gross revenues for the sector totaled on average 163 million euros for the period 2009-2013, the contribution of the predators accounts for 0,41% to 10.2% of the sectors' gross revenues.

By employing the ecological role of species through the development of an ecological simulation model, combined with a production function technique and a direct market valuation approach, we believe that economic values of non-marketable species could be estimated more realistically as compared to employing WTP estimates. This is largely due to the fact that the importance of lesser-known species to perform valuable ecological services is not known by the general public, and therefore this might impact the valuation of these species. Therefore, according to Hypothesis 3, we are convinced that the methodology applied here could contribute to the introduction of alternative methods for the valuation of biodiversity based on the ecological role of species. Research from Boerema et al. (2017) supports this hypothesis since: (i) their results show that, up until now, there was no paper on biological control examining the whole ES 'cascade', (ii) it is stated that '*measures of ecosystem functions are stronger as they give a better idea of ES supply and how this fluctuates spatiotemporally'* as compared to '*simple measures or indicators of biodiversity and population size'*, (iii) they recommend that net value, defined as "*the market price corrected for production costs*…", "*is a more appropriate measure to determine the added value*" and last, (iv) "*To quantify the sustainable supply of an ES, it is necessary to quantify the properties and functions of an ecosystem (ecological side of the cascade), whereas to quantify the importance to society it is necessary to understand and quantify the benefit to society (socio-economic side). Many researchers are only considering one side of this cascade and therefore are not succeeding in understanding the whole picture."*

The results of applying a functional role-based approach, shows that losses of natural predators for pear production could significantly reduce a farmer's income. The results of this analysis need to be viewed within a wider framework of (1) the partitioning of biodiversity effects on function into species richness, species composition and abundance effects and (2) functional redundancy.

First, in this analysis the number of predators was reduced, which also reduced total predator biomass. The resulting effects on net farm income can therefore not solely be attributed to a decline in species richness. In Winfree et al. (2015) biodiversity effects on function were split into five additive components according to the Price equation: species richness losses (RICH-L), species richness gains (RICH-G), species composition effects that capture any non-randomness with respect to function of the species that were lost (COMP-L) and of the species that were gained (COMP-G) and changes in abundance of species that are always present (ABUN) (Fox, 2006;

Fox&Harpole, 2008; Fox & Kerr, 2012). Winfree et al. (2015) stated that "*abundance fluctuations of dominant species in real world conditions drives ecosystem service delivery, whereas richness changes were relatively unimportant because they primarily involved rare species that contributed little to function."* Also, Winfree et al. (2015) revealed that "…*random loss of species has (or would have) large functional effects, and that the identity of the species that are lost is also important*". Although we cannot be sure on the nature of the losses and how much each component contributes to the effects on net farm income, this does not undermine the overall effect that a reduction in the number of predators and their biomass can potentially have on farm income.

Second, the indirect use value for the presence of natural predators depends highly on the functional redundancy of these species. The concept of functional redundancy is based on the principle that some species perform similar roles in ecosystems and might therefore be substitutable with little impact on ecosystem processes (Lawton and Brown, 1993). Therefore the effect of species loss depends on (i) the range of functions and the diversity of species within a functional group, (ii) the relative partitioning of variance in functional space between and within functional groups, and (iii) the potential for functional compensation of the species (Rosenfeld, 2002). Whilst *Anthocoris nemoralis*, *Allothrombidium fuliginosum* and *Heterotoma planicornis* are all natural predators of *Cacopsylla pyri,* one might assume that they are functionally redundant and that the impact of the loss of one natural predator does not significantly alter the impact on biological pest control. However, it is argued here that although providing the same function they are not functionally redundant due to (i) exertion of ecological function occurring on different time scales: species that occur on critical timings *e.g.* when high pest density levels are expected, can be considered of higher functional importance, (ii) differences in duration of ecological function, (iii) differences in degree of specialization: whilst some species thrive in a wide variety of environmental conditions, some require specific conditions for survival, rendering them less resilient to external shocks (iv) differing impacts on other species in the ecosystem due to predation preferences: generalists versus specialists, (v) attacking different pest stages and (vi) the absolute numbers of predators. The relationship between functional redundancy and economic value of species can be represented as an exponential decline whereby the marginal value of the loss of the first species is small and the loss of the last species is infinite. Therefore, the economic values represented in this analysis do not reflect values on either of the extreme ends of the marginal value curve. It is argued here that although species perform the same function, they are not functionally redundant, that the loss of one species or abundance of the species can significantly alter the provisioning of ecological functions and that attributing an indirect use value to the loss of one species is justified. Furthermore, our simulation model does effectively take into account differences in timing, duration and prey preference. The indirect use value therefore reflects the functional differences and effectively takes into account the importance of the different species for the biological pest control of *Cacopsylla pyri*.

Finally, of equal importance in this analysis is the fact that the economic valuation of biodiversity is regarded as just one of the aspects that could strengthen the argument in favor of biodiversity conservation and hence needs to be viewed within a wider framework of biodiversity valuation. Biodiversity is by nature a multidimensional concept and expressing the importance of biodiversity in economic terms does by no means exclude the presence of an intrinsic value (Feest et al., 2010). It is our opinion that choosing the most effective valuation methodology depends both on the context as well as on the species involved. When it considers species with a high socio-cultural value, economic valuation may not be needed and its socio-cultural value alone may be sufficient to ensure protection. However, when it concerns species that do not possess such an explicit socio-cultural value (as it in our case with insects or natural predators) additional arguments such as economic valuation may strengthen the argument in favor of conservation. Within this wider framework of valuation, it is our belief that *if* an economic argument for biodiversity conservation is needed, an ecological function approach may reveal more objective values than the application of stated preference techniques, due to the complex nature of the biodiversity and ecosystem services concept on behalf of the general public.

3.7 Conclusion

It is the aim of this paper to emphasize the importance of healthy agro-ecosystems, not only for the purpose of food production but also for the contribution to the farmer’s income. It is stressed here that effective valuation of biodiversity can include both intrinsic as well as economic arguments but that, in order to take into account the effect of biodiversity losses in economic arguments, it is imperative that the ecological function is taken into account. This implies some challenges. First, modeling real systems is rarely simple and the reality shows a great variability both in ecological as well as in economic parameters. The analysis provided here therefore provides an indication of the effect of the loss of species on the provisioning of biological pest control and on the decrease of quality. Furthermore, the authors point out the limitations of the use of stated preference techniques when valuing complex concepts such as biodiversity and ecosystem functioning. Willingness To Pay may not reflect the true ecological service that is provided by beneficial insects, since only a part of the general public has limited knowledge of the concept. Our analysis therefore provides an alternative methodology for the valuation of biodiversity, taking into account the ecological function of species in the ecosystem, hereby revealing values linked to marketable agricultural outputs. Using an ecological function based approach, values for the presence of species diversity could be considered more objective compared to stated preference methods. These values could be supplied to inform policy makers about the importance of including biodiversity effects and providing a justification for the opportunity costs encountered.

Acknowledgments

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ANNEX A

Each field test sampled pear psylla eggs and nymphs on multiple days. The first dataset comprises a total number of 111 field tests in *conférence* pear orchards (7 in organic production and 104 in IPM (Integrated Pest Management)) on 15 different plots (8 in IPM and 7 in organic production) performed in Haspengouw (Belgium) for consecutive years of measurement (2004-2014). Data obtained from the plots under organic management were sampled in 2013 and 2014. Using the beating-tray method (3 beatings x 3 branches x 10 trees plot^{-1}), the nymph stages N1 to N5 are collected in a beating tray and counted (for a review of sampling methods see Jenser et al., 2010). A visual count is performed on newly developed shoot tips to assess the presence of eggs (visual counts are performed for 2 shoots per tree for 4-10 trees per plot segment with 4 plot segments per plot). Adult counts were performed sporadically with the beating-tray method but have not been included in the data due to its susceptibility to bias caused by adult mobility and the dependency on weather conditions. The mean counts of eggs per ten shoots are pooled for all consecutive years and plotted in figure A.1. For the years of measurement, it can be observed that counts in IPM orchards are considerably higher than counts in organic orchards.

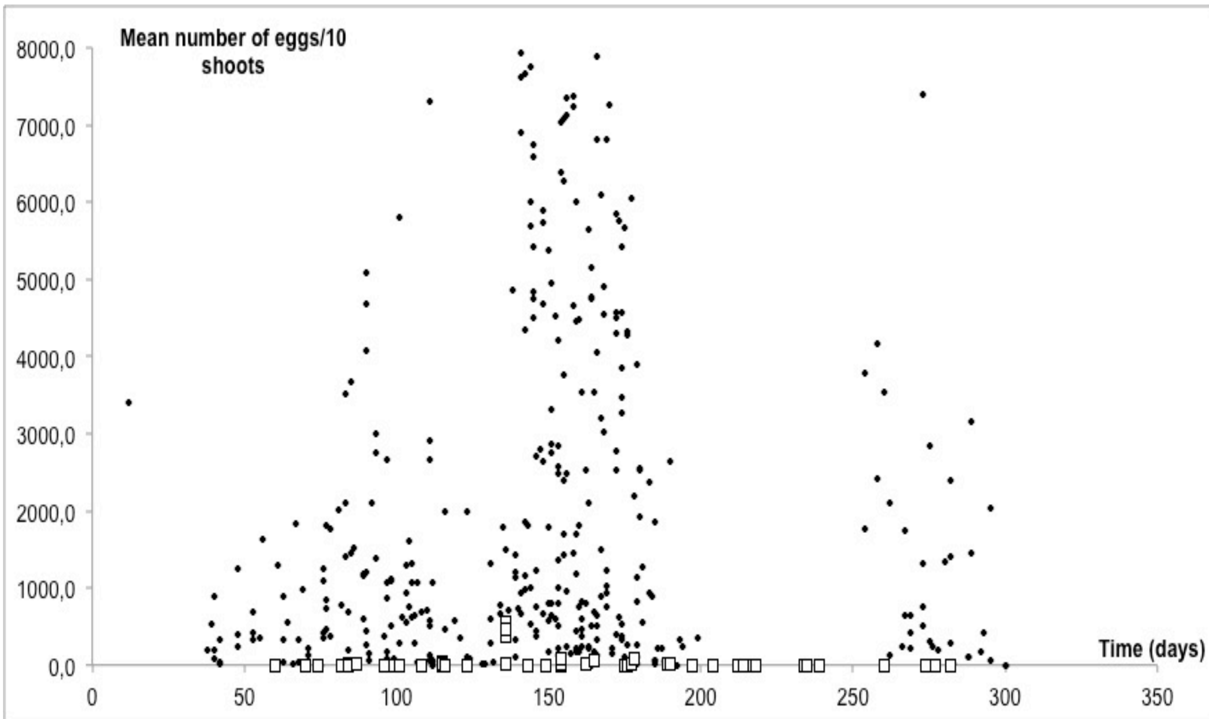


Figure A.1: pooled sample of mean numbers of pear psylla eggs per ten shoots collected between 2004 and 2014 (♦IPM; □ organic).

In 2013 and 2014, counts for the presence of beneficial insects were been performed between February and October in IPM and organic *conference* pear orchards. Linear transects of three pitfall traps (r=0.2m) per 50m per pear row for three rows per plot were filled with water and detergent and left standing for 7 days. Emptying of the containers produced members of the order of the Aranea, Acari, Coleoptera, Hemiptera and Neuroptera. Figure 2 represents the pooled counts for a selection of the species in the samples collected based on the importance of their functional role as natural predators of pear psylla *Cacopsylla pyri* (Homoptera: psyliidae): *Anthocoris nemoralis* (Heteroptera: anthocoridae), *Allothrombidium fuliginosum* (Acari: trombidiidae) and *Heterotoma planicornis* (Hemiptera: miridae).

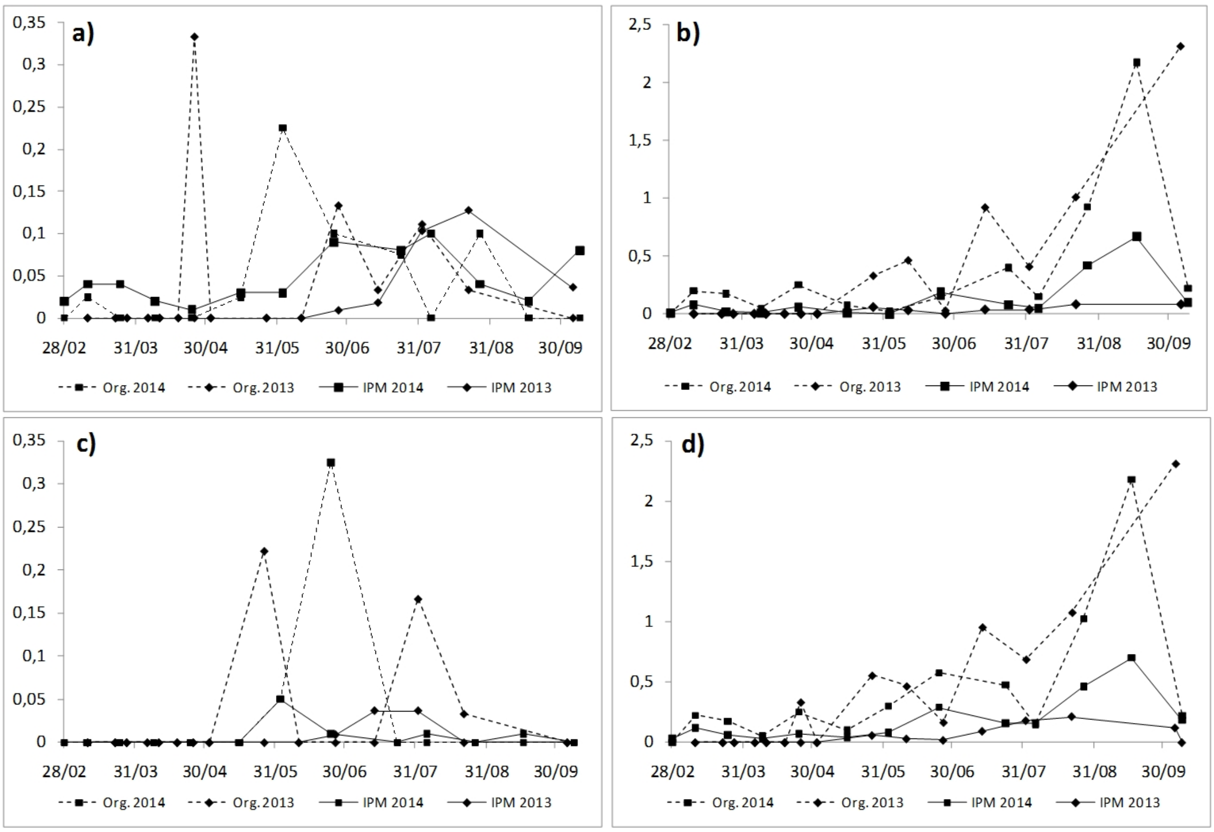
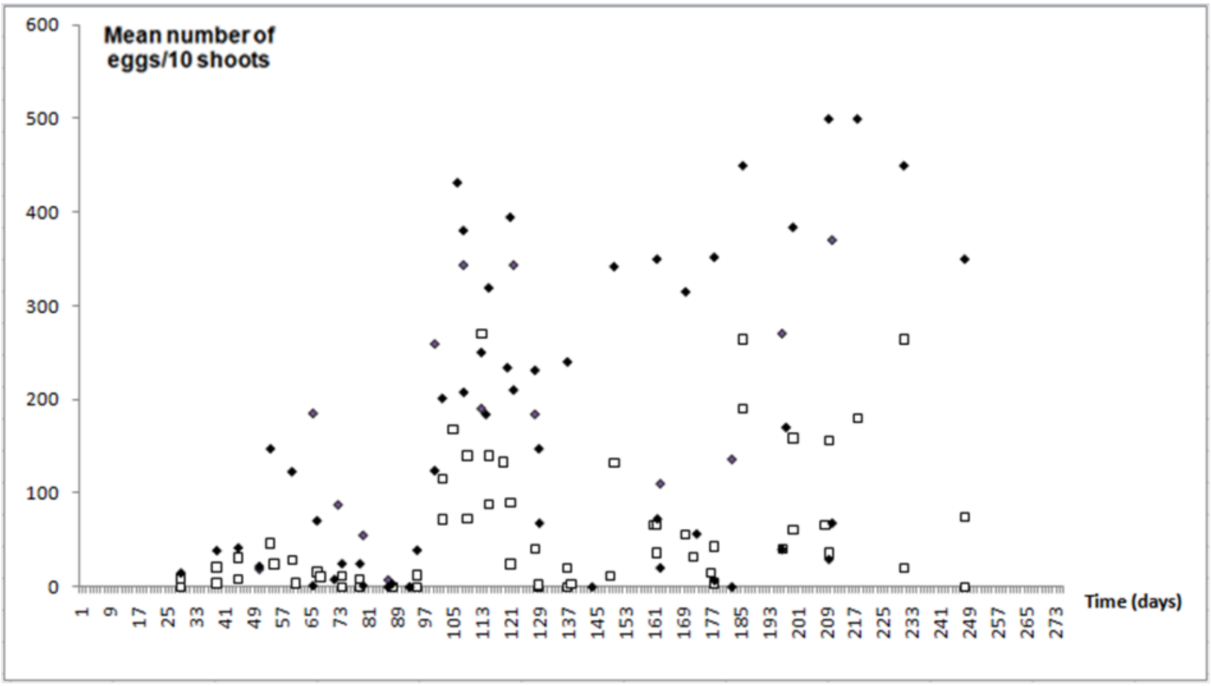


Figure A.2: absolute number of individuals per sample for a) *Anthocoris nemoralis, b) Allothrombidium fuliginosum*, c) *Heterotoma planicornis* and d) sum of the absolute numbers of a, b and c.

Figure A.2 shows (i) the difference in abundance levels of the three natural predators and (ii) the timing of occurrence. These two factors combined with their generalist/specialist nature determine the importance as natural pest controllers. Whilst *Allothrombium fuliginosum* (b) may be abundant, it is not a specialist and it preys on other insects than *Cacopsylla pyri*. *Anthocoris nemoralis* (a) is less abundant but is a specialist and therefore qualifies as a rare but highly effective pest controller. Last, *Heterotoma planicornis* (c) is both rare and a generalist and therefore differs from the two other predators.

Whilst the predators differ in terms of their generalist/specialist nature and their levels of abundance, they also differ in the timing of occurrence. Whilst *Anthocoris nemoralis* (a) is mainly encountered during the first half of the year, *Hetertoma planicornis* (c) is mainly found in the middle of the year whilst *Allothrombium fuliginosum* (b) is the main predator at the end of the year. So even when *Anthocoris nemoralis* (a) can be considered a rare species, they are highly effective and important given their ability to suppress the build-up of the pest population in the beginning of the season. The removal of one individual in the beginning of the year has an exponential effect on the pest insect density later that year, making the presence of predators in the beginning essential for controlling pest outbreaks. Equally so, *Allothrombium fuliginosum* (b) is an abundant species occurring at the end of the season, suppressing the population before the build-up in the new season.

The second dataset was obtained from field test performed every two weeks for the period 2010-2011 on 14 plots (7 in organic production and 7 in IPM) in Hageland (BE) and Gelderland and Limburg (NL). The same techniques were used to assess mean egg numbers and larvae numbers (visual scouting and beating tray method).



<u>Figure A.3</u>: Pooled sample of mean numbers of pear psylla eggs per ten shoots (♦IPM; □ organic).

Data obtained from the plots under organic management were sampled in 2013 and 2014. Using the beating-tray method (3 beatings x 3 branches x 10 trees plot^{-1}), the nymph stages N1 to N5 are collected in a beating tray and counted (for a review of sampling methods see Jenser et al., 2010). A visual count is performed on newly developed shoot tips to assess the presence of eggs (visual counts are performed for 2 shoots per tree for 4-10 trees per plot segment with 4 plot segments per plot). Adult counts were performed sporadically with the beating-tray method but have not been included in the data due to its susceptibility to bias caused by adult mobility and the dependency on weather conditions. The mean counts of eggs per ten shoots were pooled for all consecutive years and plotted.

ANNEX B

	Parameter	Model component	Initial value
(1)	Initialization adults	Ppa, Ana, Afa	$1.8 * 10^6$; 29520; $0.41*10^6$
(2)	Initialisation eggs	Hpe	$0.15 * 10^6$
(3)	Female fraction	Ppa, Ana, Afa, Hpa	0.5
(4)	Loss fraction (eggs)	Ppe, Ane, Afe, Hpe	0.3; 0.4; 0.65; 0.6
(5)	Pp Food fraction	Ann, Afn, Hpn, Ana, Afa, Hpa	0.8;0.8;0.2;0.2;0.2;0.2
(6)	Predation fraction	Ann, Afn, Hpn, Ana, Afa, Hpa	0.6

Table b presents initial parameter values for Pp, An, Af, Hp for eggs (e), nymps (n) and adults (a)

ANNEX C

NON-ORGANIC PRODUCTION				
	Mean	stdev	95% confidence interval	
Total yield (kgha^{-1})	37615,33	4565,36	33962,29	41268,38
Selling price all pears(€kg^{-1})	0,57	0,16	0,44	0,70
Selling price 1st class pears(€kg^{-1})	0,55	0,16	0,42	0,68
Selling price black pears(€kg^{-1})	0,39	0,12	0,29	0,49
GROSS REVENUES (€ha^{-1})				
Main products	20247,67	3654,52	17323,44	23171,89
Plantation growth	207,00	34,05	179,75	234,25
Other products	96,83	127,62	-5,28	198,95
Subsidies	140,00	55,00	95,99	184,01
VARIABLE COSTS (€ha^{-1})				
Fertilizers	362,33	39,51	330,72	393,94
Crop protection	1579,83	100,12	1499,72	1659,94
Seasonal wages and labour	4118,33	352,15	3836,56	4400,11
Maintenance, packaging and preservation	1329,33	62,64	1279,21	1379,46
Energy	799,33	85,55	730,88	867,79
Other variable costs	260,50	23,68	241,55	279,45
FIXED COSTS (€)				
Lease/rent	463,00	76,87	401,49	524,51

Amortization fixed equipment	1274,17	35,72	1245,59	1302,75
Amortization buildings	1033,50	85,93	964,74	1102,26
Amortizations plantations	392,83	8,77	385,81	399,85
Interests	1450,00	31,25	1424,99	1475,01
General corporate costs	1692,67	275,62	1472,13	1913,21

ORGANIC PRODUCTION

	Mean	stdev	95% confidence interval	
Total yield ($kgha^{-1}$)	30092,27	3652,28	27169,83	33014,70
Selling price all pears(€kg^{-1})	0,57	0,16	0,44	0,70
Selling price 1st class pears(€kg^{-1})	0,88	0,17	0,74	1,02
Selling price black pears(€kg^{-1})	0,39	0,12	0,29	0,49
GROSS REVENUES (€ha^{-1})				
Main products				
Plantation growth	207,00	34,05	179,75	234,25
Other products	96,83	127,62	-5,28	198,95
Subsidies	210,00	105,00	125,98	294,02
VARIABLE COSTS (€ha^{-1})				
Fertilizers	362,33	39,51	330,72	393,94
Crop protection	0,00	0,00	0,00	0,00
Seasonal wages and labour	5353,83	457,79	3836,56	5635,61
Maintenance, packaging and preservation	1329,33	62,64	1279,21	1379,46

Energy	799,33	85,55	730,88	867,79
Other variable costs	260,50	23,68	241,55	279,45
<u>FIXED COSTS (€)</u>				
Lease/rent	463,00	76,87	401,49	524,51
Amortization fixed equipment	1274,17	35,72	1245,59	1302,75
Amortization buildings	1033,50	85,93	964,74	1102,26
Amortizations plantations	392,83	8,77	385,81	399,85
Interests	1450,00	31,25	1424,99	1475,01
General corporate costs	1692,67	275,62	1472,13	1913,21

(Van der Straeten, 2016; Personal communication from Regional Auction Borgloon)

Table C presents annual accounting data on yields (kg ha^{-1}), revenues (€ ha^{-1}), variable costs (€ ha^{-}1) and fixed costs (€) for non-organic production and organic production from the Agricultural Monitoring Network (LMN) data (Van der Straeten, 2016), which are conform FADN[14] data collection procedures. The LMN dataset contains 53 non-organic pear farmers (accounting for 662 hectares) and provides means, standard deviations and the 95% confidence interval based on annual accounting data for the period 2009-2014 (Van der Straeten, 2016). Some numbers were adjusted to represent organic production taking into account the following assumptions: (1) yields (kgha^{-1}) are 80% of non-organic production with μ = 30092,27 kgha^{-1} and s = 3652,28[15], (2) organic management requires 30 % more full-time equivalents (FTEs) with μ = 4118,33 €ha^{-1} and s = 352,15 for non-organic production and μ = 5353,83 €ha^{-1} and s = 457,79 for organic production (EC, 2013).

[14] Farm Accounting Data Network

[15] With μ the average and s the standard deviation

ANNEX D

Model calibration for organic production based on field data from 2010, comparing the pooled field sample (eggs/ten shoots) with the organic model results (eggs ha^{-1}).

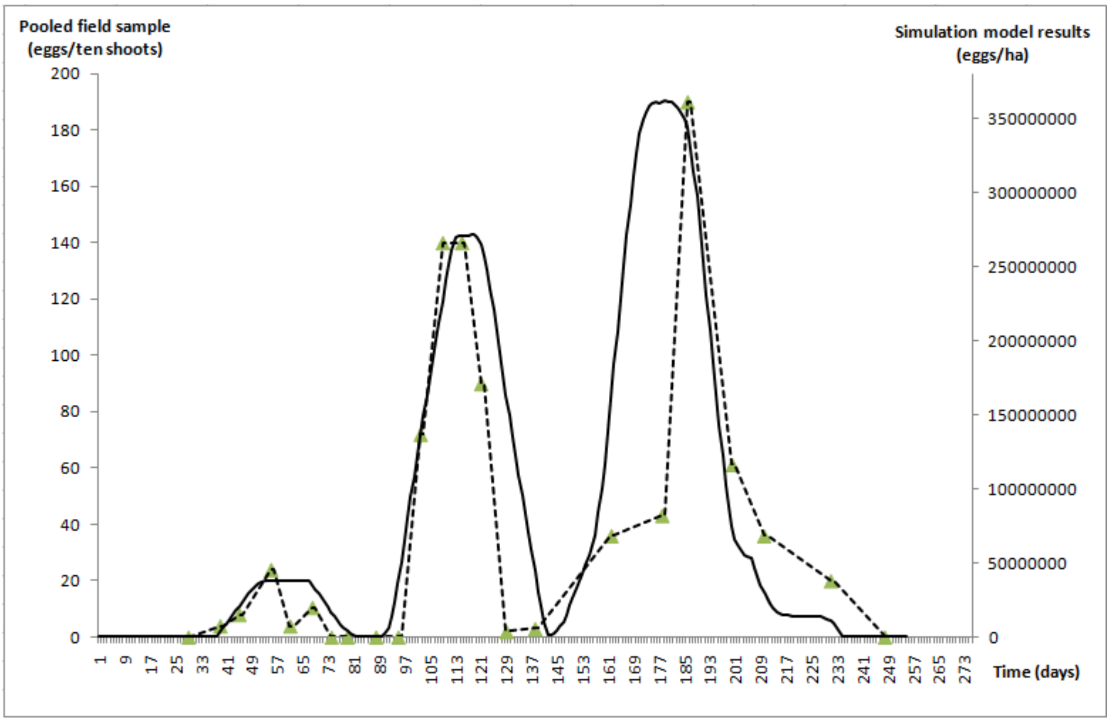


Figure D: Model calibration for organic production based on field data from 2010, comparing the pooled field sample (eggs/ten shoots) with the organic model results (eggs ha^{-1}) (-simulation model, -- field sample data). The units of field measurements (mean eggs/10 shoots) were transformed to yield model parameter units (absolute egg numbers per hectare), based on 33,84 shoots/tree on average, 5% of the eggs captured and 1714 trees per hectare (Van der Straeten, 2016).

CHAPTER 4

The Economic Value of Changes in Aquatic macro-invertebrate Diversity for Chinook Salmon Spawning

Parts of this chapter are under review in:

Daniels, S., Bellmore, J.R., Benjamin, J., Witters, N., Vangronsveld, J., Van Passel, S. Quantification of the Indirect Use Value of Functional Group Diversity based on the Ecological role of Species in the Ecosystem.

CHAPTER 4: The Economic Value of Changes in Aquatic macro-invertebrate Diversity for Chinook Salmon Spawning

4.1 Abstract

Chinook salmon, also referred to as "king" or "Tyee" salmon, are the largest species of Pacific salmon. Here it is examined what (i) the relationship between the diversity of aquatic macroinvertebrate prey and juvenile chinook salmon (*Oncorhynchus tshawytschas*) salmon in rivers and streams is, (ii) quantity of adult chinook salmon are later available to the commercial salmon fishery, and ultimately, (iii) the economic value of freshwater macroinvertebrate diversity is.

Here it is assessed whether the methodology employed in chapter 2 could be also be used in different circumstances: (i) a larger number of species, (ii) another type of ecosystem (freshwater river systems instead of an agricultural production system), and (iii) another ecosystem service (salmon production instead of biological pest control). Furthermore, chapter 4 expands the methodology by accounting for the contribution of the individual effects of changes in species richness, species composition and species abundance to determine the indirect use value of biodiversity.

4.2 Introduction

Chinook salmon, also referred to as "king" or "Tyee" salmon, are the largest species of Pacific salmon (Figure 3.1).


Figure 3.1: Chinook salmon (left) are the largest species of Pacific salmon, and have long been harvested for commercial purposes. The photograph on the right shows seine netters catching salmon on the Columbia River, Oregon, USA, circa 1914.

Due to their large size and high fat content, adult chinook salmon are a prized and highly sought-after resource by commercial, recreational and subsistence fisherman. The importance of chinook salmon for the economy stems from the annual commercial chinook salmon landings and values[16]. For the period 2000-2015, commercial chinook salmon landings averaged 8176 tons per year with an average yearly value of 4,3 million $ (or 5,24 $\$kg^{-1}$)[17].

Like other salmon species, chinook salmon have a complex life cycle that spans oceans, estuaries and rivers. Although chinook salmon generally spend a majority of their life in salt water, the first one to two years of their life is spent in freshwater enviroments, generally streams and rivers. During their freshwater residence salmon consume a variety of food resources, but aquatic macroinvertebrates species—especially insects (Figure 3.2)—make up a majority of their diet. For example, Bellmore and others (2013) observed 37 different aquatic macroinvertebrate taxa in juvenile Chinook salmon diets (Figure 3.2), and found that most of these taxa were important for fish growth. Although aquatic invertebrates have no direct value to humans, this and other studies suggest that changes in aquatic macroinvertebrate diversity could impact the capacity for streams to support juvenile salmon, which in turn, could impact the number and total value of adult salmon caught by the commercial fishing industry.

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[16] https://www.st.**nmfs.noaa**.gov *Fisheries of the United States*, issued annually by the National Marine Fisheries Service (NMFS) and the National Oceanic and Atmospheric Administration (NOAA) (last updated June 15th, 2017).

[17] http://www.st.nmfs.noaa.gov (last updated April 4th 2017).

Figure 3.2: Chinook salmon primarily forage on aquatic insects and other macroinvertebrates durig their freshwater residence. The images above are common aquatic insects consumed by juvenile chinook, which include (from left to right): mayflies (Ephemeroptera), stoneflies (Plecoptera) and caddisflies (Trichoptera).

4.3 Defining the relationship between macroinvertebrate diversity and their contribution to the fishing industry (step 1)

In a first step, the Ecosystem Services Cascade defines the scope and sets the boundaries for the analysis, linking the diversity of macroinvertebrate in freshwater river systems to the economic value created for the commercial fishing industry. This ultimately results in an indirect use value for freshwater macroinvertebrates (table 3.1).

Functional group (FG)		Ecosystem Properties (EP)	Ecosystem Function (EF)	Ecosystem Service (ES)	Benefit (B)	Value (V)
Freshwater aquatic macro-invertebrates in salmon streams	➔	1) Diversity and population parameters of the aquatic macro-invertebrates 2) Consumer-resource interactions 3) Inputs of energy, nutrients and organic matter 4) Environmental conditions	Food availability for juvenile salmon in fresh water	Number of adult salmon	Availability of salmon for salmon fishing industry	Annual revenues of the commercial fishing industry

Table 4.1: defining the Ecosystem Services Cascade to examine the relationship between freshwater aquatic macroinvertebrate diversity and their contribution to the commercial fishing industry.

The functional group to be valued are the macroinvertebrates in freshwater rivers and streams along the north Pacific coast where juvenile Chinook salmon reside. These flowing water-bodies generally contain a diversity of different macroinvertebrate species that are consumed by juvenile salmon (Bellmore et al., 2013; Nielsen, 1992; Reece and Richardson, 2000).

The four main ecosystem properties (EP) determining ecological function are:

1) Diversity and population parameters of the aquatic macroinvertebrates: *i.e.* species richness $s=25$, species composition i=1,2,3…, 25, biomass a_i (g/m^3) and functional contribution z_i.These macroinvertebrates provide a functional contribution z_i to the overall ecosystem function (EF) of interest, which is the availability of food resources necessary for the growth and survival of juvenile salmon (step 3).
2) Consumer - resource interactions *i.e.* predator-prey interactions
3) Inputs of energy, nutrients and organic matter
4) Environmental conditions: *i.e*. river discharge, water temperature, water clarity, dissolved nutrient concentrations, light availability, and channel hydraulics

The ecosystem services provided to humans are provisioning services in terms of the number juvenile chinook salmon that survive to adulthood. The benefits from the ecosystem services stem from the availability of these adult chinook salmon for the commercial fishing industry and human consumption. The value of benefits is derived from the annual revenues of the commercial fishing industry. Ultimately, the indirect use value is determined by the change in annual revenues due to changes in aquatic macroinvertebrates.

4.4 Quantitatively linking macroinvertebrate diversity to salmon survival (step 2)

The dynamic ecological model explores the relationship between freshwater macroinvertebrate diversity and the presence of Chinook salmon, by examining the food web reponses to changes in macroinvertebrate diversity. Because populations of many macroinvertebrates observe strong seasonal fluxuations in abundance, a dynamic ecological model is capable of accounting for these seasonal dynamics. The model used here is the Aquatic Trophic Productivity (ATP) model (Bellmore et al., 2017).

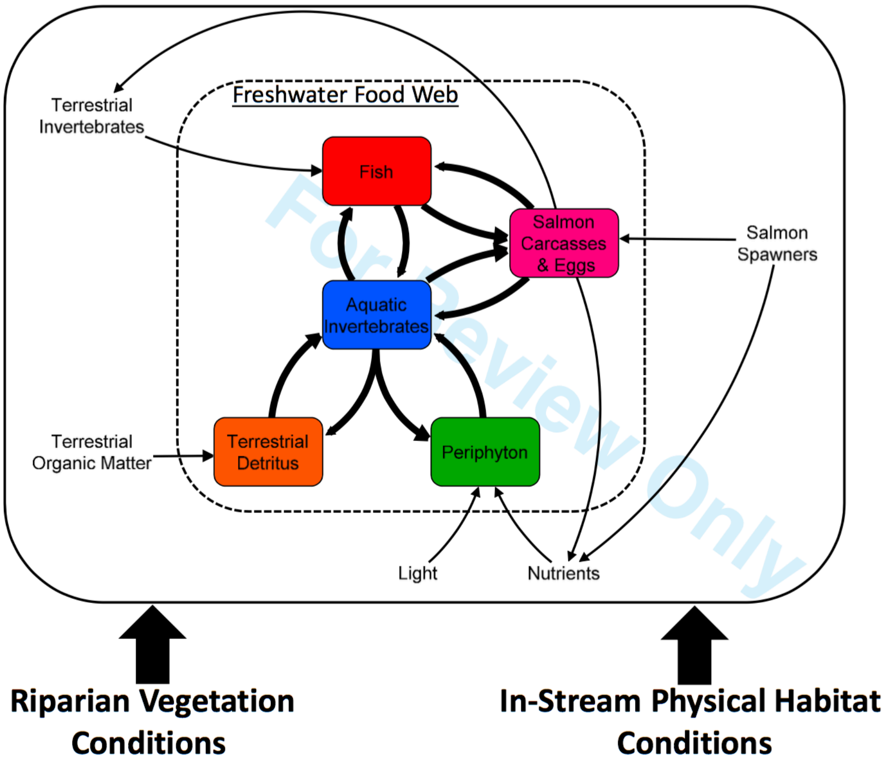

Figure 4.3: The Aquatic Trophic Productivity Model is a system dynamic model consisting of (i) biomass stocks, (ii) consumer-resource interactions, (iii) inputs of energy, nutrients and organic matter and (iv) linkages to in-stream physical habitat conditions and riparian vegetation conditions

The ATP model represents the generalized trophic structure of river food webs (Figure 4.3), whereby aquatic macroinvertebrate populations are linked to the dynamics of upper (fish) and lower trophic levels (periphyton and terrestrial detritus such as leaf litter) via a series of linked consumer-resource equations (see Bellmore et al. 2017). In turn, the strength of these consumer-resource interactions, are connected to the environmental conditions of the stream and the adjacent riparian zone. These environmental conditions include: river discharge, water temperature, water clarity, dissolved nutrient concentrations, light availability, and channel hydraulics (i.e., water depth, width and velocity). Water temperature, for example, influences consumption and respiration rates for all the members of the food web, including macroinvertebrates. The model simulates the biomass-dynamics of aquatic macroinvertebratres on a daily time-step in units of grams of ash-free-dry-mass per square meter of stream bed (g AFDM m^{-2}). For further details on the model see Bellmore et al. (2017).

As invertebrate populations fluctuate, either due to top-down predation by fish or variation in other environmental conditions, fish switch to forage on those macroinvertebrates that are most abundant. Fish consumption and growth is linked to juvenile fish survival in two ways: (1) starvation mortality, if food is limiting fish lose mass and succumb to starvation, and (2) size-based mortality, smaller fish have higher mortality rates than larger fish, thus, when fish grow faster (i.e., when macroinvertebrate food resources are plentiful) they "escape" higher mortality rates. Following this logic, reductions in macroinvertebrate diversity may result in longer periods of low food availability, higher juvenile salmon mortality, and ultimately, fewer salmon that grow to adulthood and are available for the commercial fishery.

The ATP model was used to simulate the dynamics of 25 different aquatic macroinverebrate species, which were coded into the model as 25 separate biomass stocks. Stocks were not coded to represent any specific set of macroinvertebrate species, but rather, physiology parameters (e.g., consumption and respiration rates, food preferences, foraging efficienies, temperature sensitivity, etc) were adjusted, via a randomization process, to create a diverse assemblage of macroinvertebrates that respond differently to environmental and food web conditions. Details on coding of the 25 macroinvertebrate species are in Annex 1.

We parameterized the model with environmental conditions (i.e., water temperature, dischare, channel hydraulics) representative of Pacific Northwestern streams where juvenile chinook salmon rear before migrating to the ocean. The spatial scale of the modeling exercise was restricted to a one-kilometer section of the Methow River system in the Northwestern USA, however, we interpret modeled results as representative of the general relationship between macroinvertebrate diversity and survival of chinook salmon to adulthood. Although we acknowledge that this relationship likely varies significantly across the range of Chinook salmon, this first-order approximation sets the stage of more location specific analyses.

4.5 Alternative scenario development (step 3)

Alternative scenarios were created by conducting removal experiments by iteratively removing one aquatic macro-invertebrate species at a time from the freshwater food web. A total of 100 removal experiments were conducted, each starting from the reference scenario R_r containing all 25 aquatic invertebrates (s =25). Each experiment randomly removed one species at a time until no species were left, resulting in 25 alternative scenarios per experiment. Therefore a total of 100 experiments and 2500 alternative scenarios were developed.

4.6 Quantifying changes in ecosystem function with reduction of macroinvertebrate diversity (step 4)

Aquatic macroinvertebrates provide many important ecological functions EFs (e.g., organic matter processing, nutrient cycling, etc.) in stream ecosystems. However, the EF of interest in this analysis is the amount prey or food resources macroinvertebrates provide to juvenile salmon. Thus, the change in total ecosystem function between the baseline scenario T and each alternative species removal scenario T' is the difference in total macroinvertebrate biomass (summed across all species).

4.7 Effects of macroinvertebrate diversity on adult salmon abundance (step 5)

The relationship between the ecosystem function, macroinvertebrate biomass, and the ecosystem service, adult salmon abundance was quantified using the dynamic equations contained within the ATP model (see section 4.2 and Bellmore et al. 2017). For the reference scenario (S_r) and each removal experiment whereby successively one species at random was removed, the resulting total number of salmon spawners Y was modeled (Figure 7). The reference scenario, including all 25 aquatic invertebrate species reveals 1005 individual salmon spawners. The average total number of salmon spawners $\bar{Y}$ for decreasing aquatic invertebrate richness was calculated resulting in $\bar{Y}$ for each level of species richness (i.e. $\bar{Y}$ for 24 random aquatic invertebrate species equals 996 and $\bar{Y}$ for 23 random aquatic invertebrate species equals 988). Also, the average number of salmon spawners per number of aquatic macroinvertebrate species ($\bar{Y}/s$) with s=1,2,…25 was analysed (Figure 4.4).

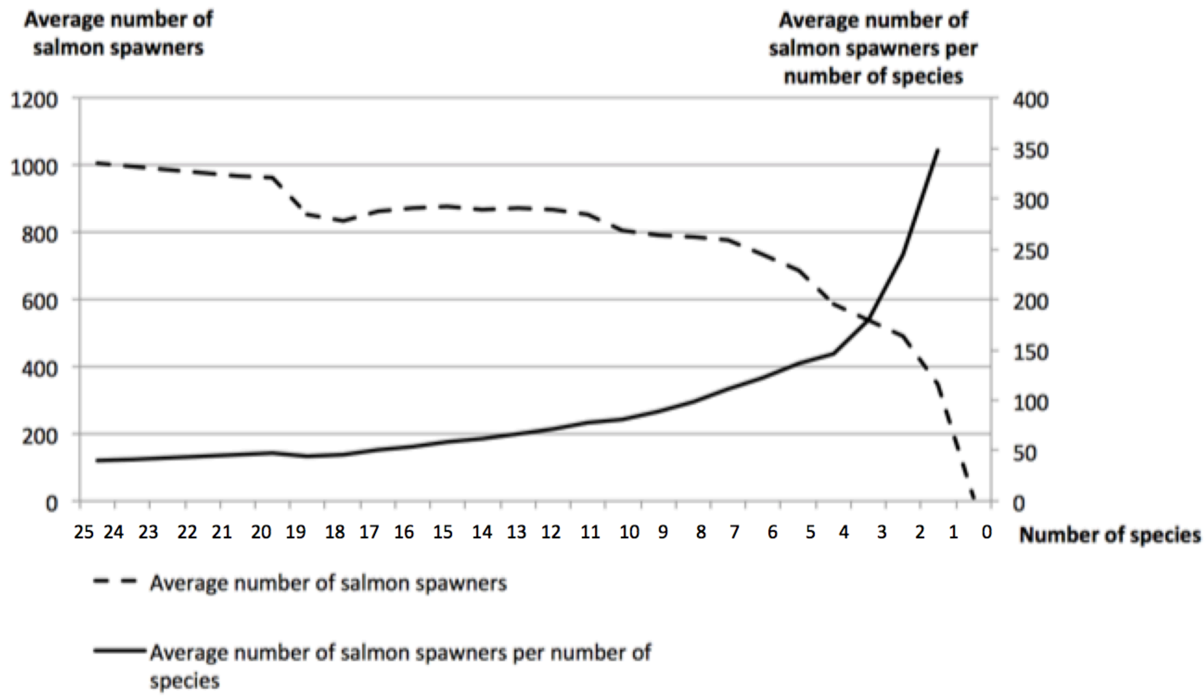

Figure 4.4: represents the average total number of salmon spawners $\bar{Y}$ and the average number of salmon spawners per number of species $\bar{Y}/s$ for s=1,2…25.

For each scenario developed, the individual functional contribution $x_i$of each aquatic invertebrate species s_i to the total number of salmon spawners Y is calculated and standardised per gram of biomass per m^2 for species s_i (Figure 4.5)

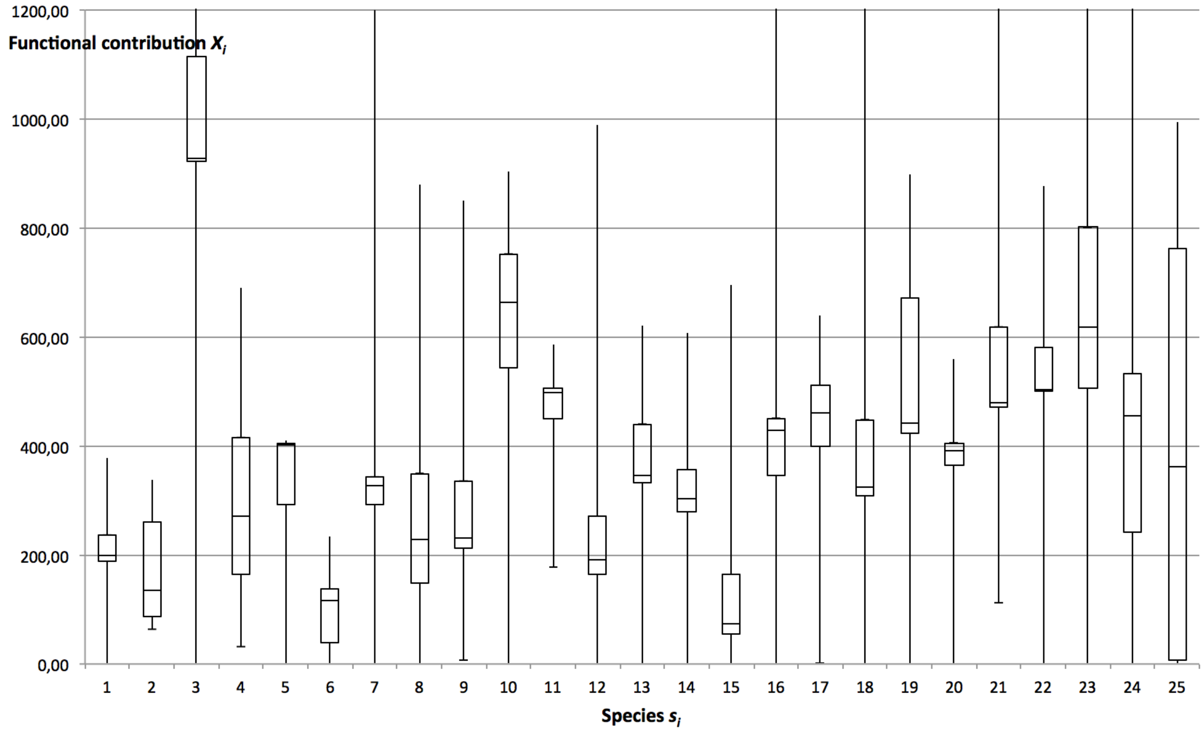

Figure 4.5: boxplot showing the functional contribution x_i of species i to salmon spawning Y (in number of salmon spawners)

Table 4.2 represents an overview of the results with column (1) to (4) the results from the dynamic ecological model, and column (5) to (14) the extrapolation of these results for their effect on the commercial fishing industry. They show that a reduction in species diversity of 1 decreases the total salmon spawners by 0,88% (column 4). A decrease of species diversity of 2 reduces total salmon spawners by 1,67% up until the complete loss of aquatic species diversity would result in the loss of all salmon spawners. The decrease in species diversity results in non-linear losses with relatively higher losses under lower species diversity, indicating the importance of a high species diversity.

4.8 Benefits of aquatic invertebrate species richness for salmon availability (step 6)

The benefits (B) derived from the change in ecosystem services $\Delta ES = ES - ES'$ are the related changes in catch by the commercial fishing industry. The decrease in aquatic invertebrate species diversity reduces the number of salmon spawners, thereby reducing the potential for commercial catch. For the period 2000-2015, commercial chinook salmon landings averaged 8,176 kton per year. The ecological economic linking function therefore links the provisioning of ecosystem services to the benefits delivered to humans. The relationship between abundance and catch per unit effort is represented by a logarithmic relationship (Guzzo et al., 2014) (figure 4.6; table 4.2).

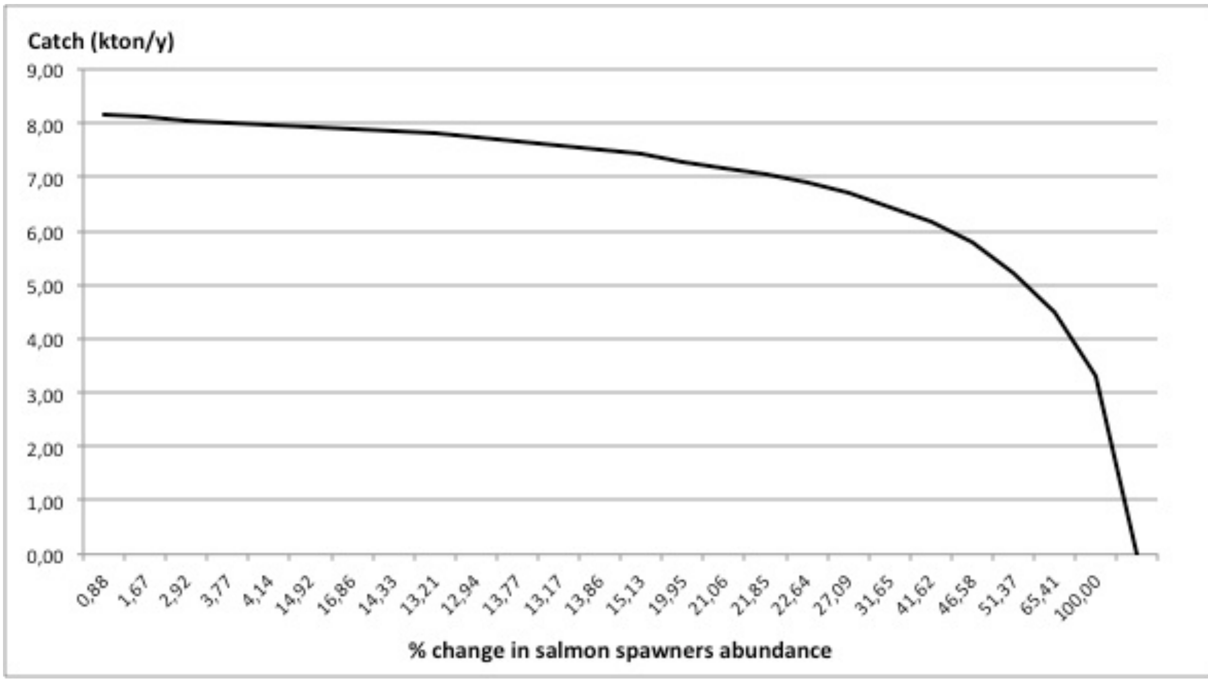

Figure 4.6: The ecological economic linking function shows the effect of a change in salmon spawner abundance on the catch of salmon.

Reducing the species diversity with 1 resulted in a decrease of 8,88% in the number of salmon spawners, thereby reducing catch from 8,18 to 8,14 kton per year. Due to the logarithmic shape of the function, the losses in catch have a higher impact when the decrease in salmon spawners abundance is higher (see table 3.2 column (5)).

4.9 Separating the effects of macroinvertebrate species richness, composition and abundance (step 7)

In order to separate the effects of richness, composition and abundance on the number of salmon spawners Y, all components of the Price equation are calculated according to eq.3 (figure 4.7).

RICH-G and COMP-G = 0 for all scenarios since no species were added. The results show that with high species diversity, the effect of species loss on the total number salmon spawners is relatively low (2%) and that the composition (50%) and abundance (48%) of species are the main determinants for functioning. However, when species diversity decreases, the effect of species loss becomes increasingly important (48%) while the effect of composition (4%) decreases in importance.

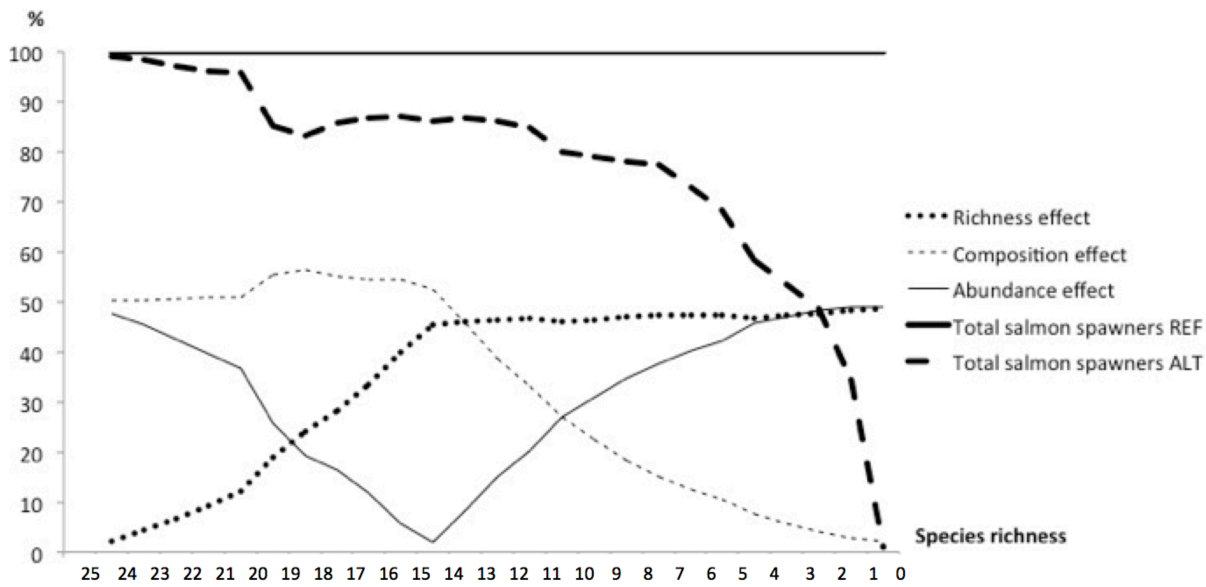

Figure 4.7 shows the effect of richness, composition and abundance on the number of salmon spawners Y at each level of macroinvertebrate species richness.

4.10 The economic value of salmon (step 8)

A reduction in catch due to reduced salmon presence has important effects for the commercial fishing industry's income generation and annual total catch value losses (table 4.2 column 7).

Column 12, 13 and 14 (table 4.2) represent the total catch value losses which can be attributed to changes in species diversity, changes in species composition and changes in species abundance and is represented in figure 11. The separation of effects reveal that species composition effects are the most important factor under high species diversity but ceases to be the most important factor when more than 13% of ecosystem services provisioning (salmon abundance) is lost, after which species richness becomes the most important factor. Only at extreme low levels of diversity (and when > 50% of ES are lost), species abundance becomes the most important factor to which value losses can be attributed.

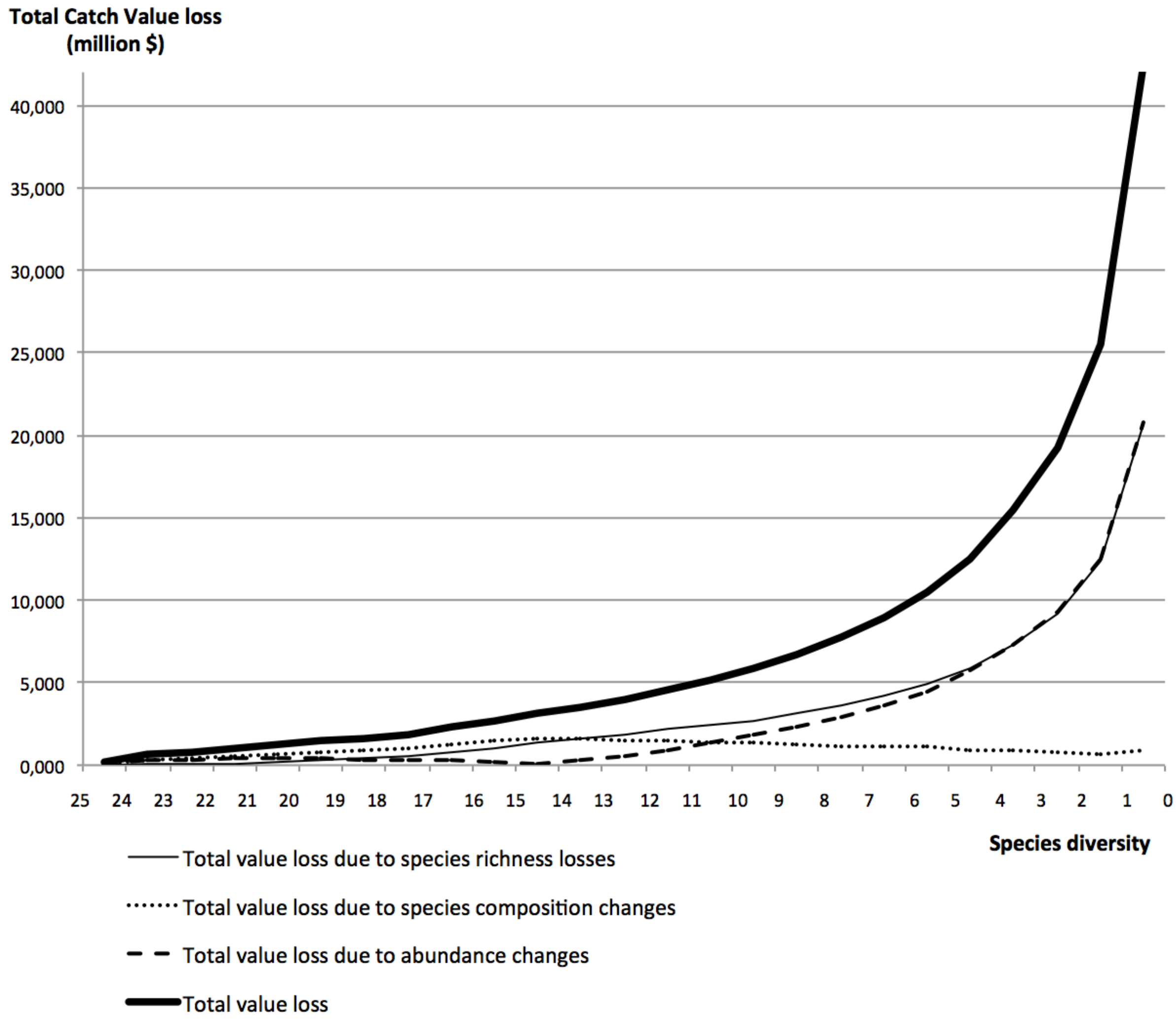

Figure 4.8: shows the total value losses of salmon catch in terms of gross revenues, as well as the total value losses due to changes in species richness, composition and abundance.

MODEL OUTPUTS				EXTRAPOLATION TO PACIFIC NORTHWEST RIVER SYSTEMS									
Species richness (1)	Salmon (2)	delta ES (3)	delta ES (%) (4)	Catch (kton/y) (5)	Total Catch Value (million $) (5,24$/kg) (6)	Total Catch Value Loss (million $) (7)	(%)	Value lost (million $) per species lost (8)	RICH-L % (9)	COMP-L % (10)	ABUN % (11)	Total value loss due to species richness losses (12) (million $)	Total value loss due to species composition changes (13) (million $)	Total value loss due to species abundance changes (14) (million $)
25	1004,61			8,18	42,84									
24	995,74	8,88	0,88	8,14	42,65	0,19	0,44	0,19	2,07	50,22	47,70	0,004	0,095	0,090
23	987,85	16,77	1,67	8,06	42,23	0,61	1,42	0,30	4,31	50,42	45,27	0,026	0,306	0,275
22	975,27	29,34	2,92	8,02	42,02	0,82	1,91	0,27	6,71	50,74	42,55	0,055	0,415	0,348
21	966,75	37,86	3,77	7,98	41,82	1,03	2,40	0,26	9,33	50,96	39,71	0,096	0,523	0,408
20	963,04	41,58	4,14	7,94	41,61	1,24	2,89	0,25	12,22	51,05	36,72	0,151	0,631	0,454
19	854,73	149,89	14,92	7,90	41,40	1,45	3,38	0,24	19,03	55,28	25,69	0,275	0,799	0,372
18	835,20	169,42	16,86	7,86	41,19	1,66	3,86	0,24	24,27	56,32	19,41	0,402	0,933	0,321
17	860,68	143,94	14,32	7,82	40,98	1,87	4,35	0,23	28,43	55,05	16,52	0,530	1,027	0,308
16	871,88	132,73	13,21	7,74	40,56	2,28	5,33	0,25	33,61	54,54	11,85	0,768	1,246	0,271
15	874,58	130,03	12,94	7,66	40,14	2,70	6,31	0,27	39,79	54,43	5,78	1,076	1,472	0,156
14	866,32	138,29	13,76	7,58	39,72	3,12	7,29	0,28	45,39	52,65	1,96	1,418	1,644	0,061
13	872,27	132,34	13,17	7,50	39,30	3,54	8,27	0,30	46,21	45,60	8,19	1,637	1,615	0,290
12	865,35	139,26	13,86	7,42	38,88	3,96	9,25	0,30	46,55	38,54	14,92	1,844	1,527	0,591
11	852,59	152,02	15,13	7,30	38,25	4,59	10,71	0,33	46,73	33,19	20,08	2,145	1,523	0,922
10	804,20	200,42	19,94	7,18	37,62	5,22	12,18	0,35	46,17	26,87	26,96	2,410	1,402	1,407
9	793,01	211,60	21,06	7,06	36,99	5,85	13,65	0,37	46,51	22,59	30,89	2,720	1,321	1,807

8	785,15	219,47	21,84	6,90	36,16	6,69	15,61	0,39	46,92	18,29	34,79	3,137	1,223	2,326
7	777,15	227,47	22,64	6,70	35,11	7,73	18,05	0,43	47,31	15,04	37,65	3,659	1,163	2,912
6	732,42	272,20	27,09	6,46	33,85	8,99	20,99	0,47	47,23	12,43	40,34	4,247	1,118	3,627
5	686,65	317,96	31,64	6,18	32,38	10,46	24,41	0,55	47,27	10,43	42,31	4,944	1,090	4,425
4	586,52	418,09	41,60	5,78	30,29	12,56	29,31	0,63	46,83	7,54	45,63	5,879	0,947	5,729
3	536,67	467,94	46,56	5,22	27,35	15,49	36,15	0,74	47,20	5,81	46,98	7,312	0,900	7,278
2	488,55	516,06	51,35	4,50	23,58	19,26	44,96	0,88	47,82	3,96	48,22	9,211	0,763	9,288
1	347,53	657,08	65,39	3,30	17,29	25,55	59,64	1,11	48,30	2,69	49,01	12,340	0,688	12,522
0	10,05	994,57	98,97	0,00	0,00	42,84	100,00	1,79	48,78	2,20	49,02	20,898	0,941	21,001

Table 4.2: overview of the results. The underlined numbers in column 12, 13, 14 indicate which aspect of diversity (richness, composition or abundance) contributes most to value losses encountered.

4.11 The indirect use value of aquatic invertebrates (step 9)

The cost of losing species increases with decreasing species diversity. The marginal value of species (table 3 column 8) is defined as the total catch value loss divided by the total number of species lost. The marginal value varies from 0,19 million $ per species lost at high species diversity, to 1,79 million $ at low species diversity. For example, in the case when only 10 species out of 25 remain, the industry will likely encounter an average annual gross value loss of 5,22 million $ representing a loss of 0,35 million $ per species lost. Under high species richness ($20<s<25$), the loss of a single aquatic invertebrate species represents an average gross revenue loss of 0,19 – 0,25 million $. When species diversity is lower ($19<s<5$), average gross annual loss are higher amounting to 0,24 – 0,55 million $ per species lost, and increase to 1,79 for the loss of all species, irrespective of the identity of the species lost (see figure 12). Separating the effect of species richness from composition and abundance changes (see section 4.6) also indicated that the importance of species richness increased with declining species diversity. The richness effect for the loss of the first species accounted for 2,07% of the value lost and increased gradually to 48,78% for the loss of the last species (see table 3). Hence, the effects of species richness represent a gross revenues loss of 0,004 million $ for the loss of the first species and increases under high species diversity to 21 million $ for the loss of a single species under low species diversity (table 3.2 column 12).

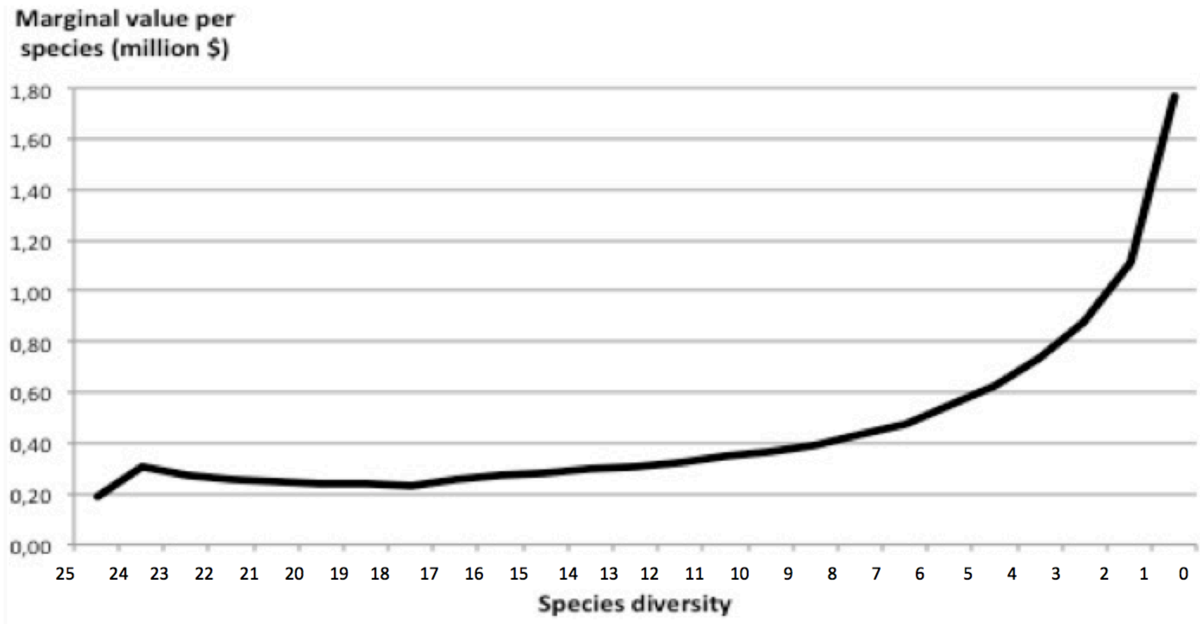

Figure 4.9: shows the cost of losing species (marginal value per species) in function of the level of species richness encountered, irrespective of the identity of the species.

4.12 Discussion

The results presented here give an indication of the order of magnitude of the economic losses for the commercial salmon fishing industry when macro-invertebrate diversity is lost. No external costs of the effects of macro-invertebrate losses in other parts of the ecosystem were included in the analysis. Also, this analysis only focussed on gross revenue losses for the commercial fishing industry, while at the same time, salmon also has a value for the recreational fisherman, tourism and has spiritual values. These values are also likely to be affected by a change in macro-invertebrate diversity.

The decrease in species diversity results in non-linear losses with relatively higher losses under lower species diversity, indicating the importance of high species diversity. This also suggests that ecosystems with higher macroinvertebrate diversity may be more resilient to environmental alternations that result in species extirpations, versus those with already low diversity.

In their analysis, Winfree et al. (2015) state that it is species abundance of common species that drives ecosystem service delivery whereas richness changes are relatively unimportant because they primarily involve rare species that contribute little to function. In our analysis, this statement can partly be supported, but only at extreme low levels of diversity (2 species or less out of 25 remain). It might be that the V-shaped curve of the abundance effect represented in figure 8 is related to the compositional response driven by the presence of certain dominant species early on. The removal of important dominant species early on influences the abundance of other species because they are released from strong competition. However, as more species are removed randomly, the impact of removing a dominant competitor decreases (because there are less species to release from their competitive effects). Eventually, this switches as the removal of the few species that remain are more likely to be important to maintaining salmon spawners.

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ANNEX 1

Creating Invertebrate "Pseudo-species"

We 'created' 25 different aquatic invertebrate species for the model analysis by randomly selecting the values of 11 parameters in the Aquatic Trophic Productivity (ATP) model that control invertebrate physiology and population dynamics (Table 1). These "pseudo-species" were created via Latin hypercube sampling (LHS). In LHS, the specified range for each parameter is divided into *N* strata of equal width (where *N* = 10,000), and a random parameter value is selected within each strata. From the *10,000* possibilities for each parameter, LHS randomly selected one of these values (without replacement). We did this 10,000 times, to create 10,000 randomly selected parameter combination that represent 10,000 'potential' aquatic invertebrate species. These parameter combinations were then simulated in the ATP model to create modeled biomass dynamics for 10,000 aquatic invertebrate species. However, many of these parameter combinations produced species that were unrealistic. Many parameter combinations, for example, produced invertebrate biomasses that quickly crashed (or approached zero), or were unrealistically high. To account for this, we removed those species those maximum modeled biomass for the year was (after reaching equilibrium) <0.02 and >1.9. Removing these species left 1,281 species that we deemed to be "realistic"; i.e., produced invertebrate biomasses that are similar to those reported in the literature (Water 1977; Huryn and Wallace 2000; Bellmore et al. 2013). From those 1,281 species we randomly selected 25 to include in our analysis.

Para-meter	Para-meter Description	Units	Value Range	Sources
$cons_{max,I}$	maximum rate of consumption when	g g^{-1} day^{-1}	0.05–0.8	(D'Angelo et al., 1997; Grafius and Anderson, 1979; McIntire, 1996;

Para-meter	Para-meter Description	Units	Value Range	Sources
	temperature is optimum			Rutherford et al., 2000)
$Temp_{opt,I}$	optimum temperature for consumption	°C	5-25	(McIntire, 1996; Rutherford et al., 2000)
γ_I	dimensionless self-interaction parameter	unitless	1-10	(Bellmore et al., 2017)
k_I	prey biomass half saturation level	g AFDM m^{-2}	1-15	(Bellmore et al., 2017)
$r_{ref,I}$	rate of respiration at 20°C	g g^{-1} day^{-1}	0.01–0.1	(D'Angelo et al., 1997; McIntire, 1996; Rutherford et al., 2000)
m_I	daily mortality rate	g g^{-1} day^{-1}	0.005–0.07	(Bellmore et al., 2017)
a_I	shape parameter for export rate equation	unitless	2-15	(Bellmore et al., 2017)
B_I^*	refuge biomass that is invulnerable to predation	g AFDM m^{-2}	0-1	Assumed
$Pref_{carcass}$	preference of aquatic invertebrates to consume salmon carcass material	unitless	0-1	Assumed
$Pref_{periphyton}$	preference of aquatic invertebrates to consume	unitless	0-1	Assumed

Para-meter	Para-meter Description	Units	Value Range	Sources
	perphyton			
$Pref_{detritus}$	preference of aquatic invertebrates to consume terrestrial detritus	unitless	0-1	Assumed

Table Appendix 1. Parameters used to code aquatic invertebrates in the ATP model, including: a description of each parameter, parameter units, the range of values used in the Latin hypercube analysis to create alternative species, and literature source(s).

CHAPTER 5

Discussion

Chapter 5: Summary and discussion

5.1 Summary

In this dissertation, **a stepwise methodological framework for the valuation of biodiversity is introduced, based on the ecological role of species in the ecosystem**. The framework is designed to quantify the indirect use value for biodiversity by integrating a production function approach with a market price-based approach.

Chapter 1 identified the importance of biodiversity for the well functioning and resilience of ecosystems and framed this analysis within the global policy context. While the necessity for increased knowledge of the economic consequences of biodiversity losses is obvious, the development of a framework for the valuation of biodiversity and the inclusion of its components, placing at the center the ecological function of species in the ecosystem, encounters **four key challenges**: (i) The plurality and multiplicity of valuation languages as well as the ambiguity on the definition of biodiversity and the object of valuation weakens the credibility of the use of economic values of non-marketed goods for decision-making purposes, (ii) no established framework has been agreed upon that effectively assesses biodiversity losses for their effects on economic performances, (iii) ecological uncertainty and ambiguity exist on the relationship between species diversity and ecosystem services and (iv) biodiversity is a multi-dimensional concept and requires multiple proxies for quantifying it.

Chapter 2 introduced **a generic methodological framework that quantifies the indirect use value of changes in functional diversity**. It quantified the effects of changes in non–marketable species diversity for their impact on economic activities through the delivery of ecosystem services and attached an indirect use value to species diversity. It integrates (i) a dynamic ecological model simulating interactions between species with (ii) an economic model assessing the effect of changes in species diversity for net revenues. The model both (i) quantifies the contribution of species diversity to net revenues through the use of a production function technique, and (ii) attributes a monetary value to species diversity by employing a direct market based technique based on the changes in the provisioning of a marketable good.

The introduction of such a framework contributes to closing the research gaps for biodiversity valuation by (i) decreasing the reliance on public perception and knowledge of what biodiversity is worth to them, (ii) decreasing the reliance on stated valuation techniques for complex concepts such as biodiversity, (iii) providing a strong link between economic theory and ecological research, (iv) exploring and refocusing economic valuation of biodiversity towards production based methods, and (v) stressing the functionality of biodiversity and placing the ecological role of species at the center of biodiversity valuation studies.

Moreover, on a national policy level, the methodological framework could contribute to achieving the Aichi Biodiversity Targets by improving "…*knowledge, the science base and technologies relating to biodiversity, its values and functioning".*

Chapter 3 then set off to apply the methodological framework, quantifying the indirect use value of biodiversity by integrating production functions with a market-based approach. Through the elaboration of a case study titled "the economic valuation of natural predators for biological pest control in pear production in Flanders (BE)", the first sub question "What is the **indirect use value of natural predators** for **biological pest control** in **pear production** in Flanders?" was analyzed**.** The methodology resulting from the case focused on the ecological role of a **limited number of species** and effectively integrated an ecological-economic model to derive the indirect use value of changes in biodiversity.

Results indicated that the loss of three predators could decrease net farm income with 88.86 €ha^{-1} to 2186.5 €ha^{-1}. For the pear production sector in Flanders in 2011, this constitutes to an indirect use value of 0.68 million € for one predator and 16.63 million € for the presence of three predators. Considering that the gross revenues for the sector totaled on average 163 million euros for the period 2009-2013, the contribution of the predators accounts for 0,41% to 10.2% of the sectors' gross revenues. Differences between the 25% best performing farms and 25% of worst performing farms can account for 10.000€ per hectare.

These results supported the hypotheses that (i) a decrease in natural predators causes a significant decrease in the provisioning of the ecosystem service biological pest control (ii) a reduction in natural predators considerably reduces the quality of marketable agricultural production and (iii) the occurrence of lower quality yields due to reductions in species diversity considerably decreased net farm income.

It was the aim of this chapter to highlight the importance of healthy agro-ecosystems, not only for the purpose of food production but also for its contribution to farmer's income. It was emphasized here that effective valuation of biodiversity can include both intrinsic as well as economic arguments but that, in order to take into account the effect of biodiversity losses in economic arguments, it was imperative that the ecological function is taken into account.

<u>Chapter 4</u> assessed whether the methodology employed in chapter 3 could be also be used in different circumstances: (i) a **large number of species**, (ii) another type of ecosystem (**freshwater river systems** instead of an agricultural production system), and (iii) another ecosystem service (**salmon production** instead of biological pest control). Furthermore, chapter 3 expanded the methodology by **accounting for the contribution of the individual effects of changes in species richness, species composition and species abundance** to determine the indirect use value of biodiversity. Therefore the subquestion addressed in chapter 4 was: "What is the **indirect use value of aquatic macro-invertebrates** for salmon production in the US North West?".

The results reveal that the cost of losing species increases with decreasing species *diversity*. The indirect use value varies from 0.19 million $ per species lost at high species diversity to 1,79 million $ per species lost at low species diversity. Separating the effect of species richness from composition and abundance changes also indicated that the importance of species *richness* increased with declining species diversity. The richness effect accounted for 2,07% for the loss of the first species and increased gradually to 48,78% for the loss of the last species. The results give an indication about the order of magnitude of the economic losses for the commercial salmon fishing industry when macro-invertebrate diversity is lost. The decrease in species diversity resulted in non-linear losses with relatively higher losses under lower species diversity, demonstrating the importance of high species diversity. This also suggests that ecosystems with higher macro-invertebrate diversity may be more resilient to environmental fluctuations that result in species extirpations, versus those with already low diversity.

The aim of this analysis was to provide a justification for the argument for biodiversity conservation, based on the ecological function of species, through the delivery of comparable monetary standards. These values could be supplied to inform policy makers about the importance of including biodiversity effects and providing a justification for the opportunity costs encountered. Also, it could be used as a financial risk analysis tool, informing the private sector of the effects of changes in the supply of natural resources on business operations and supply chain management.

5.2 Market based valuation techniques

First, integrating a market-based approach with a production function approach relies on actions that occur in the market and makes use of market prices for products or services that rely on renewable natural resources as inputs into a production process. Market-based methods usually focus on private costs and benefits, thereby neglecting the social costs and benefits of changes at the ecosystem level. Future research could examine how the social costs and benefits can be included in the framework to result in a more holistic value for biodiversity.

Second, agreeing with Hamilton (2013), in that the aggregation of different values derived from biodiversity may give rise to issues of double counting, the marginal values derived here are not to be used in cost benefit analysis or national accounting, since the marginal values are already capitalized in the marketable goods from which they were estimated. The marginal value estimates derived here can provide information to be included in financial risk analysis, when private companies are dealing with uncertainty over natural resources and the provision of marketable goods depending on functional diversity.

Third, the question can be raised if all functional groups can be valued similarly; or in other words, if for all functional groups ultimately a marketable good can be identified to which the functional group indirectly contributes. Key to answering this question is twofold: (1) a fundamental ecological understanding of trophic cascades and (2) the flexibility of the methodological approach in valuing not a marketable product as an endpoint, but rather the ecosystem service provided.

As an example of the fundamental understanding of trophic cascades, the Yellowstone Wolf Project Annual Report (2016) identifies the unexpected impact of the reintroduction of wolves in Yellowstone National Park in 1995. As was to be expected, wolves are at the top of the trophic cascade, preying on deer and other animals. One of the most interesting findings was that besides taking lives, they also gave life and ultimately changed the course of rivers in the park. This was due to behavioral changes observed in the deer population to avoid the wolves, allowing riverbanks to develop from grasslands to woodlands, thereby stabilizing the banks, reducing erosion and changing the course of rivers. With the development of woodlands, birds returned, followed by birds of prey and with the changes observed in the course of rivers, beavers returned creating niches and habitats for fish and reptiles. In spite the fact that wolves do not contribute - directly or indirectly - to the production of a marketable good, wolves are considered important actors in the ecological functioning of the ecosystem, thereby providing services such as reduced erosion which can be valued as an ecosystem service. Hence, a fundamental ecological understanding of the impact of the species or functional group of the provisioning of services is essential for determining values, as well as the possibility to ultimately value the ecosystem services provided and not a marketable good produced.

The flexibility of the framework in valuing not a marketable product as an endpoint, but rather the ecosystem service provided is in line with the current discourse in ecological economics whereby monetary values are placed on ecosystem services delivered to humans. It follows the same reasoning as when a marketable good is employed as the endpoint in that the use of a dynamic ecological model serves to quantify the contribution of changes in biodiversity for the monetary valuation of the service identified. When a marketable good is identified the change in functional diversity is related to the change in income, when valuing services, the change in functional diversity are related to changes in the provision of services and the change in the values of the services provided.

Also, it is a common misconception that placing an economic value on biodiversity should replace its intrinsic value. It is argued here that placing an economic value on biodiversity does by no means undermine or replace the intrinsic value that biodiversity has. Moreover, the importance in monetary valuation arises from the fact that when we do not place a monetary value on biodiversity, it cannot be incorporated into policy decision-making outweighing the costs and benefits of conservation. Also, it may spur economic incentives when it turns out that action in favor of nature conservation is more cost-effective than inaction.

5.3 Dynamic ecological model development

It can be argued that the methodological framework, which integrates a production function approach with a market-based approach, results in objective measurements of biophysical parameters as the basis for biodiversity valuation. It is the belief that the well functioning of ecosystems is of primary importance and should be based on sound and integrated ecological and economic reasoning. This does not need to contradict the fact that the general public has a clear perception of what biodiversity is worth to them. It does however mean that, when devising management plans, the well functioning of ecosystems is of primary importance and cannot be guided by public perception of importance or value of species but should be guided by ecological functioning to secure the provisioning of ecosystem services. With the proposed framework, we hope to facilitate and encourage further research on the effect of changes in biodiversity for the economy and human well-being that effectively take into account the importance of species diversity for ecological function, with the ultimate aim of assessing the effects of ecosystem management for the well functioning of ecosystems and, ultimately, for human well being.

Building a *dynamic* ecological model proved to be fundamental to the methodological framework introduced here. A dynamic ecological model allows for (i) continuous spatial and intertemporal variations, (ii) interactions between species, (iii) the effects of these interactions and variations on the ecosystem functions, services and values, (iv) comparison of realistic alternative scenarios of species richness, composition and abundance, and therefore (v) a valuation of *all* species in the functional group.

Empirical measurements that do not additionally model the dynamics of species interactions for their effects on functioning, and only rely on field experiments that relate measurements of diversity with ecological functioning at specific time intervals or simultaneously on different geographic locations at simultaneous locations, are only able to value the changes in diversity observed for the different measurements. Therefore they are not able to assess the effect of consequences of other diversity compositions for ecological functioning. A dynamic model is capable of doing so, and therefore possesses the strength to value the marginal changes in diversity, as opposed to empirical models.

On the downside, building dynamic ecological models requires in-depth knowledge on the functioning of the ecosystem, the relationships between the actors and the availability of data supporting it. This may lead to difficulties in the construction of a dynamic model simulating real-world conditions. In many instances, data on species richness, abundance and composition may be available, however the parameters needed to quantify the relationship between these actors may not yet be readily available. Potentially, this leads to high uncertainty in the quantification of the ecological function and services stemming from species interaction simulations. Furthermore, as is the case in chapter 2, data on the ecological economic linking function was not available at the time of writing. Also here, assumptions were made, leading to increased uncertainty of the results. Using a sensitivity analysis, not only to incorporate variability of economic parameters, but also including variability of ecological parameters, may provide insight in the variability of results and the degree of uncertainty involved.

Whilst building a dynamic ecological model may seem a daunting task, recent software development has put a lot of effort in devising user-friendly interfaces that do not require the learning of code but allows for building visually attractive stock and flow diagrams. Still however, this may question the practical applicability of the methodological framework. In order to improve the practical applicability, it may be possible to build a generic model. It would then need to be explored whether an interface can provide a practical solution for personalizing existing generic models to users requirements and therefore considerably reduce the effort in building new models. Interface building is currently provided by iThink Stella software, in such a way that the model itself does not need to be built but in which the user indicates the number of species, their characteristics and the interrelationships.

Also, the methodological framework itself requires information on the links between ecological function and the provisioning of ecosystem services, as well the benefits experienced. As for our first case of natural predators for biological pest control, the link between the pest insect density and the percentage of black pears occurring as a consequence was not available at the time of writing. Therefore, multiple potential relationships were examined, resulting in confidence intervals and increased uncertainty of the results. As a result, PCFruit (Proefcentrum Fruitteelt, Belgium) has now set up trial designs in order to investigate the relationships between the timing of occurrence of the pest insect, their density and the number of black pears encountered.

5.4 Extension of the framework to evaluate management practices

The methodological framework could be extended to include the effect of management practices. Now, the different steps do not attribute a cause for the potential changes in biodiversity. It starts to explore the consequences of potential losses of biodiversity. However, the methodological framework can be extended to include the valuation of management practices by examination of their effects on biodiversity and hence for their effects on the marketed good or ecosystem services delivered. The introduction of a step 0 could read "determining the effects of ecosystem management for functional diversity". In doing so, the model might be used to evaluate management practices and to answer questions such as "how might the system respond to planned restoration actions?" or, "where and what type of restoration actions are the most effective?".

The concept of introducing management practices in the methodological framework is exemplified by the Aquatic Trophic Productivity Model (see chapter 3), consisting of (i) biomass stocks, (ii) consumer-resource interactions, (iii) inputs of energy, nutrients and organic matter and (iv) linkages to in-stream physical habitat conditions and riparian vegetation conditions. These linkages can effectively be used to explore the effect of management practices for their effects on aquatic macro-invertebrate species dynamics and the resulting indirect use values. Several management options were already included in the model such as: (i) riparian restoration, (ii) habitat restoration by reconnecting side channels and (iii) the addition of salmon carcasses. In chapter 3, the effect of potential management actions was not taken into account and the analysis set off by considering the consequences of changes in biodiversity. By introducing the effects that management practices have for species diversity, the framework could effectively be used to organize the understanding of these systems, and guide restoration and monitoring in the context of an adaptive management framework.

5.5 The commodification of biodiversity

In spite the need for a monetary valuation of biodiversity that outweighs the costs and benefits of conservation, many have criticized the commodification of biodiversity. Commodification is defined as the transformation of nature into objects of trade. Many critics point out that environmental degradation stems from the same processes of commodification and point out three broad problem: (i) practical issues relating to the feasibility of turning nature into a commodity, (ii) moral issues questioning the ethical implications of commodification and (iii) issues relating to the consequences of this commodification on nature itself.

As Foster points out in *Ecology Against Capitalism*, the environment is not a commodity, but it is the biosphere that sustains all life as we know it. He wrote: " *Economic growth theorist Robert Solow wrote in the American Economic Review in May 1974, that, "if it is very easy to substitute other factors for natural resources, then there is in principle no 'problem.' The world can, in effect, get along without natural resources, so exhaustion is just an event, not a catastrophe." Solow, who later received the Nobel Prize in economics, was speaking hypothetically and did not actually go so far as to say that near–perfect* *substitutability was a reality or that natural resources were fully dispensable.*"

Since then, many recognize the finite nature of resources and the biophysical limits of Earth as a crowded and finite space, with limited resources for extraction and a narrowing capacity for waste disposal and pollution (Daily et al., 2000; Ehrlich and Harte, 2015). There can be no doubt that our ecological systems that provide the services on which we and our economics depend, are in distress. Therefore, the ultimate aim of this analysis is to promote the understanding of the importance that biodiversity has for the well functioning of ecosystems, its contribution to our economy and human health. By introducing a methodological framework that allows for the monetization of the function that biodiversity has for our well-being, it is our hope that the importance of biodiversity becomes more visible and can be included in cost-benefit analyses. We hereby hope to support a fundamental change in thinking about economic optimization, based on a sustainable and efficient management of ecosystems.

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