International cooperation for biodiversity conservation: an economic analysis

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A mi familia

Table of Contents

Chapter 1

General Introduction

1. Overview

Biodiversity decline poses significant threats to current and future generations. According to the Planetary Boundaries framework, the boundary associated to loss of biodiversity has already been overstepped (Rockström et al. 2009). Current biodiversity loss is mainly an anthropogenic effect. Extinction of species has been a natural process since the formation of the Earth (Rockström et al. 2009, Barnosky et al. 2011). Yet, the recent rate of extinction of species is estimated to be from 100 to 1000 times more than what is considered normal when compared to fossil records (MA 2005, Rockström et al. 2009). Almost all of the Earth's ecosystems have been dramatically transformed and some of them are being pushed towards critical thresholds that could risk overall livelihoods and wellbeing of the human population (MA 2005, Pereira et al. 2012, CBD 2014).

Implications of severe biodiversity loss include irreversible alterations of ecosystem services, vulnerability to natural disasters, human health risks, threats to food and energy security, depletion of natural resources and damage to social relations (Chapin III et al. 2000, MA 2005, UNEP 2010). Moreover, consequences of biodiversity loss have a stronger effect on the most vulnerable populations such as subsistence farmers, women, and indigenous and local communities (MA 2005, Timmer and Juma 2005, Díaz et al. 2006, CBD 2014).

There is an urgent need to study and develop efficient conservation instruments such that decision makers can implement them to halt the ongoing rate of biodiversity loss. However, this is not a simple task given the multidimensional nature of biodiversity (levels of biological organisation) and the diverse geographical scales of concern (from local to global).

This thesis examines the functioning and effectiveness of different economic instruments for biodiversity conservation at diverse scales. In Chapter 2 of this thesis, market theory and contract theory are applied to assess, first, the economic conditions under which markets for biodiversity are expected to function, and second, the potential to scale up local or national payment mechanisms to a global level. The other chapters of the thesis present game theoretical analyses on the modelling and functioning of International Environmental Agreements (IEAs) for biodiversity conservation. Game theoretical analyses provide a novel opportunity to study in detail the impact of key features of biodiversity conservation on the effectiveness and stability of conservation agreements. This type of analysis is then applied to a case study of habitat conservation for a migratory bird species.

Section 1.1 of this chapter describes the problem of biodiversity loss in more detail and Section 1.2 presents some of the current responses to this problem. In Section 1.3

I describe existing economic instruments to address biodiversity management and conservation. Section 1.4 covers the objectives of the thesis, and Section 1.5 introduces the methodological approach used to meet these objectives. Finally, the outline of the thesis is presented in Section 1.6.

1.1 Biodiversity loss

Biodiversity is a multidimensional concept. The Convention on Biological Diversity (CBD) defines it as:

> *'The variability among living organisms from all sources including, inter alia, terrestrial, marine, and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems.'* (UN 1992, p.3)

This definition encompasses several levels of organisations of biological variation or diversity: genetic, species and ecosystems (Gaston 2000, MA 2005). Most analyses of spatial variation employ a biodiversity concept measured by the number of species observed or estimated to occur in an area (Gaston 2000), i.e. species richness or abundance (e.g. Weikard et al. 2006a). However, the term biodiversity also includes important biological considerations such as the genetic makeup of populations and endemism (UNEP-WCMC 2014). Moreover, in recent years the relationship between biodiversity and ecosystem services has gained attention; see Costanza et al. (2007), de Groot et al. (2010), Bullock et al. (2011), Salles (2011), Mace et al. (2012), Bastian (2013), and Balvanera et al. (2014). No single measurement can capture all dimensions of biodiversity (Pereira et al. 2012), which makes quantification of biodiversity a complex task.

The world is experiencing the fastest rate of species extinction known in geological history (UNEP 2010, Pimm et al. 2014), up to the point where it is believed that the planet has entered its sixth mass extinction (Ceballos et al. 2015). The global Living Planet Index (LPI) reveals a continuous decline of vertebrate populations over the past 40 years. While the global LPI showed a decline of 52% in overall vertebrate species populations between 1970 and 2010, both the terrestrial and the marine index fell by 39%, and the freshwater index fell by 76% over the same period (WWF 2014). According to the 2004 IUCN Red List, around 12% of bird species, 23% of mammal species, 32% of amphibian species and 34% of all gymnosperms are threatened with extinction (Baillie et al. 2004). Current trends indicate that i) the rate of biodiversity loss does not appear to be slowing down (MA 2005, Butchart et al. 2010, and WWF 2014) and that ii) pressures on biodiversity will continue to increase (CBD 2014).

Although there are natural drivers that trigger biodiversity loss, most of them are human-induced. Among the increasing pressures that biodiversity is currently facing, the Global Biodiversity Outlook 4 reports i) the loss and degradation of natural habitats, ii) the overexploitation of biological resources, iii) pollution (in particular the build-up of nutrients in the environment e.g. nitrogen and phosphorus), iv) impacts of invasive alien species on ecosystems and their services, and v) climate change and acidification of the oceans (CBD 2014).

1.2 Responses to biodiversity loss

To address the current rate of biodiversity loss, the United Nations established the Convention of Biological Diversity (CBD) that was open for signature in 1992 and entered into force one year later. The CBD is *'an international legally binding United Nations treaty to deliver national strategies for the conservation and sustainable use of biodiversity'* (UNEP-WCMC 2014). It has three main objectives: the conservation of biological diversity, the sustainable use of the components of biological diversity, and the fair and equitable sharing of the benefits arising out of the utilisation of genetic resources (UNEP-WCMC 2014). As of today, it has been ratified by 196 parties (CBD 2015).

In 2002, ten years after the CBD was established, the parties to the Convention developed a Strategic Plan to achieve a significant reduction of the current rate of biodiversity loss at different scales by 2010. This objective was also incorporated as a new target under the United Nations Millennium Development Goals (UN 2015). Still, pressures on the natural world increased and the international community failed to address the underlying drivers of biodiversity loss; hence, the target was not met by 2010 (Butchart et al. 2010, CBD 2010, Mace et al. 2010, Adenle 2012). Consequently, signatory countries adopted a new 'Strategic Plan for Biodiversity 2011-2020' in 2010 in Nagoya, Japan. This plan includes twenty global targets, better known as the Aichi Biodiversity Targets (CBD 2015a). The goals and targets of the new plan were set to be accomplished at the global level, but under a flexible framework that also establishes targets at a national and regional scale. As a way to support and contribute to the implementation of the new plan, the United Nations declared the years 2011-2020 as the United Nations Decade on Biodiversity.

Initiatives for biodiversity conservation vary in terms of their objectives, scale, and level of outreach. There are local conservation programs with a direct participatory approach, such as the voluntary conservation program for the protection of marine sea turtles in the Caribbean coast of Costa Rica (EPI 2015) and the volunteer program on lemur monitoring and reforestation in Kianjavato, Madagascar (MBP 2015). There are also initiatives at a global scale, with a technical, scientific approach. For instance, The

Economics of Ecosystems and Biodiversity (TEEB) initiative was created with the aim *'to mainstream the values of biodiversity and ecosystem services into decision-making at all levels'* (TEEB 2015). Also, in 2012, the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) was established to synthesize, review and evaluate relevant information and knowledge on the state of biodiversity and ecosystems coming from both the scientific and policy actors (IPBES 2015). Finally, the most recent global initiative is the United Nations 2030 Agenda for Sustainable Development, which seeks to build on the Millennium Development Goals and complete what these did not achieve. The agenda includes the halt of biodiversity loss as a part of one of its 17 main goals (UN 2015a). Specifically, it states that urgent action must be taken *'to reduce the degradation of natural habitats, halt the loss of biodiversity and, by 2020, protect and prevent the extinction of threatened species'* and it also aims *'to integrate ecosystem and biodiversity values into local planning, development processes, poverty reduction strategies and accounts'* (UN 2015a, p.21).

Targeting the problem of biodiversity loss requires interdisciplinary, integrative actions that should be established in joint collaboration between different sectors such as governments, academia, scientific organisations, and NGOs. The field of economics plays a key role in the identification of action plans that need to be implemented to address the multiple causes of biodiversity loss. Economic instruments *'can generate financial resources (…), create incentives for investment, and increase the involvement of private agents in environmental protection'* (UNEP 2004, p.23). The next section covers the existing economic approaches used to deal with biodiversity conservation.

1.3 Economic instruments for biodiversity conservation

Economic tools have been implemented to halt biodiversity loss and to effectively manage biodiversity conservation at different scales. The determination of economic instruments for biodiversity conservation is directly linked to the type of goods and services biodiversity provides (public vs. private) and the scale at which they are considered (local vs. global).

If we consider global biodiversity as the set of all genes, species and ecosystems in the world, we are dealing with the nature of biodiversity as a public good. As type of good, biodiversity is non-exclusive and non-rival. However, there are types of biodiversity services that are best described by having a semi-private nature. For instance, ecotourism can be an example of biodiversity as a non-rival good that can be enjoyed exclusively by those that have access to it (i.e. club good). Finally, individual components of an ecosystem are often considered as private goods (Perrings and Gadgil 2003). Some examples of private biological resources are timber, fish, and bioprospecting activities.

Different general approaches address the notion of capturing biodiversity values in a more generalised context. National accounts as indicators of nature-derived welfare have been a topic of concern, mainly because they are known to measure goods produced from natural resources, but they do not measure the 'bads' (Stiglitz et al. 2009). This type of measure of economic performance has failed to deliver an adequate sustainability appraisal. Consequently, several approaches have emerged that deal with natural capital accounting, e.g. the Natural Capital Approach (Voora and Venema 2008), the System of Environmental-Economic Accounting (SEEA) Experimental Ecosystem Accounting (UNSD 2015), and the Wealth Accounting and the Valuation of Ecosystem Services (WAVES) (WAVES 2015).

Despite recent progress in recognising interconnections between social and ecological systems and nature's contribution to human wellbeing (Selomane et al. 2015), the implementation of these interconnections into decision-making processes is still insufficient. Guerry et al. (2015) identified three elements related to ecosystem service information that need to be addressed to achieve the UN Sustainable Development Goals: *'(i) developing solid evidence linking decisions to impacts on natural capital and ecosystem services, and then to human well-being; (ii) working closely with leaders in government, business, and civil society to develop the knowledge, tools, and practices necessary to integrate natural capital and ecosystem services into everyday decision-making; and (iii) reforming institutions to change policy and practices to better align private short-term goals with societal long-term goals.'* (Guerry et al. 2015, p.7348). Tackling these issues would assist in the development of an inclusive wealth metric (Polasky 2015).

There is no global, harmonised observation system set to measure and deliver standardised information of biodiversity (Pereira et al. 2013, Alvarado-Quesada et al. 2014, UNEP-WCMC 2014). Moreover, just as there is no single measurement for biodiversity, there is no unique economic instrument to deal with biodiversity management. Some economic instruments are more appropriate to deal with the public nature of biodiversity, whereas others are more suitable for its private nature.

In the rest of this section I present some practical methods used to manage biodiversity conservation. I first introduce the two specific instruments that are addressed in further detail in this thesis, namely market-based mechanisms and IEAs. Then I refer to other existing economic instruments.

1.3.1 Market-based mechanisms

Market-based mechanisms are instruments used to replicate the functioning of a market for conservation purposes. They assist in the allocation of resources and the provision of economic incentives to preserve biodiversity (UNEP 2004). Market-based mechanisms arise as an alternative to address market failures originated from the public-good nature of biodiversity, either i) by incorporating the external cost of production or consumption activities by fees or charges on processes and products, or ii) by establishing property rights and facilitating the creation of a proxy market for environmental services (EEA 2005, Chobotová 2013).

This broad category covers a highly heterogeneous group of instruments with different links to markets as defined by economic theory (Broughton and Pirard 2011). For the purpose of this thesis I focus on those mechanisms in which a biodiversity credit represents the unit of preserved biodiversity that is traded in a market (examples of other market instruments are found in Section 1.3.3). The selection of market-based mechanisms that I analyse in Chapter 2 includes a combination of biodiversity offsets and conservation banking schemes, namely BioBank, BushBroker, Conservation Banking, Malua BioBank, and Wetland and Stream Mitigation Banking. Biodiversity offsets are tools that *'seek to compensate for residual environmental impacts of planned developments after appropriate steps have been taken to avoid, minimise or restore impacts on site'* (McKenney and Kiesecker 2010, p.165). As for conservation banks, they are defined as *'a parcel of private property that is conserved and managed in perpetuity under a conservation easement for the benefit of rare species. The party that holds the easement is granted credits by a federal or state agency for the land's species and habitat value. A bank owner may use or sell the credits within a predesignated service area to address mitigation required by state or federal law'* (Fox and Nino-Murcia 2004, p.997).

1.3.2 International Environmental Agreements (IEAs)

International agreements to address environmental problems are known to exist since the 19th Century (Mitchell 2003). IEAs arise to solve the common property resource dilemma (Wagner 2001), exemplified in cases such as the ozone protection, the acid rain problem and transboundary river pollution. If countries organise themselves in the management of their shared environmental resources, their overall collective wellbeing can increase (Barrett 1994). However, countries can adopt a strategic behaviour to benefit from the environmental improvement without contributing to its achievement; i.e. they perceive incentives to free ride. Furthermore, IEAs need to be self-enforcing, as countries cannot be forced to sign an agreement. From an economic perspective, game theory has been implemented since the 1990s as an approach to study the incentives and disincentives for players to participate in IEA (Wagner 2001).

1.3.3 Other economic instruments

Agri-environment schemes (AES)

AES are instruments to mitigate negative environmental effects caused by agricultural intensification (Ekroos et al. 2014). In Europe, AES are embedded in the Common Agricultural Policy (CAP) as a way to encourage farmers to protect and enhance the environment in their farms by paying them for the provision of environmental services and the adoption of environmentally-friendly farming techniques (European Commission 2015). The initial purpose of AES to protect threatened habitats or landscapes has shifted over time into an approach more focused on the prevention of species' loss and ecosystem maintenance (Batáry et al. 2015). The effectiveness of these schemes in conserving biodiversity has been questioned (Kleijn et al. 2001, Phalan et al. 2011). Different suggestions have been posed to achieve more effective conservation outcomes of AES, e.g. implementing targeted schemes (Kleijn and Sutherland 2003), using lower numbers of large resource patches as opposed to many fragmented patches (Whittingham 2007), and differentiating biodiversity conservation schemes from ecosystem services schemes (Ekroos et al. 2014).

Conservation auctions

Conservation auctions (or tenders) are an approach to fund conservation by allocating conservation contracts through a bidding process. Latacz-Lohmann and van der Hamsvoort (1997) were among the first to argue the implementation of auctioning conservation contracts in order to create a market structure for the management of public goods. In an auction scheme, landholders submit a bid to undertake conservation efforts on their property and define the cost of conducting such efforts. Consequently, bids are ranked according to best value for money. This mechanism has been widely adopted in Australia, where several tender schemes can be found (Doole et al. 2014). With this approach, governments can gain insight on farmers' cost of participation in the program. Furthermore this approach allows for conservation of biodiversity values at a lower cost than with other conservation alternatives. Auctions can be preferred to fixed-price programs in terms of economic performance (Stoneham et al. 2004, Schilizzi and Latacz-Lohmann 2007). Yet, additional aspects need to be considered when assessing overall conservation outcomes (Müller and Weikard 2002, Hanley et al. 2012, DePiper 2015).

Debt-for-nature swaps

Debt-for-nature swaps are a conservation approach that emerged in the 1980s as the consequence of extensive foreign debt and degraded natural resources in developing nations (Sheikh 2010). The objective of this approach is for an indebted developing country to undertake the use of local currency funds to finance a conservation programme, in exchange for the cancellation of a certain amount of their foreign debt (Hansen 1989, Potier 1991). It is estimated that since 1987, debt-for-nature swaps have generated around US$1 billion for conservation actions in developing countries (Sheikh 2010). One of the largest debt-for-conservation swaps took place in 2010 when the government of the United States, in collaboration with Conservation International and The Nature Conservancy, forgave US$26 million of Costa Rica's debt in exchange for Costa Rica to spend that amount of money on tropical forest conservation in a period of 16 years (Conservation International 2007, The Nature Conservancy 2010). Some pitfalls and limitations detected when applying this type of financial mechanism are swaps being too small to create indirect positive economic effects, a mismatch between the swap's alignment with national policy and national systems (Cassimon et al. 2011) and the disregard of livelihood needs of local people (Shandra et al. 2011).

Payments for environmental services (PES)

A payment for an environmental service (PES) is a voluntary transaction where a well-defined environmental service is being purchased by a service buyer (direct user or representing agent, e.g. government or NGO) from a service provider, if and only if the service provider secures its provision (Wunder 2005). PES are considered useful instruments to translate non-market values of the environment into financial incentives for local actors to provide environmental services. Their application is limited to environmental problems where ecosystems are mismanaged because *'many of their benefits are externalities from the perspective of ecosystem managers'* (Engel et al. 2008, p.663). Although the main objective of PES should be related to environmental outcomes, many programs consider poverty alleviation as either an additional objective (Wunder et al. 2008), or as an indirect side-effect (Pagiola et al. 2005, Wunder 2013). Some of the common concerns linked to implementation of PES are additionality, leakage, lack of permanence, the role of targeting, and the potential social inefficiency that could arise from adopting insufficient payments or inefficient land use (Engel et al. 2008). Other issues of concern include – as with other policy instruments – the dependence of outcomes on the interplay of local political forces, and the potential crowding out effect on the intrinsic motivations to carry out an activity given the type of payment (Muradian et al. 2013).

1.4 Research objectives

The objective of this research is to examine and develop economic instruments for biodiversity conservation at diverse scales. To achieve this objective, the following research questions are addressed:

Q1: What are the economic conditions under which market-based mechanisms for biodiversity conservation function at the local level and can they be scaled up to a transnational level?

The first research question of this thesis focuses on finding the economic characteristics that biodiversity markets should have in order to work. To meet this objective, I make use of market and contract theory to identify the required conditions to guarantee efficiency of biodiversity markets. In light of these conditions, I analyse the efficiency of five market-based instruments for biodiversity conservation that have been implemented in different countries. The chosen sample of mechanisms intends to represent differences location, operating times, scale of implementation, and type of markets. Furthermore, I assess the upscaling potential of the five selected schemes given their current performance.

Q2: What are the key features required to design an IEA for biodiversity conservation?

The second research question deals with an assessment of the characteristics that an IEA for biodiversity conservation should have. In light of the literature on stability of IEAs, I first identify three specific features of biodiversity that differentiate the biodiversity case from the emission abatement case, which is prominent in the literature. I then proceed to construct a game-theoretic model of biodiversity conservation that includes these three features in order to analyse coalition stability and the effectiveness of a biodiversity agreement. Finally, I investigate the effect of including a transfer scheme on overall coalition composition and stability.

Q3: What role does the inclusion of a spatial structure play in the stability of an IEA for biodiversity conservation?

The third research question considers the effects of the inclusion of a spatial structure on a self-enforcing IEA for biodiversity conservation. I first account for regional biodiversity benefits as positive spillovers from neighbouring countries. I introduce a general framework that considers the impact of both distance between and location of countries on coalition stability and efficiency. In order to meet this objective I introduce a definition of distance in terms of species dissimilarity. Furthermore, I construct a game-theoretic model for biodiversity conservation with a spatial structure derived from industry location models that best explains cooperation among neighbouring countries.

Q4: How can an IEA with a spatial structure be applied to habitat conservation of a migratory bird species?

The fourth research question deals with the application of the spatial structure addressed under research question Q3 to a case study on regional conservation of the non-breeding habitat of a migratory bird species. I first extend the game theoretic model mentioned under research question Q3 by including specific aspects of migratory behaviour. Furthermore, I calibrate the model parameters with empirical information from benefits and costs of habitat conservation per country. The case study is related to the conservation of the Golden-winged Warbler (*Vermivora chrysoptera*), a migratory bird species that spends its non-breeding season in specialised microhabitats in Central and the north of South America, and that is currently undergoing an accelerated decline of its population.

The research questions formulated above are addressed separately in the following chapters of the thesis. In the next section, I introduce the different research methodologies implemented to answer these questions.

1.5 Methodology

I use four main methodological approaches throughout this thesis: market theory, contract theory, game theory and industrial organisation theory. In Chapter 2, I address research question Q1 by using market and contract theory. As for research questions Q2-Q4, I implement game theory as the main methodology in Chapters 3, 4 and 5, respectively. Elements of industrial organisation theory are also applied to answer research questions Q3 and Q4.

1.5.1 Market theory

Market theory is used to evaluate the performance and efficiency of markets. In a market economy, transactions between agents are mediated by markets, and individual consumers exert an insignificant force upon them (Jehle and Reny 2011). Basic structural characteristics of a perfectly competitive market include clear and enforceable property rights, a sufficient number of buyers and sellers, information completeness, homogeneous products, zero transaction costs, and no barriers of entry and exit. In Chapter 2 of this thesis I make use of these criteria to evaluate and compare the functioning of a selection of market-based mechanisms for biodiversity conservation, in particular the following biodiversity schemes: BioBanking, BushBroker, Conservation Banking, Malua BioBank and Wetland and Stream Mitigation Banking.

1.5.2 Contract theory

As previously mentioned, clear and enforceable property rights are one of the features of market efficiency. A solution to define property rights in biodiversity markets is the creation of contracts. Contracts are institutional arrangements created with the purpose to facilitate exchange between two or more parties (Williamson 1979, Slangen et al. 2008). In particular, contract theory deals with the theory of incentives, information and economic institutions pertaining such exchanges (Bolton and Dewatripont 2005). In Chapter 2 of this thesis I use contract theory to assess the type of contractual arrangements from diverse type of transactions carried out in biodiversity markets. I use the framework proposed by Lyons and Mehta (1997) to classify the market-based mechanisms according to their type of contractual relationship.

1.5.3 Game theory

Game theory is *'a formal, mathematical discipline which studies situations of competition and cooperation between several involved parties'* (Peters 2008, p.1) that emerged with the interdisciplinary collaboration of Von Neumann and Morgenstern (1944). Agents interact with each other and choose a strategy upon a specific issue (e.g. pollution abatement, water use, conservation) based on such interactions and the resulting payoffs. This research approach is applied to Chapters 3-5 of this thesis.

Coalition formation games

Seminal literature regarding self-enforcing agreements includes the work of D'Aspremont et al. (1983), Carraro and Siniscalco (1993), Barrett (1994), and Chander and Tulkens (1997). Most literature regarding the economic analysis of the formation and stability of IEAs refers to the problem of pollution abatement and IEAs for greenhouse gas (GHG) emissions control (Barrett 1994, Pittel and Rübbelke 2012, Finus and Rübbelke 2013). There are only a few examples of IEA literature linked to the biodiversity domain (see Barrett 1994a, Punt et al. 2010, Punt et al. 2012, Winands et al. 2013, Walker and Weikard 2014). In this thesis I focus on coalition formation games to explain agents' behaviour in IEAs for biodiversity conservation. Specifically, I consider two-stage, cartel games to tackle research questions Q2-Q4. In the first stage of the game, countries decide to join or not the coalition; and in the second stage, those countries joining the coalition maximise their collective benefits of conservation. For the stability analysis I focus on the case where only one coalition is formed and where the interest relies in the size and composition of such coalition.

Several studies have investigated coalition formation with asymmetric countries (Fuentes-Albero and Rubio 2010, Pavlova and de Zeeuw 2013, Winands et al. 2013); and others have combined analytical results with simulation exercises (Eyckmans and Finus 2006, McGinty 2007, Finus et al. 2009, Nagashima et al. 2009, Dellink 2011). Throughout the thesis I analyse coalition stability for symmetric countries (Chapter 4), asymmetric countries (Chapter 5), and for both (Chapter 3). Furthermore, I develop simulation exercises to answer research questions Q2 and Q3. As for research question Q4, I calibrate the model with empirical information regarding habitat and species characteristics of the Golden-winged Warbler.

Transfer schemes

Transfer schemes are used as vehicles to increase the number of signatories in coalition formation. Several studies have addressed the effect of including transfer schemes cooperative games (Eyckmans and Finus 2006, Weikard et al. 2006, Weikard 2009, Fuentes-Albero and Rubio 2010, Finus and McGinty 2015). In Chapter 3 and Chapter 5 of this thesis I investigate the implications of including a transfer scheme on the stability of international biodiversity agreements. Specifically, the transfer scheme that is implemented in this thesis is based on an optimal sharing rule (Weikard 2009).

1.5.4 Industrial organisation theory

A subfield of industrial organisation theory focuses on industry location models (see Hotelling 1929 and Salop 1979). To answer research question Q3, I develop a general framework that includes a spatial structure capable of representing cooperation among neighbouring countries. I based this general framework on Salop's (1979) industry location model where countries are located on the circumference of a circle and therefore have two directly neighbouring countries. Furthermore, the use of this particular spatial structure allows to relax the assumption of unidirectional interactions between countries. Nevertheless, the adoption of a circular spatial structure is not essential to our approach: the model framework is general and would work for other spatial structures too.

1.5.5 Spatial scales

A novel aspect of this thesis is that it investigates economic approaches for biodiversity conservation that cut across different spatial scales. In Chapter 2, I examine the efficiency of biodiversity markets with different scales of implementation (e.g. provincial, national, and multinational). Moreover, I assess the potential to upscale such markets to a global market for biodiversity in order to protect areas that are rich in biodiversity but unprotected under baseline conditions. In Chapter 3, I address the mismatch between scales at which costs and benefits of biodiversity take place. I account – in addition to global benefits – for local benefits of biodiversity conservation as a relevant feature of the biodiversity game. In Chapter 4, I introduce a model for an international biodiversity agreement that includes regional benefits of biodiversity conservation. I consider this feature to be dependent on a defined spatial structure that reflects notions of distance and location. Chapter 5 does the same as Chapter 4, but this time considering a different spatial structure that best describes the migratory connectivity of the bird species at stake.

1.6 Outline of the thesis

This thesis comprises five additional chapters. Chapter 2 provides a review on existing market-based instruments for biodiversity conservation under the light of market and contract theory. I assess the upscaling potential of the market-based schemes under their current performance. Chapter 3 provides an analysis of key features specific to biodiversity conservation in the design of IEAs. It also includes a sensitivity analysis by changing relevant model parameters, and incorporates a transfer scheme based on an optimal sharing rule to examine its impact on coalition stability. Chapter 4 investigates the inclusion of a spatial structure in the design of an IEA for biodiversity conservation. I define a concept of distance and make use of a specific spatial structure that best represents cooperation among neighbouring countries. Chapter 5 explores the application of the spatial model from Chapter 4 to a setting of habitat conservation for migratory bird species. Finally, Chapter 6 provides the main conclusions of the study, policy implications and recommendations for further research.

Chapter 2

Market-based mechanisms for biodiversity conservation: a review of existing schemes and an outline for a global mechanism[1]

Continuous decline of biodiversity over the past decades suggests that efforts to decrease biodiversity loss have been insufficient. One option to deal with this problem is the use of market-based mechanisms for biodiversity conservation. Several studies have analysed such mechanisms individually, but there is no comprehensive review with a comparative assessment of the performance of various mechanisms. This chapter presents (i) an analysis of the economic conditions under which markets for biodiversity can be expected to function; (ii) an analysis of the efficiency of five selected biodiversity markets in the light of market and contract theory; and (iii) an assessment of the potential to scale up local or national payment mechanisms for biodiversity conservation. Our analysis shows the difficulties that market-based mechanisms face, among which are the need to ensure long-term conservation and the lack of a standardised unit of measurement for biodiversity. We provide a number of recommendations on how to overcome these difficulties. We argue that the set-up of a global registry embedded within the framework of the Convention on Biological Diversity (CBD) would facilitate measurement, reporting and verification of biodiversity credits to support market-based mechanisms.

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[1] This chapter is based on Alvarado-Quesada, I., Hein, L., & Weikard, H-P. (2014) Market-based mechanisms for biodiversity conservation: a review of existing schemes and an outline for a global mechanism. *Biodiversity and Conservation* 23: 1-21.

2.1 Introduction

Biodiversity loss is occurring at a fast pace and there are no indications that this trend is reversing (MA 2005, Butchart et al. 2010, Mora and Sale 2011). The abundance of vertebrate species fell by almost a third on average between 1970 and 2006, and continues to fall globally, with particularly severe declines in the tropics (CBD 2010). Among the major causes of biodiversity loss are habitat loss and destruction, introduction of invasive non-native species, overexploitation of natural resources, pollution and contamination, and climate change (UNEP 2011).

The continuous decline of biodiversity suggests that efforts to decrease or halt the rate of biodiversity loss have been insufficient and/or ineffective. These efforts include subsidies for land protection, debt-for-nature swaps, funding for protected areas, and multilateral assistance for biodiversity conservation (James et al. 2001, Pearce 2007). Market-based mechanisms have been proposed to deal with the problem of biodiversity loss as a complement for other existing conservation efforts (Bardsley 2003, OECD 2004, Simpson 2004, Kroeger and Casey 2007, Nijkamp et al. 2008, Hein and van der Meer 2012).

Several studies have analysed individual market-based mechanisms for biodiversity conservation (Fernandez and Karp 1998, Fox and Nino-Murcia 2004, Hallwood 2007, Burgin 2008). McKenney and Kiesecker (2010) present a descriptive and comparative study of five biodiversity offset mechanisms according to six criteria that the authors define as key issues for offset implementation. What is missing in the literature, however, is a comprehensive study that includes a comparative assessment of the performance of market-based mechanisms for biodiversity conservation in the light of market and contract theory.

The aim of this study is to present (i) an analysis of the economic conditions under which markets for biodiversity can be expected to function; (ii) an analysis of the efficiency of five selected biodiversity markets in the light of market and contract theory; and (iii) an assessment of the potential to scale up different local or national payment mechanisms for biodiversity. An advantage of scaling up markets for biodiversity to a global level (referred to as 'upscaling' from now onwards) is that areas rich in biodiversity but unprotected under baseline conditions might be preserved. Developing countries that are rich in biodiversity but short in funds for conservation purposes could therefore benefit from a global biodiversity market that would allow people from other countries to invest in conservation.

In order to achieve our objectives, we study five cases of market-based mechanisms for biodiversity conservation that use units of protected land as their transaction commodity in the market. The five cases of market schemes are: BioBanking, BushBroker, Conservation Banking, Malua BioBank, and Wetland and Stream Mitigation Banking. We first introduce the methodology to study the main conditions required for biodiversity markets to be effective. Then, we present an overview of the five cases of market-based mechanisms. Next, we examine the mechanisms in the light of these conditions and present their contributions and limitations to biodiversity conservation. Finally, we assess the upscaling potential of the market-based schemes. We conclude by summarising our findings.

2.2 Methodology

2.2.1 Biodiversity as an economic good

To study market-based mechanisms for biodiversity conservation one must clarify the concept of biodiversity that is considered and the type of good that it represents. A standard definition is given by the Convention on Biological Diversity (CBD) which states that biodiversity is *'the variability among living organisms from all sources, including, inter alia, terrestrial, marine, and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species, and of ecosystems'* (UN 1992, p.3).

Biodiversity can be interpreted as a public good: one cannot usually prevent people from enjoying biodiversity (non-excludability), and a person's enjoyment of biodiversity does not deplete its availability to others (non-rivalry). These public-good characteristics are causes of market failure. The price mechanism does not function well for the provision of such goods because consumers do not have an incentive to pay. Hence, producers do not have an incentive to supply. Consequently, an appropriate institutional design is needed to correct the market failure (Ostrom and Ostrom 1977). In addition to its public-good aspects, biodiversity also contains features of private goods (Salles 2011). Private benefits can be generated from activities such as bioprospecting or through the club-good features of biodiversity (e.g. ecotourism). Management of biodiversity, therefore, needs to facilitate efficient and sustainable use of both public and private services provided by biodiversity.

Biodiversity has multiple roles in the provision of ecosystem services: as a regulator of ecosystem processes; as a final ecosystem service; and as a final good (Mace et al. 2012). The latter role is the one that we emphasise in this research. The value of biodiversity as a final good is reflected in the demand for measures to conserve ecosystems and their biodiversity. This value is related to the public-good aspect of biodiversity and consequently is not represented in markets for biodiversity.

A complement of other existing conservation efforts that consider the value of biodiversity as a final good is the use of markets-based schemes for conservation of biodiversity in private areas. Governments facilitate, sometimes in collaboration with private partners, market-based structures to preserve biodiversity, where the good at stake is a unit of preserved habitat, or an individual or a group of species. The unit of preserved biodiversity is represented by a credit that can be traded in a market. The respective legislator is in charge of the definition of the property rights in order to facilitate the trade.

2.2.2 Market characteristics relevant for biodiversity markets

Several conditions need to be fulfilled to guarantee the efficiency of a market. First, clear and enforceable property rights must be established to control the management of the good. Property rights define the access, use and transfer of physical or more intangible properties, and also define the positions and responsibilities of the parties involved in the market exchange (Brousseau and Glachant 2002). When externalities arise due to missing markets for certain goods or services, property rights are implemented to develop markets and achieve efficient outcomes.

When property rights are well-defined, market efficiency requires the presence of large numbers of buyers and sellers. Under perfect competition, agents do not have market power and there is no room to set prices strategically. Another requirement for market efficiency is perfect information. Often producers have better information about the good than consumers. This information asymmetry could lead to market failure because the consumer is faced with a product of uncertain quality (Akerlof 1970). In addition to these conditions, market efficiency requires zero transaction costs and free access to and exit from the market.

2.2.3 Features of contract and transaction cost theory relevant for biodiversity markets

In a complete contract every contingency is anticipated, the associated risk is efficiently allocated between the parties, and all relevant information is communicated (Cooter and Ulen 2003). In reality, however, most contracts are incomplete because of three main factors: unforeseen future contingencies, difficulty of negotiations between parties over their individual plans, and the struggle to write down the agreement in a way that content and meaning could be enforced by an outsider (Hart 1995). In order to characterise the type of contractual relationship of each market-based mechanism according to their undertaken transactions, we consider the framework proposed by Lyons and Mehta (1997) that distinguishes three types of contracts: classical, neoclassical and relational. In a classical contract the identities of parties are irrelevant, a discrete transaction is specified, and written documents overrule any verbal agreement. In a neoclassical contract the identities of parties matter, its duration is fixed and written documentation provides status quo basis for negotiation. Finally, in a relational contract the identities of the parties are crucial, the duration is often unspecified and values and norms are of greater importance than written documents in settling disputes. Information asymmetries as a cause of market failure can lead to adverse selection and moral hazard problems in the market-based schemes.

The aforementioned conditions for the functioning of markets, the characteristics of contract design and transaction theory, and the study of opportunities for upscaling are the foundation of the analysis of the five selected cases of market-based mechanisms for biodiversity conservation.

2.3 Review of selected biodiversity markets

2.3.1 Selection of studied markets

The report '2011 Update: State of Biodiversity Markets' (Madsen et al. 2011) gives an estimate of 45 existing compensatory mitigation programmes worldwide in 2011. These include mitigation banking of biodiversity credits; programmes that channel development impact fees; one-off offset policies; and 27 additional programmes in development. For our study we selected five examples of market-based mechanisms which specify a traded unit for biodiversity conservation. The chosen sample intends to represent different features of market-based mechanisms. First, we include examples of both regulatory and voluntary markets. Regulatory markets are managed by regulatory bodies that set a limit to the degree of ecosystem use or damage permitted in an area, and allow firms and individuals to trade credits to meet their obligations. Voluntary markets occur when sellers and buyers enter a market without reference to regulatory requirements (TEEB 2010). From the five examples of our study, Malua BioBank is a voluntary market and the other four are regulatory markets.

Other aspects considered in the selection of our sample are the differences among market-based mechanisms in terms of location, operating times, methodologies for credit definition, and scale of implementation (e.g. provincial, national and multinational). Although the traded units in these five markets are credits or certificates for species or habitat conservation, each market has its own methodology to define credits and uses different units (e.g. hectares or breeding pairs). Biodiversity credits traded in these markets are not always homogeneous, either between or within the markets. We present an overview of the main characteristics of each of the five markets in Table 2.1.

2.3.2 Review of the cases

BioBanking (Australia)

The BioBanking Scheme is a market-based mechanism of biodiversity credits created in New South Wales (NSW), Australia, to address the loss of biodiversity. The scheme introduced by the NSW government provides a framework to (i) assess and manage biodiversity offsets created to counterbalance impacts of land use damage, and (ii) to create incentives for landowners to generate profits by managing their private land for conservation (DECC 2007, OEH 2012). The 'Compliance Assurance Strategy' is the regulatory guideline of BioBanking. The premise of the scheme is that the purchase of credits secures conservation of biodiversity in perpetuity (DECC 2007).

Traded units are ecosystem credits and species credits. The number and type of biodiversity credits per area are calculated using an assessment methodology and a credit calculator developed within the scheme (OEH 2011). Landowners and the Minister for Climate Change and the Environment sign biobanking agreements that enable landowners to generate biodiversity credits. This agreement is attached to the land title, which means that future landowners are also bound by the agreement. After establishing such agreements, developers can buy credits from landowners to offset their development projects. Prices of credits depend on the characteristics of the site (e.g. value of the land, condition of the vegetation). They vary from AU$2500 to AU$9500 per credit (OEH 2012).

Buyers of credits are developers seeking to offset biodiversity loss from an approved development site, or individuals or groups interested in nature conservation. Sellers are landowners with properties suitable to generate credits who commit to enhance and protect biodiversity in their own lands. Until May 2012 BioBanking registered the transfer of 1272 ecosystem credits in the scheme. No transactions of species credits have taken place yet (OEH 2012a).

BushBroker (Australia)

BushBroker is an initiative established by the State Government of Victoria in Australia. It intends to improve the quality and extent of native vegetation by following the 'net gain approach': avoid adverse impacts, minimise impacts if they cannot be avoided, and identify the proper offset options (DSE 2009a). BushBroker registers and trades credits for use as offsets for development activities when there is no suitable site on the property to be protected or when the buyer cannot manage the native vegetation in the long run. BushBroker is not involved in the negotiations; it only records the information and prepares the landowner agreements (between landowners and the Secretary of the Department of Sustainability and Environment (DSE)), and the credit agreements (between landowners and permit holders) (DSE 2011).

Table 2.1. Summary table of selected market-based mechanisms for biodiversity conservation [a/]

	Criteria							
Market-based mechanism	Number of buyers	Number of sellers	Traded unit	Number of transactions	Time span of agreement	Price per unit	Payment	Regulatory body
BioBanking (Australia)	4 developers	3 biobanking sites	Ecosystem or species credits [b/]	1272 ecosystem credits purchased and retired from the market, for a total market volume of AU$4,697,208.	Conservation in perpetuity	From AU$2500 to AU$9500 excluding GST (Goods and Services Tax)	One single payment from the developer to the Trust Fund. From this money, the landowner gets an annual payment to manage the biobank site.	Office of Environment and Heritage of New South Wales (OEH)
Bush Broker (Australia)	n.a.	approx. 50 landowners	Native Vegetation Credits (NVCs) of 3 types: vegetation or habitat, 'large old trees' (LOTs), and 'new recruits' [c/]	Over 450 credit agreements. Total market volume: AU$34 million.	Protection and maintenance of native vegetation in perpetuity	From AU$20,000 to AU$400,000 per habitat hectare alone or habitat hectare + LOTs, and from AU$500 to AU$2,300 for LOTs only [d/] (all prices excluding GST)	One or various payments from the permit holder to the Secretary of the Department of Sustainability and Environment (DSE). This money is held in a trust for subsequent payments to the landowner for 10 years over the total amount stated in the agreement, and for costs associated with permanent protection of the site.	Department of Sustainability and Environment of the State of Victoria (DSE)
Conservation Banking (United States)	n.a.	111 active and sold-out banks	Habitat or species units (measured in acres of protected habitat, breeding pair, etc.)	On average total payments for conservation per annum: US$200 million.	Permanent conservation of land in conservation bank	From US$2500 to US$300,000	Generally an endowment is generated to fund the long-term management of the conservation bank, which includes monitoring and maintenance of habitats for species.	US Fish and Wildlife Service (USFWS) and the National Marine Fisheries Service (NMFS)
Malua Biobank (Malaysia)	n.a.	one –the bank itself	Biodiversity Conservation Certificates (BCCs)	n.a.	At least, 50 years of conservation	From US$5 to US$100	One single payment from the individual or company to the Malua Biobank.	The Malua Trust, a steering committee and an advisory committee
Wetland and Stream Mitigation Banking (United States)	n.a.	711 active and sold-out banks	Wetland and stream offset credits (measured in acres of protected habitat, functional assessment method, or a combination)	On average total payments for conservation per annum: $US1.3 to US$2.2 million.	Wetland protection is permanent	From US$3000 to US$653,000 for wetland credits and from US$15 to US$700 for stream credits	Long term management of the site must be guaranteed and endowed by the bank sponsor through financial assurances such as performance bonds, escrow accounts or casualty insurance.	US Army Corps of Engineers (USACE), US Fish and Wildlife Service (USFWS) and the National Marine Fisheries Service (NMFS)

a/ The table intends to include the most recent information available from each market-based mechanism.
b/ Determined by the BioBanking Credit Calculator, based on the rules stated in the BioBanking Assessment Methodology.
c/ Credits are based on ecological vegetation classes (EVCs) within Victoria's 28 bioregions, accounting for 2,500 possible types of EVC credits.
d/ Price values for both habitat hectare credits and large old tree credits correspond to the available information of 80% of agreements in the Victorian bioregions.

Traded units are Native Vegetation Credits (NVCs). NVCs represent a gain to the extent and/or quality of native vegetation that, according to the BushBroker rules, comes attached to a secure, on-going agreement. All credits are registered in the Native Vegetation Credit Register and attached to the land title. This register binds current and future landowners to the completion of a 10 year Management Plan (DSE 2009) that states how the native vegetation will be protected and maintained in perpetuity. Prices are set through the market, from a minimum of AU$500 for large old-tree credits, to a maximum of AU$400,000 for habitat hectare credits[2].

Buyers (or permit holders) are developers interested in offsetting their activities through the purchase of credits. They can select their preferred matching credit and BushBroker then provides information on landowner(s) in charge of the credit, in a way that both parties can negotiate a transaction. There can also be buyers of credits mainly interested in the conservation of nature. Sellers are landowners with suitable land to implement NVCs. Approximately 50 landowners have engaged in trading to date (Barrett 2011).

Conservation Banking (United States)

Conservation Banking is a market mechanism created to offset adverse impacts to endangered or threatened species by protecting lands that contain natural resource values. The mechanism emerged in the early 1990s using wetland banking as leading example of a mitigation system. By 2003, the official federal guidance for the establishment, use, and operation of conservation banks was formally released. Conservation Banking acts as an incentive for landowners to permanently protect their lands because they can benefit from selling species or habitat credits while keeping their land intact and obtaining other benefits, such as tax reductions (USFWS 2011).

Conservation banks sell habitat and species credits. A credit can represent an acre of habitat for a particular species, the size of habitat to support a breeding pair, a linear foot of riparian habitat (for aquatic species), or some other measure of habitat or its value to a species. The range of prices for Conservation Banking credits is between US$2,500-US$300,000. The average price of a credit is US$31,683[3] (Madsen et al. 2010a).

Buyers are public or private developers that may impact a threatened or endangered species and require an offset mechanism e.g. government agencies, private firms, extractive industries. Credit sellers (or bank owners) are conservation banks that produce credits. Sellers are often firms or government agencies in need of mitigation, but recently there are more private landowners and specialised companies in mitigation

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[2] Quoted prices values for habitat hectare credits and large old tree credits reflect 80% of agreements in the Victorian bioregions.

[3] Average price of data from 2005-2009 for different types of conservation credits. Six unit-based credit price points were not used to obtain the average price.

banking involved in the market (Ecosystem Marketplace 2011). There are 130 banks registered in the Ecosystem Marketplace dataset up to November 2011, of which 111 are either active or sold-out banks (banks that sold out all their credits) (Species Banking 2011).

Malua BioBank (Malaysia)

Malua BioBank is a public-private partnership between the government of Sabah, Malaysia and several private corporations. This partnership was created to invest in the rehabilitation and protection of the Malua Forest Reserve in Malaysian Borneo. The bank was launched in 2008 and encloses 34,000 hectares of critical habitat for orang-utans and other endangered species. The Sabah Government committed to stop all logging activities in the Malua Forest for a period of at least 50 years provided that enough credits are sold. Conservation management activities of the Malua Forest are stated in the Conservation Management Plan and will be financed and supervised by the Malua Trust endowment (MWHCB 2010). The progress of the plan is reviewed on an annual basis by a steering and an advisory committee.

The bank sells Biodiversity Conservation Certificates (BCCs). Each certificate represents a minimum of 50 m^2 of restoration and protection of the Malua Forest for a period of 50 years, and it goes up to 1000 m^2. All certificates are listed on Markit Environmental Registry (Markit Group 2011). The price per credit goes from US$5 for the minimum protected area (50 m^2) up to US$100 for a protected area of 1000 m^2 (Piqqo 2012).

Buyers are individuals and businesses interested in supporting biodiversity conservation and ecosystem restoration as viable land use activities. Businesses can include the BCCs into their operations to address environmental impacts of their activities. Contrary to other schemes, purchases of BCCs represent a contribution to rainforest conservation in Malua and not an offset mechanism for impacts on rainforests in other places. The only seller is the Malua BioBank.

The direct retail programme from Malua BioBank finished in the beginning of 2011 when they had already sold a certain amount of credits. After that the only way to engage in the project was to provide small funds through a partnership with the tea company Tetley. By the end of 2011 a new website was launched in which individuals could preserve an area by donating a contribution and receive personalised certificates (not credits) in return. Currently the bank is looking for big corporations to become involved in supporting this business model.

Wetland and Stream Mitigation Banking (United States)

The wetland and stream offset programme in the United States in known as compensatory mitigation. This programme consists of the creation of offsets via restoration, establishment, enhancement and/or preservation of aquatic resources.

Within compensatory mitigation, Wetland Mitigation Banking is a national offset programme that is regulated by federal policy. Implementation, nevertheless, takes place at a regional level in 38 Districts of the United States Army Corps of Engineers (USACE) with banks located in 11 states. The long-term management of the site is supposed to be guaranteed and endowed by the bank sponsor in a way that the wetland functions will be protected in perpetuity for the sold credits (Ecosystem Marketplace 2010).

Traded units are offset credits that cover different types of wetland and stream ecosystems. Credit calculation methods differ according to the region where they are applied. Most of them are based on acreage, a functional assessment method, or a combination of acreage and functional assessment (Madsen et al. 2010). The national range of wetland credit prices is between US$3,000 and US$653,000 and the average credit price is US$74,535[4]. For the case of streams, the national range of price is between US$15 and US$700, with an average price of US$260 (Madsen et al. 2010a).

Buyers are public or private developers in need of a permit to impact a stream or wetland. They can be non-profit, government and for-profit organisations. Sellers of credits are the mitigation bank sponsors. There are 711 banks registered in the Ecosystem Marketplace dataset up to January 2011. This number includes only the active and sold-out mitigation banks (Species Banking 2011).

It is not easy to compare and determine the level of success of these five market-based schemes up to date for two reasons. First, each scheme has its own methodology to define credits. Also, the Australian mechanisms as well as Malua BioBank are relatively recent compared to mitigation banking in the United States, which has been taking place for more than a decade in the case of species conservation, and for more than two decades in the case of wetland mitigation. To facilitate the comparison, we chose the average market volume and average protected area per year as indicators of performance (Table 2.2).

Results in Table 2.2 show that Wetland and Stream Mitigation Banking is the scheme that has the largest market volume per year, with around US$2 billion transferred per year for conservation purposes. The lowest market volume corresponds to BioBanking with transactions slightly over US$1 million. It is important to mention, however, that mitigation banking in the United States takes place at a national scale, whereas BioBanking and BushBroker are state initiatives. In terms of protected area, Wetland and Stream Mitigation Banking also has the highest average protected area per year. For Malua BioBank there is no information available on market volume or protected hectares so far.

[4] Average price of data from 2005-2009 for different types of credits. Tidal wetland credits and vernal pool wetland credits are not included because their prices are considerably higher than regular wetland credit prices.

Table 2.2. Indicators of performance for selected market-based mechanisms

Market-based mechanism	Criteria		
	Date of commencement	Average market volume (US$ per year)[a/]	Average protected area (ha per year)[b/]
BioBanking (Australia)	July 2008	1.1 million	100
Bush Broker (Australia)	2006	5.1 million	21[c/]
Conservation Banking (United States)	May 2003	200 million	3,760
Malua Biobank (Malaysia)	August 2008	n.a.	n.a.
Wetland and Stream Mitigation Banking (United States)	1980s[d/]	1.3- 2.2 billion	16,660

a/ For the two Australian schemes it represents the total market volume divided by the years of functioning (from date of commencement of each mechanism up to and including 2012). For the two US schemes it represents the total payments for conservation in 2008 for Wetland Banking, and total payments for conservation in 2009 for Conservation Banking.
b/ Estimations of total protected area divided by the number of years of functioning of the market scheme (from date of commencement of each mechanism up to and including 2012).
c/ These are habitat hectares. Habitat hectares consist of the product of the number of hectares times a 'habitat score' that can go from 1 to 100, e.g. 10 hectares with a habitat score of 50% count as 5 habitat hectares.
d/ Wetland Mitigation started in the early 1970s, but more sophisticated mitigation credit banking systems started in late 1980s and early 1990s. Records of transactions are calculated from 1990 onwards. This makes more difficult the comparison with the rest of the market schemes.

2.4 Assessment of selected biodiversity markets based on market and contract theory

2.4.1 Market efficiency

Clear and enforceable property rights

One of the key requirements for developing ecosystem markets is the presence of well-defined and enforceable property rights (TEEB 2010). Markets for ecosystem services differ significantly in terms of government regulation and the maturity of supporting institutions. For example, on one end of the spectrum the US wetland mitigation scheme has clear legal requirements, well-defined liability and enforceable property rights. On the other end, developing countries that are currently preparing to design and implement REDD+ programmes are still struggling with strengthening security of land tenure and of property rights in forests and with the improvement of systems to enforce these rights (Williams 2013).

Rights of land management and ownership are established in the agreements. When buyers purchase their credits, they are paying to ensure the protection of land in perpetuity. However, there is uncertainty about whether there will be enough funds in the future to continue the implementation and the long-term monitoring of the mechanisms (Burgin 2008, Wotherspoon and Burgin 2008). Also, in case funds are available, the best way to enforce management requirements in the long term is unknown (O'Connor-NRMPty 2009). An example is the agreement between the Department of Forestry of Sabah and Malua Biobank, which has established current conservation rights to the Malua Forest reserve for a period of 50 years. After that period, the funding endowment is expected to be fully capitalised and will be used either to renew the conservation rights of the reserve or to establish a conservation bank in a different area with high biodiversity value. In case of non-compliance, investments and total revenues from sales of the credits must be repaid to the Malua Trust (MWHCB 2010). If there is compliance with the plan and funds are sufficient to protect the entire reserve for 50 years, there is no guarantee that the protection of the area will continue after the conservation rights have expired. In conclusion, rights of credit buyers to demand secured conservation in the long run are confronted with the inherent difficulties of ensuring monitoring and compliance of activities into the far future.

Number of buyers and sellers: Thinness of the markets

Thin markets are characterised by small numbers of buyers and sellers (Pagano 1989) and are considered a fundamental problem for administration of biodiversity offsets (Walker et al. 2009). In the US schemes there is no indication of market thinness: the average market volume is US$200 million per year for Conservation Banking[5] and US$1.3-US$2.2 billion per year for Mitigation Banking[6] (Madsen et al. 2010a) (Table 2.2). Agents from Wetland and Conservation Banking have the possibility to compare credit prices with other offset alternatives: in-lieu fee programmes and permittee-responsible mitigation. The alternative to choose between the different options through RIBITS (the system facilitating information on mitigation and Conservation Banking and in-lieu fee programmes across the United States) ensures competitiveness.

BioBanking and BushBroker can be considered thin markets. Both market-based mechanisms have been operating for less than seven years and the overall number of transactions in each market is relatively low as compared to the Conservation Banking or Wetland Mitigation Banking schemes. In BioBanking, the availability of credits for some types of vegetation can be limited at specific points in time, which can restrict

5 This information only considers Conservation Banking and not species compensation through in-lieu fee funds or permittee-responsible mitigation.

6 This information includes data from In Lieu Fee Programmes, Permittee-Responsible Mitigation and Mitigation Banking.

the participation of developers in the market (OEH 2012). As for BushBroker, a lack of understanding among landowners of the functioning of the mechanism appears to be a main barrier to participation in the scheme (O'Connor-NRMPty 2009). There is no information available about the number of credits sold in the Malua BioBank. However, given that the scheme has only recently started and that there are few comparable schemes (Madsen et al. 2011), it is likely that also the voluntary global biodiversity credit market can be seen as a thin market.

From the five selected biodiversity schemes, it can be observed that market volume is higher for those schemes that have been functioning for a longer time frame, and that have additional and accurate information to compare their credits to other similar activities. Evidence of market thinness in our selected sample is related to recent mechanisms with limited availability of credit options.

Incomplete information/ information asymmetry

Incomplete information as an obstacle for the proper functioning of biodiversity markets has been acknowledged in several studies (Lerch 1998, Goeschl and Lin 2004, Walker et al. 2009). The analysis of the selected schemes with respect to asymmetric information is similar for the US and Australian schemes: the problem of adverse selection is not frequent in these mechanisms. Adverse selection is an ex-ante information problem, which implies that one party is better informed about the good being traded than the other. Credits are defined and assigned to the landowners by a third party, therefore the quality and characteristics of the land are assessed when the credits are created. This assessment guarantees a minimum level of quality for credits to be traded. The case of Malua differs because it is only known that each credit represents a unit of forest being preserved; the quality of conservation in situ is not assessed. An example of a signalling mechanism used to minimise this problem is the reference of World Wide Fund for Nature (WWF) and South East Asia Rainforest Research Programme (SEARRP) as partners of the Malua initiative. The endorsement of these renowned organisations underlines the importance that conservation certificates meet certain quality standards, e.g. the Gold Standard for optimal carbon offsets supported by WWF.

Moral hazard problems are more frequent in all market-based schemes. Landowners in charge of protecting their properties (or for the Malua case the biobank itself) may neglect conservation activities agreed in the contracts (e.g. grazing) or may engage in activities that are prohibited under the contract (e.g. intentionally removing native vegetation). Opportunistic behaviour may arise from the fact that the contracts establish protection of the sites for a long term or even perpetuity, while landowners anticipate changes in the rules of the game from the side of the governments. Commitment to conservation in perpetuity through governmental regulation has to be credible in the eyes of the landowners. Original rules for the contracts of three schemes (BioBanking, BushBroker and Malua) were established under the guidelines of governments at a given time and are still valid to date. However, mitigation banks in the United States have already displayed changes in their regulatory frameworks.

The main instrument used by the schemes to reduce moral hazard is the implementation of monitoring measures. For the Australian schemes, landowners are required to provide an annual report detailing undertaken actions and path towards expected goals. Also, the Office of Environment and Heritage for BushBroker (OEH) and the DSE for BioBanking monitor a number of sites randomly to ensure the correct implementation of the agreed contracts. The US schemes are subject to monitoring from the USACE for the case of Wetland Banking, and from the UN Fish and Wildlife Service for the case of Conservation Banking. Both wetlands and species conservation are driven by federal policy but implemented at a regional level. This allows for a wide range of differences in the regional interpretation of national regulations. New regulations for compensatory mitigation on wetlands came into effect in 2008, and there is evidence that local offices of the USACE were enforcing such rules unevenly (Madsen et al. 2010).

For some schemes punishments in case of non-compliance do not represent a strong threat for landowners or conservationists because the terms in which they would be executed remain unclear. The BioBanking Compliance Assurance Strategy states that the OEH may take actions in case of an agreement being breached, but it does not specify any type of punishment or fee other than withholding payments to landowners if necessary. Malua does not offer any kind of reimbursement to credit buyers in the event of non-compliance with the plan. The only requirement is that all investment and total revenues from the sales of credits have to be repaid to the Malua Trust (MWHCB 2010). The case of Wetland Banking is different. The scheme is regulated under the Clean Water Act and, as many other facilities regulated under high-profile environmental laws, banks have to file regular self-monitoring reports as source of information on compliance. These reports are generally considered reliable because of the high administrative penalties or even criminal prosecutions in case of violations or falsifications of the reports (Gray and Shimshack 2011).

From the analysis we conclude that opportunistic behaviour is a common threat for all schemes due to the long-term span of the contracts and the possible anticipation of changes in the regulatory frameworks. Monitoring is the common procedure to avoid moral hazard. Still, penalties due to noncompliance are in some cases low, unclear or non-credible and hence are not perceived as real restraints.

Transaction costs

One of the factors obstructing mainstream business participation in emerging markets for payment for environmental services (PES), and for biodiversity and ecosystem services are high transaction costs for the investors (Lambooy and Levashova 2011). Generally in payment schemes transaction costs are highest when many smallholders are involved, when institutions and property rights are weak, and where monitoring performance of the scheme is costly (Wunder 2007), for instance due to inaccessibility or the size of the land area involved.

Transactions that are asset-specific and infrequent are often carried out with the help of an external intermediary (Slangen et al. 2008). This is the case for four of the studied market-based schemes. Only Malua BioBank has no intermediaries to facilitate the contact between buyers and sellers of the credits because the bank itself is the seller. Presence of these intermediaries reflects transaction costs.

In all mechanisms covered by our study buyers incur the cost of searching for the right type of credit to offset impacts on biodiversity or to achieve their desired level of conservation. Ex-ante transaction costs that landowners have to bear are the fees related to submission of an expression of interest, offset matching and for the landowner agreement. For instance, for the case of BioBanking, landowners have to pay a consultant fee for assessment of their land (in the order of AU$10,000) and an application fee to the OEH (AU$648). When the fees cover only the costs made by the biodiversity bank without any supplementary charge or hidden subsidy, these fees are indicators of transaction costs.

Free access to and exit from the market

Studies on markets for biodiversity have identified several types of potential entrance/exit barriers to the markets. FAO and IFAD (2008) found that land-based climate change mitigation projects in rural areas faced barriers to enter the carbon market such as high start-up and transaction costs, expensive entry fees, insufficient knowledge about project registration cycles, small project scale, and fragmentation. Miyata (2007) identified possible barriers that prevented suppliers from certifying forest management practices: high initial costs of certification and uncertainty of price premiums, among others.

There are clear examples of barriers to enter the biodiversity market-based mechanisms. In the four regulatory markets the land of bankers or landowners has to be evaluated by the respective entity to determine whether the area is suitable for credit implementation and, eventually, for the sale of credits. Landowners can engage in the market only after the evaluation and the approval from the entity. Also, the two Australian schemes request a payment fee from credit purchasers and credit suppliers to enter the market.

With respect to exit barriers, credit suppliers (landowners) have to bear the costs of interrupting an agreement. In some cases an agreement termination has to be submitted accompanied by the appropriate fee (DECC 2009), although the amount of money to be paid is not specified. For Conservation Banking, every conservation agreement must provide a method for disposal of the property to a third party capable of continuing the management of the site in case that the current owner is unable to do so for any reason (USFWS 2003). Alternatively, landowners from BushBroker are required to pay back associated costs of non-compliance to the DSE and to create a replacement offset for the cleared site (DSE 2009). For BioBanking, the Minister for the Environment can seek an award of damages against the owner of the biobank site for a breach of a biobanking agreement (NSW 2006). The magnitude of such damage is not specified either.

Hence, there are exit barriers for credits suppliers in all reviewed biodiversity schemes, although the level of strictness varies. Conversely, participation of credit purchasers in most market schemes is reduced to a single transfer of money. After that payment, buyers do not face any other obligations.

2.4.2 Contract theory

As stated in the previous subsection, one of the characteristics to guarantee market efficiency is clarity and enforceability of property rights. In order to define the property rights for biodiversity markets, contracts must be established. Contracts need to fulfil certain criteria relevant for establishing a market. For biodiversity markets in particular the most important of these criteria are the rights of land management, land ownership and the duration of the agreement. The market-based mechanisms we consider here are examples of incomplete contracts. The main missing factors in the contracts are possible contingencies that can arise but cannot be anticipated, and the fact that not all possible events can be included in the written agreements for future examination and enforcement.

Two types of contracts can be identified in all market-based schemes. The first refers to transactions of credits between buyers and sellers and the term of the contract is short. The second refers to contracts between the landowner and the banking system that specify the terms and conditions in which the land or the defined species will be preserved. The latter agreement is for the long term. This kind of agreement is the focus of our analysis.

Type of contracts

By using the framework from Lyons and Mehta (1997) and the spectrum of contracts from Slangen et al. (2008) for our analysis, we can observe that all five selected market-based mechanisms are examples of neoclassical contracts, with some characteristics of a classical contract. Contracts state a long-run arrangement: for the case of Malua, land is preserved for a minimum of 50 years, and for the rest of the schemes land is preserved in perpetuity. Identity of the parties does matter in this relationship: credit buyers need to find the most suitable credits to fulfil the requirements of their specific offset activities among all the supplied credits (only the Malua scheme offers one single type of credit). Another feature is that regulatory entities must monitor landowners' performance.

Asset specificity is large for the credits: site location of the protected area is important and once it is used for conservation purposes, it cannot be used to develop other activities. As for the coordination mechanisms for these schemes, there is a combination between the role of the price and the 'handbook' (rules, directives and safeguards specified in the contract) as coordination mechanisms. For the case of buyers who wish to contribute to biodiversity conservation, credit price represents the coordination mechanism. For developers who need to meet requirements for offsetting activities, not only price but also some specifications are required for the transaction to take place. The importance of safeguards is high in neoclassical contracts and positively linked to high asset specificity. This applies to the five schemes. Safeguards in written documentation are important for the case of nature conservation if arbitration procedures are required.

The role of reputation is important when parties want to show commitment to their contracts. For example, BioBanking started in 2008, yet all but one biobanking agreements have occurred over the past two years. Developers and landowners have become more familiar with the scheme over time and consider it as a trustful land management alternative.

2.5 Upscaling potential of selected biodiversity markets

The scaling up of new markets for biodiversity and ecosystem services could represent major business opportunities and a significant solution to the biodiversity finance challenge (TEEB 2010). Most of these market mechanisms have been developed for specific ecosystem services (such as carbon sequestration or hydrological services), rather than for biodiversity as such (i.e. regardless of the economic benefits provided by biodiversity). In particular the REDD+ mechanism, once it becomes effective, would be able to generate significant funds for sustainable ecosystem use, however only for those ecosystems that contain important carbon stocks. The question now is if the various market mechanisms for biodiversity conservation, as reviewed in this chapter, have the potential to be scaled up to the regional or global scale. A specific consideration in this regard is that much of the world's biodiversity is concentrated in the tropics, whereas many low and middle-income countries located in the tropics have limited resources to pay for biodiversity conservation (Hein et al. 2013).

From the review of the five selected biodiversity markets, we have examined the main obstacles to scale up these schemes. First, it may be difficult to charge entry fees in some regulatory biodiversity markets at a global level. Fees are used as tools to finance monitoring activities and payments to landowners. Upfront costs of establishing an agreement have prevented landowners from establishing protected sites in their properties (OEH 2012b). Hence, if landowners would consider that transaction costs to engage in a global voluntary biodiversity market are too high, they might be discouraged to carry out conservation activities in such a market and would use their properties for other purposes instead.

Another obstacle for voluntary and regulatory biodiversity markets is the diversity of methodologies used to define tradable units. Each particular scheme has its own measurement unit to define its credits. This variety hinders the comparison between credits from different schemes.

The intention to offset development activities through the purchase of credits with different biodiversity values than those in the area being impacted is already generating controversy in biodiversity markets (OEH 2012). The issue of adequacy of offsets could become even more important when upscaling biodiversity markets to a regional or global context. For example, credit buyers from regulatory markets could find suitable credits to counteract for their local damages in another country. The vast variety of ecosystems worldwide makes it hard to define equivalences between preserved areas and areas altered by developments in different regions of the world. This could, at worst, lead to the destruction of certain biodiversity values that are not compensated through conservation in other sites. On the other hand, buyers on voluntary markets (conservationists) could buy credits in areas with higher levels of biodiversity than those found in their own countries.

One common obstacle for all market-based schemes is the difficulty to ensure conservation in perpetuity in a credible way. Uncertainty exists on whether there will be sufficient funds to cover the maintenance costs of landowners in the long run. Furthermore, changes in governmental policy act as a threat to the credibility of the agreements.

Finally, punishments and actions in the event of non-compliance with conservation activities are negligible or unclear for some of the selected biodiversity markets. As we observed for Wetland and Conservation Banking, federal regulation on environmental protection is difficult to implement evenly at a national scale. Monitoring activities could become more complicated when being expanded to an international context for a global market for biodiversity credits.

None of the selected market-based mechanisms, as they are currently performing, is suitable to be easily scaled-up. Yet, in light of their current obstacles, we can derive the conditions to overcome them. First, although not easy, the standardisation of a biodiversity unit worldwide would be an important step to facilitate comparison between credits offered by the different schemes. Moreover, it would be a necessary step to expand biodiversity markets to a global scale. This standardisation process is also compulsory in the definition of equivalences of biodiversity values between offset areas and development areas because credits would be defined in a homogeneous way worldwide. As for monitoring activities, schemes could make use of remote sensing techniques to cope with the complexities of standardising monitoring activities. This would help current markets and would allow for further development of biodiversity markets at an international level. Finally, further research must be done to deal with the lack of credibility in long-term arrangements for biodiversity conservation, which remains to be an important obstacle for a larger scale development of these markets.

We argue that there is scope to develop a global registry to support measurement, reporting and verification (MRV) of biodiversity credits. The registry would support both sellers and buyers involved in the various emerging markets for biodiversity, as well as in the design of biodiversity offset mechanisms, where a number of related issues are at play, e.g. development of standards for MRV, development of biodiversity units (Bishop et al. 2008, BBOP 2012); see also Loreau et al. (2006) and Hein et al. (2013) for potential additional tasks to be conducted under such a mechanism. This registry would have to be voluntary to ensure that those interested in preserving land through the sale of credits have the possibility to engage in it. The tropics are consistently emphasised as priority areas for biodiversity conservation (Brooks et al. 2006), but the willingness and ability to pay for biodiversity protection is the highest in OECD countries of the temperate zone (Hein et al. 2013, Miller et al. 2013). Therefore, a global registry could assist in the scaling up and implementation of biodiversity payment schemes worldwide.

The Malua BioBank is an example of a voluntary market for biodiversity conservation. The scheme allows for conservation of a biodiversity-rich rainforest, and it is financed by the private sector through a public-private partnership. It could be very useful to apply this model of conservation in biodiversity-rich areas located in developing countries where institutions are weak or funding is not sufficient for the required conservation efforts. Other initiatives deal with similar conservation projects, such as Conservation International (CI) and WWF. The Business and Biodiversity Offsets Programme (BBOP) is another example of a voluntary programme, initiated by industry, policy makers and NGOs in 2004 in order to develop standards for biodiversity offsets. However to date the BBOP standard on biodiversity offsets is still a first version. BBOP members are currently looking for organisations willing to test and refine the standard based on experience and practice (BBOP 2013).

Individuals or businesses may lack enough incentives to participate on voluntary initiatives for conservation. This constraint on voluntary markets reflects the public-good aspect of biodiversity: everyone is entitled to enjoy it, but there is often no strong incentive for people to preserve it unless they are forced to do so through a regulatory mechanism (Pearce 2007). Even if there is a voluntary incentive to preserve, voluntary contributions for conservation are not enough to compensate for the on-going decline of biodiversity.

The lack of trust in a voluntary global market for biodiversity can be addressed by involving well-known global NGOs (e.g. CI, WWF) and/or UN agencies (such as CBD). Moreover, the creation of a global registry with an overview of transactions and credit availability in the biodiversity markets would be an efficient way to inform potential buyers of the available opportunities to purchase biodiversity credits. Buyers can keep track of the available credits for biodiversity conservation through an online registry and decide which type of credits would best suit their offset and conservation activities according to the type of vegetation, equivalence of biodiversity units, and price of the credits. The benefits of setting up a registry include the strengthening of existing and new market schemes by providing technical support and credibility, and supporting potential buyers in finding and purchasing credits.

It needs to be examined in which international agency this global registry could best be placed, based on the selection criteria expertise, credibility and efficiency. Our suggestion is that a new, dedicated registry should be set up within the Secretariat of the CBD. We consider that this is a suitable place where such a registry should be embedded because of the global mandate of the CBD to promote the conservation and sustainable use of biodiversity. Currently, four countries in the world are not parties to the CBD: United States, Andorra, South Sudan and The Holy See (Vatican City); and it needs to be examined if and how these countries could be made eligible for participation, perhaps through a separate agreement covering participation in the global registry for biodiversity credits. The registry should focus on international biodiversity markets, and also offer existing international biodiversity credit schemes such as the Malua Biobank the opportunity to participate. National markets have specific, well-defined contractual and MRV requirements specified by national regulations but may still benefit from specific technical support or possibilities to register credits.

Further research is required to determine the most appropriate institutional setup of the global registry. It should be linked to the key international players (e.g. Global Environment Facility (GEF), the International Union for Conservation of Nature (IUCN)), United Nations Environment Programme (UNEP) and the World Bank) and should also involve key environmental NGOs as well as selected private sector entities with an interest in biodiversity markets and/or offsets.

2.6 Conclusions

This article assesses the functioning of five market-based mechanisms for biodiversity conservation: BioBanking, BushBroker, Conservation Banking, Malua BioBank and Wetland and Stream Mitigation Banking. Conditions that we analyse as critical to the efficiency of these markets involve: i) clear and enforceable property rights, ii) a sufficient number of buyers and sellers, iii) information completeness, iv) transaction costs, and v) barriers to entry and exit of markets. We also analyse the type of contract of the five schemes that we examined according to the classification proposed by Lyons and Mehta (1997).

Our analysis shows marked differences between the examined schemes. Conservation and Wetlands Mitigation Banking are consolidated schemes with high market volume. The frequent trades occurring in these schemes are considered as indicators of success in offering offset alternatives for developers in the United States. The Australian BioBanking and BushBroker schemes have increased their activities in the recent years, although their market volume is still considerably smaller than those of the US schemes. This increase indicates that landowners and developers are getting more acquainted with the system. Still, high entry costs remain an obstacle for scaling up these schemes. There is no information available on the performance of the Malua BioBank in terms of amount of sold credits.

Our findings show that a common limitation for all market-based schemes is ensuring long-term conservation. It is difficult to avoid the uncertainty around the availability of sufficient funds to cover the maintenance costs of landowners as well as the monitoring activities for periods of several decades. Furthermore, changes in governmental policy related to the schemes may threaten the credibility of the agreements. Landowners can anticipate changes in regulations and show opportunistic behaviour, which results in inefficient outcomes for biodiversity conservation.

With respect to upscaling possibilities, we find that none of the market-based mechanisms can be easily scaled up internationally in the way that they are currently set up. We derive conditions to overcome obstacles that impede the upscaling of the schemes: a standardised measurement of biodiversity for its tradable units, detailed monitoring of the activities (e.g. by applying remote sensing techniques), credible safeguards for actual implementation, and a credible information system or registry to assess and compare biodiversity credits. We recommend to set up a global credit registry in support of biodiversity markets (and potentially biodiversity offset mechanisms), possibly embedded in the Secretariat of the CBD, and in direct relation to existing institutions such as the World Bank or GEF. This global registry should be established in order to provide technical support in particular in relation to MRV and defining units for measuring biodiversity, to register transactions and enhance the credibility of existing and new (international) biodiversity markets, and to bring together buyers and sellers of credits.

Chapter 3

International environmental agreements for biodiversity conservation: a game-theoretic analysis[7]

This chapter contributes to the emerging literature on International Environmental Agreements (IEAs) with an analysis of key characteristics for biodiversity conservation. We study three features that are specific to an international conservation agreement: the existence of a natural upper bound of conservation in each country, the importance of local benefits, and the subadditivity of the global conservation function. We consider asymmetries in benefits and costs of conservation and, separately, in the upper bound of conservation in each country; and we examine the impacts of these features on coalition stability and on the effectiveness of biodiversity agreements. Results show that subadditivity in the global conservation function can lead to larger stable coalitions. The inclusion of a transfer scheme that might be implemented through, e.g. international trade of biodiversity credits, can have an impact on coalition composition, and can improve conservation outcomes and the size of stable coalitions in certain ranges of the parameter space.

[7] This chapter is based on Alvarado-Quesada, I., & Weikard, H.-P. (2015). International Environmental Agreements for Biodiversity Conservation: A Game-theoretic Analysis. Submitted.

3.1 Introduction

Management of global environmental resources is a difficult task because binding rules have to be agreed internationally but need to be implemented at the national level. A wide range of International Environmental Agreements (IEAs) have been negotiated to deal with particular environmental concerns. Recent studies on the economic analysis of the formation and stability of IEAs (e.g. Finus 2001, Rubio and Ulph 2006, Pavlova and de Zeeuw 2013) have drawn on seminal work of D'Aspremont et al. (1983), Carraro and Siniscalco (1993), Barrett (1994) and others. Most of this literature refers to the problem of global warming and IEAs for greenhouse gas (GHG) emissions control. Still, an analysis of the stability of an IEA for the case of biodiversity conservation remains a gap in the literature.

Different reasons lead us to study separately the case of biodiversity conservation from the conventional emission abatement model. First, biodiversity is unevenly distributed among countries. Every country has a different biodiversity endowment that is finite and, consequently, conservation efforts within a country are limited. Second, benefits from conservation are perceived differently at different scales (from local to global). Third, efforts of conservation should not be aggregated additively as it is common for emission abatement efforts. Two plots of the same size can be very diverse in terms of biodiversity richness (as measured by a species count), therefore they should not be valued as equal. Furthermore, counting protected species in each country as a measure of biodiversity conservation can lead to double counting of protected species globally. Additionally, in some cases spatial aspects such as habitat connectivity and minimum protected area size are considered as requirements to ensure species conservation, which imply that location of biodiversity does play an important role in the conservation game. Finally, the term biodiversity encompasses features inherent to public, club and private goods (Kaul 1999, TEEB 2010, Salles 2011). This combination of features represents a challenge for the achievement of efficient and sustainable management of biodiversity.

Given the specific aspects of biodiversity, the case of biodiversity conservation deserves some special attention. Specifically, our interest relies in exploring the particularities of the biodiversity case in the light of the IEA literature. In terms of modelling, there are at least three characteristics that differentiate an IEA for biodiversity conservation from the emission abatement case, and these characteristics are the focus of our this study.

The first feature is the existence of a natural upper bound of conservation in each country. For the case of GHGs, the maximum amount of emissions that a country can emit is not limited by nature but closely linked to its economic activities, i.e. land use, transportation and industry. However, for the case of biodiversity conservation the maximum amount of biodiversity that a country can preserve in its territory is limited. We assume that as any country approaches its maximum level of conservation of biodiversity, each additional unit preserved is more costly. To represent an unlimited increase in marginal costs of conservation we make use of hyperbolic cost functions in our model, instead of the often-used polynomial cost functions (e.g. quadratic functions) in models of climate agreements.

The second feature is the mismatch between the scales at which costs and benefits of biodiversity conservation take place. Costs of biodiversity conservation are local, but the benefits from conservation are perceived at different scales, e.g. local, regional and global. Climate impacts from GHG reductions are perceived globally regardless of the country where the reductions take place, whereas impacts of biodiversity conservation offer more immediate benefits at a local scale (Perrings and Gadgil 2003). In the climate change literature attention has been drawn towards the disaggregation of benefits into public (primary) and private (secondary, local or ancillary) benefits (Rübbelke 2006, Pittel and Rübbelke 2008, Longo et al. 2012, Pittel and Rübbelke 2012, Finus and Rübbelke 2013). Local benefits have been found to have a significant size, sometimes even exceeding global benefits (Pearce 2000). In the domain of biodiversity conservation there are also studies that refer to local or secondary benefits of conservation, such as Perrings and Gadgil (2003), Hein et al. (2006), Elmqvist (2012), Perrings and Halkos (2012), and Phelps et al. (2012). Winands et al. (2013) explicitly considered local benefits from biodiversity in a numerical model of an international biodiversity conservation agreement. We also consider local benefits of conservation in our model due to the important role they might play in the benefit functions of the players.

The third feature is the subadditivity of the global conservation function. Models of IEAs focus predominantly on emission abatement and usually define global abatement levels as the sum of the individual levels of abatement of all countries. For the case of biodiversity there is no standardised, generally accepted measurement to aggregate conservation levels. Therefore, we adopt the conceptual framework developed by Weitzman (1998). In this framework conservation measurements are associated with sets of protected species or ecosystems. A diversity measure can, in principle, be built on the dissimilarity between species in a set. While such information will usually not be available, the framework can be made operational using a species count as an approximate measure of biodiversity, as argued by Weikard (2002, Proposition 1). We adopt this idea and assume that all species have the same value. Since it is plausible that two countries protect some common species, we assume that global biodiversity conservation is a subadditive function of the aggregate of all countries' individual biodiversity conservation. An explicit aggregation model of biodiversity measurement across regions has been developed by Punt et al. (2012).

Finally, the assumption of symmetric countries frequently used in IEA models is too specific: both costs and benefits of biodiversity conservation vary greatly between countries. Many countries that are well endowed in terms of biodiversity richness are among the poorest in terms of income (Swanson and Groom 2012). Moreover, the natural upper bound of conservation also differs among countries.

Asymmetry and heterogeneity between countries has been a subject of study in the IEA literature. The assumption of symmetric countries has been relaxed by McGinty (2007) who illustrates by means of simulation exercises that IEAs with asymmetric countries can indeed achieve substantial gains under an appropriate transfer scheme. Pavlova and de Zeeuw (2013) study a model with two-sided asymmetry where countries differ in both emission-related benefits and environmental damage. They conclude that large coalitions can be stable under two-sided asymmetry, even when there are no transfers, but only if the asymmetries are sufficiently large. Furthermore, large coalitions perform better under asymmetry when transfers are allowed as compared to the symmetric case. Winands et al. (2013) focus on the role of asymmetries in the stability of biodiversity conservation agreements. Their numerical study reveals that in the absence of transfers, asymmetries among countries in terms of ecosystems and wealth reduce the size of a stable coalition as compared to a symmetric model specification. The inclusion of an optimal transfer scheme for the asymmetric case, however, does allow for a grand coalition in a four-player game.

In order to account for the effect of asymmetry on coalition stability, our model includes asymmetry in two ways. First, we deal with two-sided asymmetry: both benefits and costs of conservation are different between countries. Each country then belongs to one of four distinct country types: high benefits–high costs (BC), high benefits–low costs (Bc), low benefits–high costs (bC), and low benefits–low costs (bc). Additionally, we include asymmetry among countries in the natural upper bound of conservation for three different scenarios.

Our model makes a novel contribution to the literature on international biodiversity conservation by including i) a natural upper bound of conservation in each country combined with a hyperbolic cost function, ii) the inclusion of local benefits of conservation to represent the different scales at which biodiversity benefits are perceived, and iii) the subadditivity feature of the global conservation function. For a more comprehensive analysis, we study these characteristics under the assumption of both symmetric and asymmetric countries and we also allow for transfers, possibly implemented by an international market for biodiversity credits. We focus on these features to examine how they impact coalition stability and the scope for effective biodiversity agreements.

The chapter is organised as follows. In Section 3.2 we study the impact of i) hyperbolic costs, ii) local benefits of conservation, and iii) a subadditive function for the global conservation benefits on the size of stable coalitions. Section 3.3 combines these features but also considers asymmetric countries, and allows for the inclusion of transfers. Section 3.4 summarises the main findings and concludes.

3.2 IEA stability for biodiversity conservation with symmetric countries

3.2.1 The case of linear global and local benefits and hyperbolic cost functions

To develop a model of an IEA for biodiversity conservation, we consider a two-stage cartel game with *n* countries. In the first stage of the game countries choose whether or not to join the IEA. Those who join are the signatories and they form a coalition *S* composed of *s* signatories. Those remaining outside of the coalition (n - s) are the non-signatories or singletons. In the second stage of the game signatories coordinate their actions to maximise their collective net benefits. On the other hand, non-signatories maximise their own payoff function. A common payoff specification for country *i* where *b* and *c* are the benefit and cost parameters, respectively, is (see e.g. Barrett 1994):

$$\pi_i = bQ - \frac{c}{2} q_i^2 \qquad \qquad \forall\, i \notin S,$$

with $b > 0$ and $c > 0$. In this case Q represents global abatement and q_i is country *i*'s abatement level. Notice that abatement is usually assumed to be additive, i.e. $Q = \sum q_i$.

The first feature we include in our model of an IEA for biodiversity conservation is the specification of a hyperbolic cost function. This specification is crucial for the biodiversity case because countries have a given biodiversity endowment within their borders that can be protected. This endowment determines the upper bound of conservation $\overline{q}$ which is assumed to be equal for all countries (we relax this assumption later on). We use hyperbolic cost functions (instead of quadratic cost functions) to indicate that marginal costs of conservation increase without limits as a country approaches its maximum level of conservation, $\overline{q}$. One interpretation of the conservation level q_i is the number of species preserved within country *i*, as discussed in the introduction.

The second feature of our model is the inclusion of local benefits of biodiversity conservation in addition to the global benefits of conservation. Together with the benefits of global conservation (which is a public good), countries obtain local, secondary benefits from their biodiversity conservation. Improvements of recreational opportunities, better air quality, decrease in ambient temperature, and health improvements are some of the secondary benefits that can be perceived on a local scale as a result of conservation activities (Elmqvist 2012).

Finus and Rübbelke (2013) incorporate local benefits (or ancillary benefits as they call them) in the standard two-stage, cartel formation game of climate change. In one of their examples they consider a payoff function with linear local and global benefits and quadratic costs. To study the inclusion of local benefits of biodiversity conservation in our model, we use Finus and Rübbelke's (2013) model as a benchmark, but we use hyperbolic cost functions instead of the commonly-used quadratic cost functions as explained before. The payoff function for country *i* is:

$$\pi_i = bQ + \alpha b q_i - c\left(\frac{q_i}{\overline{q} - q_i}\right) \qquad \forall i \notin S. \tag{3.1}$$

In this first model variant, *Q* represents the sum of the number of species preserved in all countries $Q = \sum q_i$, q_i is the conservation level in country *i* (number of species preserved within country *i*), and $\overline{q}$ is any country's maximum level of conservation, where $\overline{q} > 0$. Also, *b* and *c* are the benefit and cost parameters respectively, $b > 0$, $c > 0$, and α is the parameter that measures the weight of benefits from local conservation, $\alpha \geq 0$. For this model the equilibrium conservation levels in the second stage of the game are:

$$q_i^*(s) = \left(\overline{q} - \delta \frac{1}{\sqrt{s+\alpha}}\right) \qquad i \in S, \tag{3.2}$$

$$q_i^*(s) = \left(\overline{q} - \delta \frac{1}{\sqrt{1+\alpha}}\right) \qquad i \notin S, \tag{3.3}$$

with $\delta \equiv \sqrt{\frac{c\overline{q}}{b}}$. Both signatories and singletons have dominant strategies for their conservation levels. Their optimal conservation levels depend on the benefit and cost parameters, on the upper bound of conservation and on the parameter of local benefits of conservation α.

Coalition stability

Given the conservation choices of the second stage, the payoff of a signatory in coalition *S* is denoted by $\pi_i^c(s)$ and the payoff of a singleton is denoted by $\pi_i^o(s)$. A subgame perfect equilibrium implies that given the choices at the second stage, signatories do not have an incentive to leave the coalition *S*, and singletons do not have an incentive to join the coalition *S*. We can say that coalition *S* is internally (IS) and externally (ES) stable if:

$$\text{IS:} \qquad \pi_i^c(s) \geq \pi_i^o(s-1) \qquad \forall i \in S, \tag{3.4}$$

$$\text{ES:} \qquad \pi_i^o(s) \geq \pi_i^c(s+1) \qquad \forall i \notin S. \tag{3.5}$$

We derive from our model the following internal and external stability conditions (see the appendix):

$$\text{IS:} \qquad \frac{(s-1)}{\sqrt{s-1+\alpha}} \geq 2\left(\sqrt{s+\alpha}-\sqrt{1+\alpha}\right) \qquad \forall i \in S, \tag{3.6}$$

$$\text{ES:} \qquad 2\left(\sqrt{s+1+\alpha}-\sqrt{1+\alpha}\right) \geq \frac{s}{\sqrt{s+\alpha}} \qquad \forall i \notin S. \tag{3.7}$$

Note that both conditions are independent of b and c. The conditions are dependent on the number of signatories and on the parameter α that weighs the local benefits of conservation. From conditions (3.6) and (3.7) it can be shown that the model with hyperbolic cost functions and local benefits of conservation leads to an equilibrium number of signatories of $s \leq 2$ for any $n \leq 2$, irrespective of the size of α.

Finus and Rübbelke's model (2013) with linear benefit functions (and local benefits included) and quadratic cost functions leads to a maximum number of signatories in a stable coalition of $s \leq 3$ for any $n \geq 2$ also irrespective of the size of α. They conclude that the inclusion of local benefits has no impact on the size of the stable coalition. We observe in our model that the smaller size of a stable coalition compared to a model with quadratic cost functions is a consequence of the use of hyperbolic cost functions, and not of inclusion of local benefits.

We confirm for our model that including local benefits does not alter the equilibrium size of the coalition. Results are not encouraging in this case since the size of stable coalitions in our model is smaller compared to Finus and Rübbelke's (2013) findings for quadratic cost functions. This indicates that forming effective biodiversity agreements could be even more difficult than international agreements for pollution abatement.

3.2.2 Subadditivity of the global conservation function

The third feature we include in our model is the subadditivity of the global conservation function. Conventional models of IEAs for climate change define global abatement as the sum of the individual abatement levels of all countries, $Q = \sum q_i$. We argue that for the case of biodiversity this specification is not convincing.

There is no official, standardised measurement for biodiversity conservation. Some of the common measurements to account for conservation are: size of protected areas, number of protected species, and number of ecosystem services. The measurement of biodiversity conservation by means of the size of protected areas presumes that each protected hectare offers the same level of biodiversity. This presumption seems to be too strong if we compare e.g. one hectare of protected forest in Indonesia with one hectare of protected dryland in Kenya.

In this study we make use of a species count as an approximate measurement of biodiversity (see Weikard 2002). We define global biodiversity conservation as the total number of species protected in the world. This definition assumes that all species have the same value. We consider this definition to be more appropriate for the analysis of the biodiversity problem because it allows us to exhibit the subadditivity aspect of conservation. Note that with such definition global conservation is not simply the sum of individual countries' conservation. We need to account for the fact that some species are jointly protected by two or more countries (Weitzman 1998). If, say, species z is protected in country i and in country j, then species z should be counted only once. Hence our suggestion is to use a subadditive global conservation function.

The conservation level q_i represents the total number of species that are protected in country i. Global biodiversity conservation G describes how the aggregate conservation maps to species protection. G must be smaller or equal to the sum of the individual conservation levels, $G \leq Q = \sum q_i$. In line with this definition, global biodiversity conservation G is a subadditive function of the sum of conservation levels of all countries, $Q = \sum q_i$. Generally, a function $f : A \rightarrow \boldsymbol{R}$ is subadditive if

$$f(x+y) \leq f(x) + f(y) \qquad \forall\, x, y \in A.$$

We specify global conservation in our model as a parabolic function of Q to represent its subadditive aspect. For simplicity we use a quadratic function for the specification of G because its properties allow us to illustrate its subadditive feature. However, other functional forms would also allow the inclusion of subadditivity in the model (e.g. a natural logarithmic function). The specification of global biodiversity conservation is:

$$G = \theta\left(-Q^2 + 2\bar{Q}Q\right) \tag{3.8}$$

where θ is a parameter for subadditivity and Q is the sum of the species preserved in all countries. We define $\bar{Q}$ as the sum of individual countries' species endowments $\bar{q}$. That is, for n symmetric countries, $\bar{Q} = n * \bar{q}$. The maximum value that G can take, $\bar{G}$, is obtained when $Q = \bar{Q}$ (see Figure 3.1).

Subadditivity requires concavity of G. Additionally, we require $\frac{1}{n\bar{Q}} \leq \theta \leq \frac{1}{2\bar{Q}}$ to ensure that the slope of the function is always less than 1 in its relevant part, $0 \leq Q \leq \bar{Q}$, and that the global species endowment must (weakly) exceed the species endowment in any individual country[8].

[8] Barrett (1994) also considers a quadratic specification for the benefit function. However, the quadratic function stated in his model is subadditive only for certain parameters but not in general. In our model we want to make sure that we include parameter constraints such that our quadratic function is always subadditive, i.e. $f'(q_i) < 1$.

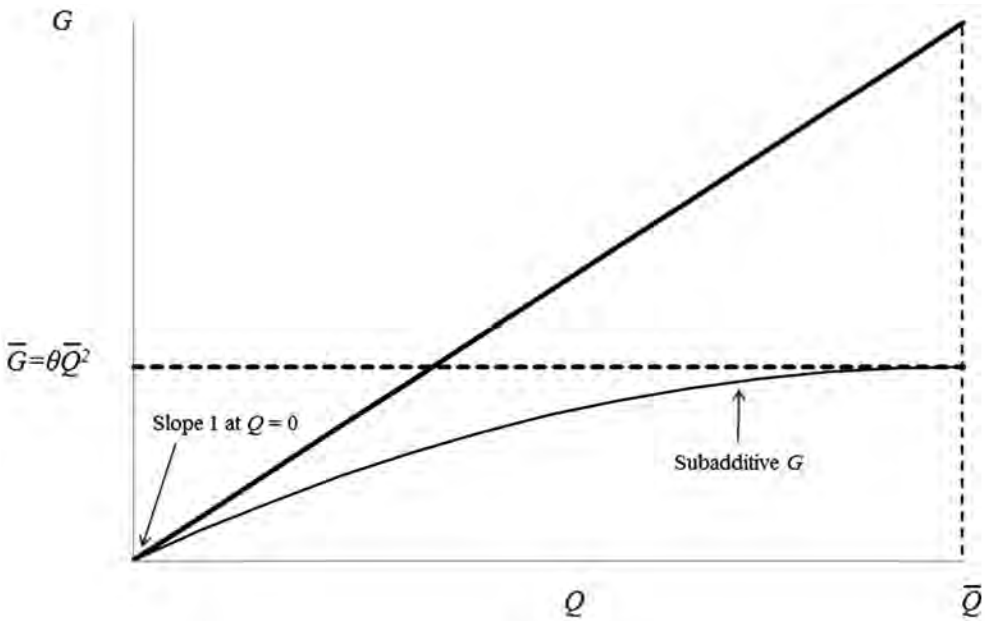


Figure 3.1 Subadditivity of the global biodiversity conservation function *G*

We then proceed to put together all three abovementioned features of an IEA for biodiversity conservation: hyperbolic cost functions, local benefits of conservation and subadditivity of the global conservation function *G*. We use the payoff function from equation (3.1) and we include equation (3.8) in the benefit function to analyse the impact of a subadditive function for global biodiversity conservation. The payoff function for country *i* with the subadditivity feature is:

$$\pi_i = b\left[\left(\frac{\overline{G}}{\bar{Q}^2}\right)\left(-Q^2 + 2\bar{Q}Q\right) + \alpha q_i\right] - c\left(\frac{q_i}{\bar{q} - q_i}\right) \qquad \forall\, i \notin S, \quad (3.9)$$

where we substitute θ in equation (3.8) by $\frac{\overline{G}}{\bar{Q}^2}$.

A full characterization of the reaction functions of this model and of its analytical solution when solved by computer software are too complex to make a useful interpretation, i.e. complex, extensive polynomials for $q_i^*(s)$. Alternatively, we perform a numerical simulation. We first determine arbitrary values for the parameters of the base model, and change the value of each of these parameters separately to study the impact of these changes on the size of stable coalitions.

We set the number of countries to $n = 12$ for all our model variants to facilitate our numerical appraisal. Since we consider symmetric countries, we maintain our assumption of $\bar{Q} = n * \bar{q}$. The parameter values set for the base case of the model with subadditivity in the global conservation function, local benefits and hyperbolic cost functions are: b = 1, c = 1000, α = 1, $\overline{q} = 1625$ and $\overline{G} = 6500$.

Numerical simulations reveal that for the base model the maximum size of a stable coalition is s^* = 2. There is a total of 66 stable coalitions, which means that all possible coalitions composed of any 2 countries are stable.

In order to evaluate the success of coalition formation in welfare terms, we make use of the relative welfare measure suggested by Eyckmans and Finus (2006) known as the 'closing the gap index' (CGI). The welfare CGI is defined as:

$$CGI^V = \frac{V^E - V^{NC}}{V^{FC} - V^{NC}} \tag{3.10}$$

where

V^E is the global payoff of the best coalition in equilibrium

V^{NC} is the global payoff when there is no cooperation

V^{FC} is the global payoff in the social optimum (full cooperation)

Notice that the index satisfies $0 \leq CGI^V \leq 1$.

For the base model with a stable coalition of 2 members, the value of the index is CGI^V = 0.072. This means that 7.2% of the potential gains from full cooperation can be reaped through the formation of a stable agreement with 2 members. We also calculate a CGI to express relative gain in terms of global conservation. The definition of the global biodiversity CGI is analogous to CGI^V:

$$CGI^G = \frac{G^E - G^{NC}}{G^{FC} - G^{NC}} \tag{3.11}$$

For the base model with a stable coalition of 2 members, the value of the index is CGI^G = 0.054. An agreement with a stable coalition of 2 members protects 5.4% of the global biodiversity that the grand coalition would preserve in addition to those preserved in the absence of an agreement.

In the remainder of this section we examine the impact of our model parameters on the size of the stable coalition. For the model with linear global and local benefits, and hyperbolic cost functions (Section 3.2.1) results are robust for any parameter change: the size of the largest stable coalition is always s^* = 2. For the model with subadditivity most changes in parameter values also result in a stable coalition of a maximum of 2 members. However, a larger stable coalition of size 3 is obtained when local benefits of conservation are larger in the base model (increase from α = 1 to α = 100). If the local benefit parameter increases to α = 1000, full cooperation is achieved (s^* = 12). Yet, this latter case is an instance of Barrett's (1994) 'paradox of cooperation': the gap between the aggregate payoff in the grand stable coalition and the all-singletons coalition structure is very small, therefore there is no real need for cooperation.

Our numerical analysis shows that the inclusion of subadditivity in the global biodiversity conservation function *G* allows for equilibrium coalitions larger than 2. In the same way that Barrett's (1994) quadratic-quadratic model shows larger coalitions to be stable, our three-feature model allows for coalitions larger than 2.

Table 3.1 summarises the payoff functions of the different models we have studied under the assumption of symmetric countries. We include as a point of reference other standard models from the IEA literature to compare functional forms and the maximum size of a stable coalition. In the next section we focus on coalition stability when we relax the assumption of symmetric countries.

Table 3.1. Equilibrium number of signatories for various payoff functions, as compared to some examples from main IEA literature [a/]

Model	Payoff function			Largest stable coalition (*s**)
	Benefits of biodiversity conservation		*Costs of biodiversity conservation*	
	Global benefits	Local benefits		
1. Barrett's (1994) model: linear benefit and quadratic cost functions	bQ		cq_i^2	3
2. Finus and Rübbelke's (2013) model: linear benefit and quadratic cost functions with the inclusion of ancillary (local) benefits	bQ	αbq_i	cq_i^2	3
3. Linear benefit and hyperbolic cost functions with the inclusion of local benefits	bQ	αbq_i	$c\left(\frac{q_i}{\bar{q}-q_i}\right)$	2
4. Barrett's (1994) model: quadratic-quadratic functions	$b\left(Q-\frac{Q^2}{2}\right)/N$		$\frac{cq_i^2}{2}$	$s^* \in [2, N]$
5. Three-feature model. Subadditivity of the global conservation function *G*, local benefits of biodiversity conservation and hyperbolic cost functions.	$b\left(\frac{\bar{G}}{\bar{Q}^2}\right)\left(-Q^2+2\bar{Q}Q\right)$	αbq_i	$c\left(\frac{q_i}{\bar{q}-q_i}\right)$	$s^* \in [2, N]$

a/ Models highlighted in grey are existing models in the IEA literature that we include for comparison. Models that are not highlighted are the actual models derived from this article.

3.3 IEA stability for biodiversity conservation with asymmetric countries

In this section we examine stability in our three-feature model of an IEA for biodiversity conservation when countries are asymmetric. We consider two types of asymmetries separately. First, we deal with double-sided asymmetry of countries in their benefits and costs of conservation. Then, we deal with countries with different upper bounds of conservation. For both types of asymmetries we analyse coalition stability without and with the inclusion of a transfer scheme.

3.3.1 Three-feature model with double-sided asymmetry

We start by assuming that countries have different benefits and costs of conservation. We introduce the scenario of two-sided asymmetry where each country belongs to one of four different categories:

i. Countries with high benefits of biodiversity conservation and high costs of biodiversity conservation (shorthand BC),
ii. Countries with high benefits of biodiversity conservation and low costs of biodiversity conservation (shorthand Bc),
iii. Countries with low benefits of biodiversity conservation and high costs of biodiversity conservation (shorthand bC),
iv. Countries with low benefits of biodiversity conservation and low costs of biodiversity conservation (shorthand bc).

An example of a BC country is Indonesia. According to the GEF benefits index for biodiversity (World Bank 2008), the relative biodiversity potential of Indonesia is very high; however conservation activities are relatively costly for the government. The Bc category reflects countries like Australia, where there are high benefits of preserving biodiversity and the cost of doing so is relatively low compared to other countries. An example of a bC country is Mali, where the biodiversity potential in terms of represented species and diversity of habitats is low but the costs of conservation activities are high. Finally, the bc category reflects countries like Finland where both biodiversity values and costs of biodiversity conservation are relatively low.

Again we consider a model with n = 12. This allows us to have 3 countries of each type. We consider the payoff function of equation (3.9) in Section 3.2.2. We maintain the parameter values of the base case, i.e. $\alpha = 1$, $\overline{q} = 1625$ and $\overline{G} = 6500$, with the exceptions of the benefit and cost parameters that vary depending on the country type; see Table 3.2.

Table 3.2. Value of the benefit and cost parameters for the different country types

Country category	Countries of this type in the model	Parameter values	
		b	c
BC	C1, C2, C3	100	10.000
Bc	C4, C5, C6	100	1000
bC	C7, C8, C9	1	10.000
bc	C10, C11, C12	1	1000

Numerical simulations reveal that the maximum size of a stable coalition for the model with double-sided asymmetry is equal to the model with symmetry: $s^* = 2$. However, for the asymmetric case there is a total of only 3 stable coalitions that are composed of any 2 countries with high benefits and low costs of conservation (Bc type). For the base model under asymmetry with a stable coalition of 2 members the CGI^V is 0.004. The formation of a coalition with 2 members achieves 0.4% of the potential gains from full cooperation. In terms of conservation outcomes for the stable coalition, we find that $CGI^G = 0.005$. Gains in payoff and global conservation are small.

We then examine the impact of the other model parameters on the size of the stable coalition. We performed a sensitivity analysis where we modified, one by one, the value of the parameters α, $\bar{q}$ and $\bar{G}$. Figure 3.2 shows the maximum size of an equilibrium coalition in the three-feature model under double-sided asymmetry for different parameter values.

According to the sensitivity analysis, results are robust for the changes in the parameter values of $\bar{q}$ and $\bar{G}$: the maximum size of the stable coalition of the three-feature model under double-sided asymmetry is $s^* = 2$. For changes in the parameter values of local benefits of conservation the results are different. A larger stable coalition of size 6 (consisting of all 3 members of the BC type and all 3 of the Bc type) is achieved when local benefits of conservation increase from $\alpha = 1$ to $\alpha = 100$.

From the analysis we observe that cooperation between countries in a two-sided asymmetry game is robust with respect to changes in i) the maximum level of global biodiversity conservation $\left(\bar{G}\right)$ and ii) the species endowment in each country $\left(\bar{q}\right)$, but is positively related to increases in local benefits of conservation (α). Even though higher local benefits of conservation translate into larger coalitions, they do not necessarily translate into more efficient IEAs. The reason is that the additional incentives to preserve that are due to high local benefits are irrespective of a country's participation in an IEA. Therefore, for this case, the gains in cooperation of an IEA with a large stable coalition are relatively low when compared to an IEA with a smaller stable coalition.

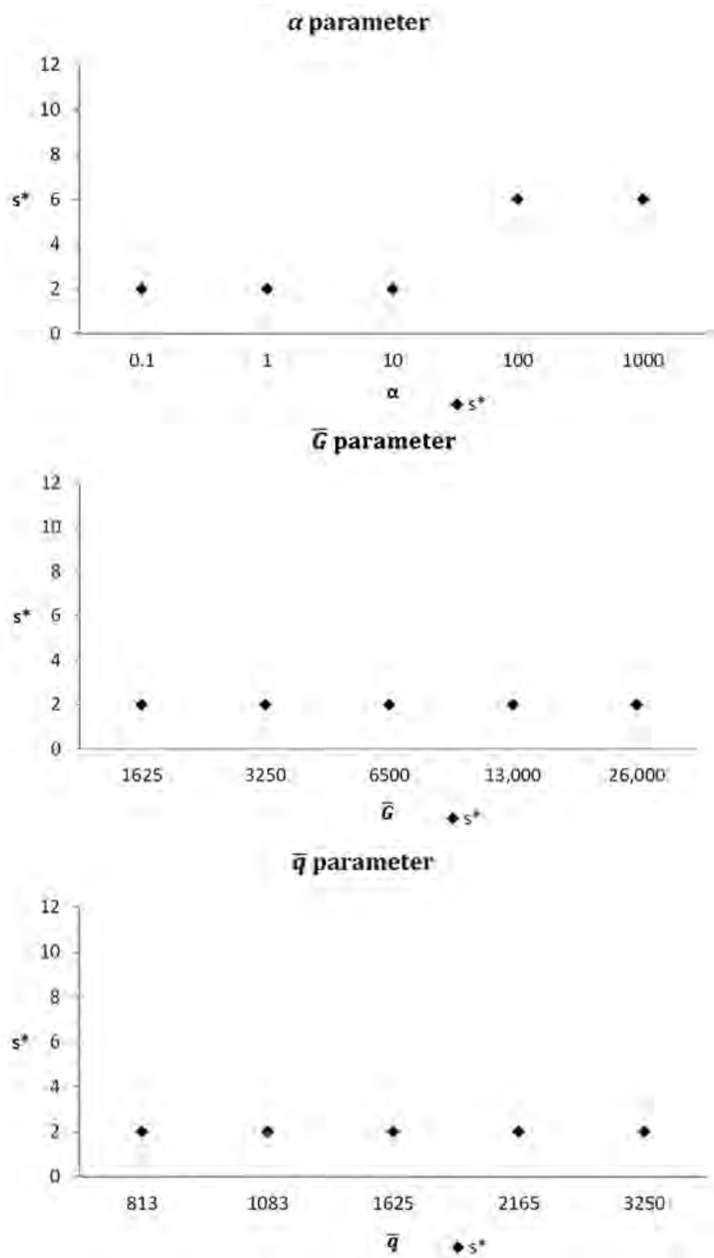


Figure 3.2 Size of stable coalitions for given changes in parameter values of the model with double-sided asymmetry. Only the value on the horizontal axis is modified. All other values remain constant as in the base case of the model.

Inclusion of transfers in the model with double sided-asymmetry

Transfers allow signatory countries of an agreement to 'buy international cooperation' (Fuentes-Albero and Rubio 2010). They can be used to set incentives to join the coalition, so that larger coalitions may satisfy the internal stability condition (Pavlova and de Zeeuw 2013). We apply an optimal sharing rule that guarantees that a coalition is internally stable when the coalition payoff (weakly) exceeds the sum of the outside option payoffs (Weikard 2009). We implement this sharing rule because it emphasises the importance of individual outside options. In the context of international agreements where membership is voluntary, it is natural to assume that each member should not be worse off than outside of the coalition.

In general transfers increase the chances for larger stable biodiversity agreements (Winands et al. 2013). One way to implement these transfers would be by means of an international market for biodiversity credits, as has been suggested by Alvarado-Quesada et al. (2014).

Table 3.3 compares the results obtained in our model with double-sided asymmetry without transfers with the results when the optimal transfer rule is applied.

Table 3.3. Comparative results: stability of coalitions and CGI in the three-feature model with double-sided asymmetry without and with transfers

	Double-sided asymmetry without transfers				*Double-sided asymmetry with transfers*			
Change in parameter	Number of stable coalitions	Largest stable coalition *(s*)*	CGI^V (%)	CGI^G (%)	Number of stable coalitions	Largest stable coalition *(s*)*	CGI^V (%)	CGI^G (%)
Base case [a/]	**3**	**2**	**0.4**	**0.5**	**21**	**7**	**87.3**	**80.8**
$\alpha = 100$	63	6	0.0	0.1	63	7	38.5	23.7
$\alpha = 0.5$	3	2	0.7	0.8	21	7	88.4	82.1
$\bar{q} = 2165$	3	2	0.3	0.4	21	7	86.6	79.9
$\bar{q} = 1083$	3	2	0.6	0.7	21	7	88.0	81.7
$\bar{G} = 13{,}000$	3	2	0.5	0.7	21	7	90.8	85.6
$\bar{G} = 3250$	3	2	0.3	0.4	21	7	81.4	72.9

a/ The parameter values for the base case of the model with subadditivity in the global conservation function, local benefits and hyperbolic cost functions with asymmetry in benefits and costs of conservation are: $n = 1$, $\alpha = 1$, $\bar{q} = 1625$ and $\bar{G} = 6500$. Values of b and c vary according to the country type as stated in Table 3.2.

From the results reported in Table 3.3 we find that the number of stable coalitions increases for all cases when transfers are allowed except for the case of $\alpha = 100$, where the number of stable coalitions remains constant (63).

Moreover, the size of the stable coalition systematically increases for all parameter changes when transfers are included. The largest stable coalitions when transfers are not allowed are composed of 2 members of the Bc type (relatively higher benefits of conservation). Only for α = 100 the largest coalition has 6 members: 3 members of the BC type and 3 of the Bc type. When transfers are allowed, however, we find stable coalitions of up to 7 members for all parameter changes. Notice that, although there are also stable coalitions of 2 members when transfers are allowed, the best coalitions in terms of payoff are composed of 7 members: 1 member of the BC type, 3 members of the bC type, and 3 members of the bc type. The composition of the larger stable structure varies with respect to the case without transfers: it has 1 member with relatively high benefits of conservation and 6 members with relatively low benefits of conservation. The reason why this coalition is more effective is that, despite being composed mainly by countries with relatively low benefits of conservation, global conservation is higher than in a stable coalition of 2 members with high benefits of conservation.

3.3.2 Three-feature model with asymmetry in the natural upper bound of conservation

Although useful for the analysis in Section 3.2, the assumption of an equal biodiversity endowment for all countries is not realistic. In this subsection we study three examples in which countries are asymmetric with respect to their natural upper bound of conservation $\overline{q}$. Even though we set different upper bounds of conservation for the countries in each scenario, the sum of all countries' species endowment is set equal to the symmetric case; i.e. $\overline{Q} = 19,500$ for all three scenarios.

Scenario I

For the first scenario we set 11 countries with the same value of $\overline{q}$, i.e. $\overline{q}$ = 1500, and 1 country (C12) with double the size of the natural upper bound of conservation than the rest of the countries, i.e. $\overline{q}$ = 3000.

Results show that the maximum size of a stable coalition is s^* = 2. All possible coalitions composed of 2 members are stable (66 coalitions in total). Eleven of these latter coalitions have the highest payoff and all of them include C12 as a member. C12 protects more than a half of its endowment $\overline{q}$. The other member of the coalition of 2 protects around 40% of its endowment, whereas the singletons protect less than a third of theirs.

Scenario II

For the second scenario we set 11 countries with the same value of $\overline{q}$, i.e. $\overline{q}$ = 1218, and 1 country (C12) with five times the size of the natural upper bound of conservation of the remaining countries, i.e. $\overline{q}$ = 6090.

Results show that also for this scenario the maximum size of a stable coalition is

$s^* = 2$. However, only 55 coalitions composed of 2 members are stable, and none of them includes C12 as a member. Signatory countries protect around one third of their biodiversity endowment, C12 as a singleton protects two thirds of its endowment, and the rest of the singletons protect less than 25% of their endowment.

Scenario III

For the third scenario we consider 10 countries with the same value of $\overline{q}$, i.e. $\overline{q} = 1392$, and 2 countries (C11 and C12) with two times the size of the natural upper bound of conservation than the remaining countries, i.e. $\overline{q} = 2784$.

Without transfers, the maximum size of a stable coalition for this scenario is also $s^* = 2$. There is a total of 65 stable coalitions, which are all possible coalitions of 2 members except the coalition formed by C11 and C12. Coalitions with the highest payoff are those with either C11 or C12 on them. For such cases, singletons with a lower $\overline{q}$ protect 30% of their endowment, and the signatory with the lower $\overline{q}$ protects around 37% of its endowment; whereas the singleton with the higher $\overline{q}$ (either C11 or C12) protects almost half of its endowment while the signatory with a higher endowment protects around 56% of its endowment. We observe that countries with a higher biodiversity endowment protect larger shares of their $\overline{q}$ than those with a lower biodiversity endowment; this is regardless of whether they act as signatories or as singletons. This is an artefact of the hyperbolic form of the cost functions we consider in our model: marginal costs of conservation become lower as the upper bound of conservation becomes higher.

Inclusion of transfers in the model with asymmetry in the natural upper bound of conservation

Table 3.4 shows a comparison between the results obtained in the three scenarios of asymmetric countries in their natural upper bound of conservation $\overline{q}$ without transfers and the model when the optimal transfer rule is applied.

Table 3.4. Comparative results: stability of coalitions and CGI in the three-feature model with asymmetry in the natural upper bound of conservation $\overline{q}$ without and with transfers

	Asymmetry in $\overline{q}$ without transfers				*Asymmetry in $\overline{q}$ with transfers*			
Scenario	Number of stable coalitions	Largest stable coalition *(s*)*	*CGI^V* (%)	*CGI^G* (%)	Number of stable coalitions	Largest stable coalition *(s*)*	*CGI^V* (%)	*CGI^G* (%)
Scenario I	66	2	8.4	6.3	66	2	8.4	6.3
Scenario II	55	2	6.5	4.9	66	2	10.4	7.8
Scenario III	65	2	8.1	6.1	66	2	9.5	7.1

We observe in Table 3.4 that the inclusion of transfers does not have an impact on the size of the largest stable coalition in any of the three scenarios. In particular, Scenario I does not show any variation in either the number of stable coalitions or the CGIs. In Scenario II however, the number of stable coalitions as well as the CGI indexes increase. For this scenario 10.4% of the potential gains from full cooperation can be reaped when allowing for transfers, as compared to a 6.5% without transfers. Also, when we allow for transfers, global conservation increases even though the size of the largest coalition remains unchanged: now 7.8% of global conservation under the grand coalition is protected. Finally, Scenario III also shows an increase in its CGIs, however to a lesser extent than Scenario II.

The outcomes of the scenarios suggest that under the inclusion of transfers, Scenario II shows the relatively largest potential gains from cooperation and conservation. Although the maximum size of a stable coalition remains equal with transfers, all countries are willing to individually transfer part of their gains to the country with the highest biodiversity endowment (C12) to make sure he joins an agreement of 2 members with them. Coalitions of 2 members with C12 on them have the best global payoff. We observe that trade is more effective if the countries involved in the flow of transfers are different.

3.4 Conclusions

In this study we develop a model of an IEA for biodiversity conservation that includes three characteristics that we consider key for the understanding of biodiversity agreements. We examine the stability of IEAs under the assumption of both symmetric and asymmetric countries, and without and with the inclusion of transfers. We derive important results that we discuss in this section.

In the first model variant under the assumption of symmetric countries, we include i) a hyperbolic cost function to represent the existence of a natural upper bound of conservation in each country, and ii) local benefits of conservation to deal with the perception of benefits of biodiversity conservation at different scales. For symmetric countries we obtain stable coalitions of a maximum of 2 signatories. These coalitions are smaller in comparison to the ones obtained in models of climate agreements that use quadratic cost functions. Furthermore, our result supports existing literature that states that the inclusion of local benefits has no impact on the size of a stable coalition.

The base case of the model variant that includes all three features of an IEA for biodiversity achieves a stable coalition of a maximum of 2 members. Larger stable coalitions can be achieved only for one parameter change: if local benefits of conservation are large relative to global benefits, the size of stable coalitions increases. Even full cooperation is possible, but in this case the gains from cooperation are minimal. We conclude that subadditivity in G allows for larger stable coalitions under certain parameter values even under the assumption of symmetric countries. However, in these cases countries have large local benefits and would adopt high protection levels even in the absence of cooperation.

When we include double-sided asymmetry in the three-feature model, the largest stable coalition for all parameter changes (with the exception of those related to α) remains equal to the symmetric model: $s^* = 2$. The difference lies in the composition of these coalitions: all three stable coalitions are composed of countries with relatively higher benefits and lower costs of conservation (Bc type). We also observe that, just as in the symmetric case, cooperation between countries is positively related to increases in α.

The inclusion of transfers in the model with double-sided asymmetry systematically increases the size of all stable coalitions under the different parameter changes. Under the setting of no transfers the maximum size of stable coalitions for different parameter values is either $s^* = 2$ or $s^* = 6$, whereas under the setting of transfers all coalitions are of size $s^* = 7$. An important outcome is that the composition of the largest stable coalition with the highest payoff differs from the case without transfers: it has 1 member with relatively high benefits of conservation (BC type) and 6 members with relatively low benefits of conservation (3 members of the bC type and 3 members of the bc type).

For the three scenarios with different natural upper bounds of conservation, we find that in the absence of transfers, the maximum size of the stable coalition is $s^* = 2$. For Scenario II, none of the stable coalitions include the country with a different upper bound of conservation, whereas for Scenario III only one of the two countries with a different upper bound can be part of a stable coalition. Regardless of their membership status (whereas they are signatories or singletons) countries with a higher upper bound of maximum biodiversity protect a larger share of their endowment than those with a lower bound because conservation becomes relatively cheaper. The inclusion of transfers does not have an impact on cooperation. Yet, the increase in potential gains from cooperation and conservation when transfers are allowed are highest for Scenario II. The creation of a transfer mechanism such as a global biodiversity market where conservation credits can be traded would allow countries with lower biodiversity endowments to transfer part of their gains from conservation to countries with higher biodiversity endowments to make sure they become part of a potential biodiversity agreement.

Our findings show that there is scope to achieve a higher degree of cooperation in a potential IEA for biodiversity conservation when subadditivity in the global conservation function is considered. Without transfers, larger stable coalitions can occur only for large parameter values of local benefits of conservation. However, this gain in coalition stability does not translate into larger gains in the aggregate payoff when compared to the all-singletons case.

Including transfers does allow for larger coalitions where gains in cooperation can be perceived. This result holds for one of the two types of asymmetries that we study in this chapter, namely the double-sided asymmetric case. The inclusion of an optimal transfer rule does not only lead to larger stable coalitions and higher potential gains of cooperation and conservation outcomes; it also results in a different composition of coalitions structures (in terms of country types) that those of the case without transfers.

Even when larger coalitions cannot be formed, as in the case of asymmetry in the natural upper bound of conservation, the flow of transfers allows for more effective coalitions in terms of global conservation. A global biodiversity market could be a good mechanism not only to increase global conservation, but to pinpoint where conservation is more effective and what characteristics potential members of an international biodiversity agreement should have. We expect that policy-makers can build upon the results presented here regarding gains of cooperation and coalition composition of biodiversity agreements when countries are asymmetric, and even more so when a transfer mechanism is implemented.

3.5 Appendix

Proof of coalition stability conditions for the model of linear global and local benefits and hyperbolic cost functions

If coalition member *j* leaves the coalition *S* in the first stage, the new conservation levels in the second stage would become:

$$q_i^*(s-1)=\left(\bar{q}-\delta\frac{1}{\sqrt{(s-1+\alpha)}}\right)\qquad i\in S_{-j}, \tag{3.2'}$$

$$q_i^*(s-1)=\left(\bar{q}-\delta\frac{1}{\sqrt{1+\alpha}}\right)\qquad i\notin S,\ i=j, \tag{3.3'}$$

with $\delta\equiv\sqrt{\frac{c\bar{q}}{b}}$.

The payoffs of country *j* in the second stage as a member of coalition *S* ($\pi_i^c(s)$) and as a singleton ($\pi_i^o(s-1)$), are the following:

$$\pi_i^c(s)=b\left[(n-s)\left(\bar{q}-\delta\frac{1}{\sqrt{1+\alpha}}\right)+s\left(\bar{q}-\delta\frac{1}{\sqrt{s+\alpha}}\right)\right]-\frac{c\left(\bar{q}-\delta\frac{1}{\sqrt{1+\alpha}}\right)}{\delta\frac{1}{\sqrt{s+\alpha}}} \tag{3.a.1}$$

$$\pi_i^o(s-1)=b\left[(n-s-1)\left(\bar{q}-\delta\frac{1}{\sqrt{1+\alpha}}\right)+(s-1)\left(\bar{q}-\delta\frac{1}{\sqrt{(s-1+\alpha)}}\right)\right]-\frac{c\left(\bar{q}-\delta\frac{1}{\sqrt{1+\alpha}}\right)}{\delta\frac{1}{\sqrt{1+\alpha}}} \tag{3.a.2}$$

The internal stability condition requires that for all *i* in *S*

$$\pi_i^c(s) - \pi_i^o(s-1) \geq 0 \tag{3.a.3}$$

or

$$\text{IS:} \qquad \frac{(s-1)}{\sqrt{s-1+\alpha}} \geq 2\left(\sqrt{s+\alpha} - \sqrt{1+\alpha}\right) \qquad \forall i \in S. \tag{3.6}$$

Now, if we assume that the singleton j joins the coalition *S* in the first stage, the first order conditions in the second stage becomes:

$$q_i^*(s+1) = \left(\bar{q} - \delta \frac{1}{\sqrt{(s+1+\alpha)}}\right) \qquad i \in S,\ i = j, \tag{3.2''}$$

$$q_i^*(s+1) = \left(\bar{q} - \delta \frac{1}{\sqrt{1+\alpha}}\right) \qquad i \notin S,\ i \neq j. \tag{3.3''}$$

The payoffs of country *j* in the second stage as an outsider ($\pi_i^o(s)$) and as a member of the extended coalition ($\pi_i^c(s+1)$) are respectively:

$$\pi_i^o(s) = b\left[(n-s)\left(\bar{q} - \delta \frac{1}{\sqrt{1+\alpha}}\right) + s\left(\bar{q} - \delta \frac{1}{\sqrt{s+\alpha}}\right)\right] - \frac{c\left(\bar{q} - \delta \frac{1}{\sqrt{1+\alpha}}\right)}{\delta \frac{1}{\sqrt{1+\alpha}}} \tag{3.a.4}$$

$$\pi_i^c(s+1) = b\left[(n-s+1)\left(\bar{q} - \delta \frac{1}{\sqrt{1+\alpha}}\right) + (s+1)\left(\bar{q} - \delta \frac{1}{\sqrt{(s+1+\alpha)}}\right)\right] - \frac{c\left(\bar{q} - \delta \frac{1}{\sqrt{(s+1+\alpha)}}\right)}{\delta \frac{1}{\sqrt{(s+1+\alpha)}}} \tag{3.a.5}$$

The external stability condition requires that for all *i* not in *S*

$$\pi_i^c(s+1) - \pi_i^o(s) \leq 0 \tag{3.a.6}$$

or

$$\text{ES:} \qquad 2\left(\sqrt{s+1+\alpha} - \sqrt{1+\alpha}\right) \geq \frac{s}{\sqrt{s+\alpha}} \qquad \forall i \notin S. \tag{3.7}$$

Chapter 4

International cooperation on biodiversity conservation when spatial structures matter[9]

This chapter considers the stability of International Environmental Agreements (IEAs) for biodiversity conservation with an explicit spatial structure. We develop a general framework that includes the location of countries and allows us to study the impact of distance between them on coalition stability. We exemplify the working of our model by analysing a circular spatial structure to study cooperation between countries that are identical in costs and benefits of conservation and in the size of their biodiversity endowment, but that differ in location. In the different spatial patterns that we assess, we obtain robust results for our specification of costs and benefits of conservation: stable coalitions have a maximum size of two members. For a scenario of equidistant countries the best global payoff is obtained when these coalitions are composed of neighbouring countries. For a scenario of increasing distances, they are composed of two countries with the smallest possible distance in the circular structure. We observe a 'remoteness effect', i.e. some coalitions of two members are unstable when one of the signatory countries is remote with respect to its other coalition member and to the singletons. This cannot be changed regardless of the possibility of transfers. We find that global payoffs associated with coalitions of two members are the highest, not only when the distance between members is smallest, but also when the singletons are located closer to each other and to the coalition. In such a situation, positive spillovers are maximised.

[9] This chapter is based on Alvarado-Quesada, I., & Weikard, H.-P. (2015). International cooperation on biodiversity conservation when spatial structures matter. Paper presented at the 17th Annual BIOECON Conference, Kings College, Cambridge, UK, 13-15 September 2015.

4.1 Introduction

Benefits from biodiversity differ among countries (World Bank 2008). Initial biodiversity endowments, costs of conservation, and dependence of a country's economy on its natural resources are some of the factors that determine countries' local benefits of biodiversity conservation. However, biodiversity benefits are not only perceived at a local scale (i.e. at a country level), but also at a regional and global scale.

Many of the services provided by biodiversity display features of global public goods. Due to this public good nature, the consequences of biodiversity loss are not confined to nation states. An overall rapid decline of biodiversity affects all countries regardless of the uneven distribution of biodiversity and of the location where the decline takes place. For this reason an important option for biodiversity management is the implementation of international environmental agreements (IEAs) for biodiversity conservation. International agreements are not easy to achieve: countries must not only sign an agreement, but also ratify and enforce it to make it effective (Wangler et al. 2013).

IEAs have been applied over the past decades as useful tools to target problems such as greenhouse gas emission control, use of transboundary watercourses, or biodiversity conservation (e.g. CBD, CITES, CMS, Ramsar Convention). A broad game-theoretic literature that studies IEAs as games of coalition formation has emerged since the seminal analysis of Hoel (1992), Carraro and Siniscalco (1993), and Barrett (1994); see surveys by Finus (2008) and Benchekroun and Long (2012). However, only a few studies explicitly address IEAs for conservation, such as Barrett (1994a), Punt et al. (2012), Ansink and Bouma (2013), Winands et al. (2013) and Alvarado-Quesada and Weikard (2015). Yet, the impact of the spatial dimension of agreements on their stability and effectiveness has not been studied thoroughly, considering that only few studies on international economic theory account for spatial aspects of countries (Egger and Egger 2010).

The aim of our study is to analyse the impact of spatial structure on the stability of an IEA for biodiversity conservation. To do so we develop a model that accounts not only for local and global biodiversity benefits – which are considered to be independent of spatial structure –, but also for regional biodiversity benefits that we model as distance dependent, positive spillovers. These spillovers are assumed to be stronger if countries are closer to each other. Although the model framework that we adopt is capable of capturing the impact of any spatial structure on coalition stability, we focus on one particular setting: a circular spatial structure. This is the simplest structure that is still rich enough to examine cooperation between neighbouring countries.

Location of and distance between countries are key variables in the decision to cooperate in an IEA (see Davies and Naughton 2013). To examine the continuous, circular structure that we want to consider, we build upon Salop's (1979) industry location model. In Salop's model, players – in our case countries – are located on the circumference of a circle. Within this spatial structure each country has two directly neighbouring countries. Hence, our model differs from standard spatial models of transboundary pollution that study location on a straight line where countries located at the edges have only one neighbour (e.g. Hotelling 1929). Unlike Gengenbach et al. (2010) or Wang (2011) who consider unidirectional flow of pollutants in river structures, we do not adopt this assumption. In our model, distance matters in any direction (on the circle). A novelty of our study is that it combines a general location model with a game of IEA formation.

Our model is general with respect to the notion of distance. Conventionally one would think of distance as geographical distance (measured in kilometres). But other notions of distance can be employed as well. Hence, a second novelty of our study is that we introduce the notion of distance as dissimilarity. Inspired by Weitzman's (1992, 1998) approach to measure biodiversity we introduce the notion of Ecosystem Dissimilarity (ED). The ED between two countries is a measure that captures how different the sets of species hosted by these countries are. For our purpose of exploring the stability of conservation agreements, geographical distance may be less important than the dissimilarity of the sets of species that two countries host. According to this notion of distance, two countries are closer when they have more species in common.

For the analysis of our spatial model we follow a maximisation approach of the net benefits of biodiversity conservation. We assume that each country carries out its conservation activities that are costly. We consider direct local benefits of conservation from the species present in a country, and global benefits from all species regardless of where they occur. The key feature of our model is, however, that we also incorporate benefits from other countries' conservation that depend on distance. These we label 'regional benefits'. With ED as the notion of distance, our model captures the idea that conservation efforts are strategic substitutes. A country benefits more from close neighbours' conservation efforts than from other countries' efforts. Close neighbours have similar ecosystems, i.e. neighbouring countries have a large set of species in common. In this setting, conservation efforts of one country can be supportive to conservation goals of another country that hosts a similar set of species. Although we adopt this particular specification and interpretation of distance, our approach is general and would work for arbitrary specifications of distances. The adoption of ED or of a circular structure is not essential to our approach.

Our model fills a gap in the literature by describing the role of location of countries and the distance between them in the design of IEAs for biodiversity conservation. The contribution of our research is the introduction of a general framework that allows to study the role of spatial structure on coalition formation. Ultimately, we intend to shed light on the potential of and the design principles for regional cooperation in multinational conservation programmes.

The chapter is organised as follows. Section 4.2 describes the structure of the model and provides analytical results. In Section 4.3 we explain our numerical simulations and we describe the scenarios of two different spatial patterns. In Section 4.4 we present the results without and with the inclusion of a transfer scheme. We also introduce an index of proximity to shed some light on the behaviour of coalitions when countries are grouped in clusters along the circumference of the circle. Section 4.5 concludes.

4.2 A spatial model for biodiversity conservation

4.2.1 Ecosystem Dissimilarity (ED)

Different concepts of distance are applied in the field of biology to pursue basic classification tasks (see Deza and Deza 2009). Our definition of distance is inspired by Weitzman's (1992) measure of diversity and his species/library model of diversity (Weitzman 1998). In a nutshell, Weitzman's measure of biodiversity assesses how many different genes are in a collection of species or, metaphorically, how many different books are in a collection of libraries. In our context it is convenient to characterise a country by the set of species it hosts and a simple species count could serve as a measure of biodiversity (Weikard 2002, Proposition 1). To define our concept of distance as dissimilarity, we consider two sets of species (representing countries), I and J. The simplest measure of dissimilarity between sets I and J, denoted $d_{I,J}$ is the number of elements that I contains and J does not. In other words, $d_{I,J} = I - I \cap J$. Under this definition, $d_{I,J} = 0$ if $I \subseteq J$, and $d_{I,J}$ reaches a maximum if there are no common elements; see Enflo (2012) for an extensive discussion of diversity concepts based on distances.

Therefore, in our model the distance between countries reflects the ecological interpretation of the dissimilarity of sets of species or ecosystems, and is represented by the location of the countries on the circumference of the circular structure that we consider in our application.

4.2.2 Circular structure of the game

Several environmental economic models dealing with spatial configurations make use of the industrial organisation literature (Goeschl and Igliori 2004, Albers et al. 2008, Ando and Shah 2010, and Punt et al. 2012). Hotelling's (1929) location model is one of the main models of industry location. In this model, consumers of a particular good are uniformly distributed along a finite line and there are two firms producing the same good. The underlying assumption is that consumers will buy the good from the firm located most closely. In equilibrium, both firms choose to locate at the mid-point of the line and capture half of the market.

For our research we build upon Salop's (1979) industry location model, which describes a product space of the industry in the form of the circumference of a circle. This assumption avoids the edges that drive Hotelling's result. In our model we employ the circular spatial structure where all countries have two neighbours, and we study a cartel formation game in this structure.

4.2.3 Model definition

We start by considering a set N composed of $n \geq 3$ countries. We follow a maximisation approach of the net benefits of biodiversity conservation. Costs of biodiversity conservation are related to the biodiversity endowment of each country. Benefits of conservation, however, can be perceived at different scales, i.e. at local, regional and global scales. We describe these benefits below.

Local benefits of biodiversity conservation

Countries obtain benefits from conservation of their biodiversity endowment. Local benefits of conservation are defined as:

$$L_i = \lambda_i \cdot b_i \qquad \forall i \in N, \tag{4.1}$$

where

L_i are benefits of local biodiversity conservation for country i

λ_i is the parameter for benefits of local biodiversity conservation for country i, $\lambda_i > 0$

b_i is the biodiversity conservation level in country i as measured by a species count

Regional benefits of biodiversity conservation

In addition to local benefits of conservation, countries can also benefit from the conservation of other countries' biodiversity endowments. We assume that country i can benefit when other countries adopt conservation measures that protect species that also belong to country *i*'s species endowment. This benefit can be perceived as an option value for country *i*: if a species is extinct (or threatened) in country *i*, it could be reintroduced in country *i* if it is conserved in country *j*. We refer to these benefits as regional ones because it is plausible to assume that countries within a geographic area with similar features are likely to share more species and thus have lower ecosystem dissimilarity (ED) between them.

Regional benefits are the ones that encompass the circular spatial structure of the game. In order to include the ED in the payoff function of our model, we define a parameter that weighs the relevance of distance (or dissimilarity in our case) in the regional benefits of conservation. The impact of biodiversity in country *j*, b_j, on the regional benefits of conservation of country *i* will be determined by a weight parameter $\omega_{i,j}$. The weight assigned to other countries' conservation levels is a decreasing function of their dissimilarity $d_{I,J}$. Regional benefits of conservation are defined as:

$$R_i = \rho_i \left(\sum_{j \in N_{-i}} \omega_{i,j} \cdot b_j \right) \qquad \forall\, i, j \in N, \tag{4.2}$$

where

R_i are benefits of regional biodiversity conservation for country *i*

ρ_i is the parameter for benefits of regional biodiversity conservation for country *i*, $\rho_i > 0$

$\omega_{i,j}$ is the distance parameter that weighs biodiversity of any other country $j \epsilon N_{-i}$ in the regional benefits of conservation of country *i*, where $0 \leq \omega_{i,j} \leq 1$

Notice that the ED is inversely related to the distance weighted parameter $\omega_{i,j}$: the smaller the ED between any two countries, the closer they are and hence, the larger the weight $\omega_{i,j}$ and therefore the impact of *j*'s conservation level on *i*'s regional benefits. The complete spatial structure of our model is described by the distance weighted parameter matrix (see Appendix).

Global benefits of biodiversity conservation

Global benefits of conservation are derived from the notion of biodiversity as a public good. The preservation of species somewhere, regardless of location, generates benefits for all countries. We define global benefits of conservation as:

$$G_i = \gamma_i \cdot M \qquad \forall\, i \in N, \tag{4.3}$$

where

G_i are benefits of global biodiversity conservation perceived by country *i*

γ_i is the parameter for benefits of global biodiversity conservation for country *i*, $\gamma_i > 0$

M is global biodiversity conservation, defined as the number of preserved species in the world

Global biodiversity conservation M describes the total number of preserved species as an effect of aggregate conservation measures. If, for example, species z is protected in several countries, it would be counted several times in the sum of individual conservation levels $\Sigma b_j = B$. Therefore, we cannot use B as our measure of global biodiversity because we would run into an over counting error. Instead, we define global biodiversity conservation as a subadditive function of the sum of total biodiversity endowments, where M $\leq B \equiv \Sigma b_{j.}$ We define global biodiversity conservation as the following parabolic, subadditive function of B:

$$M = \delta(-B^2 + 2\bar{B}\,B) \tag{4.4}$$

where

δ is the parameter for subadditivity

B is the sum of individual conservation levels of all countries in N (measured by a species count)

$\bar{B}$ is the sum of individual countries' species endowments $\bar{b}$; i.e. for n symmetric countries, $\bar{B} = n * \bar{b}$

Note that the maximum value that global biodiversity conservation can take, $\bar{M}$, is obtained when $B = \bar{B}$.

For the assumption of subadditivity of the global conservation function to hold, we require M to be concave. We also require that $\frac{1}{n\bar{B}} \leq \delta \leq \frac{1}{2\bar{B}}$ in order to guarantee that the slope of the function is smaller than 1 in its relevant part $0 \leq B \leq \bar{B}$; and that the global species endowment (weakly) exceeds the species endowment in any individual country (see Alvarado-Quesada and Weikard 2015).

If we substitute equation (4.4) in the global benefits of biodiversity conservation in equation (4.3) we obtain:

$$G_i = \gamma_i \cdot \left[\delta\left(-B^2 + 2\bar{B}B\right)\right] \qquad \forall\, i \in N. \tag{4.3'}$$

To facilitate our numerical appraisal we normalise with respect to the benefit parameter of global biodiversity conservation γ_i. Therefore we consider $\gamma_i = \gamma = 1$ for all calculations of this study. Also, we assume that $\delta = \frac{\bar{M}}{\bar{B}^2}$.

Now that we have described the specification of the different benefits of biodiversity conservation, we can develop our model of an IEA for biodiversity conservation with a defined spatial structure. We consider a two-stage cartel game with n countries. In the first stage of the game countries choose to join or not the IEA. In the second stage of the game, those countries that join the agreement – the signatories – coordinate their actions to maximise the collective net benefits of biodiversity conservation. The set composed of signatory countries is defined as *S*. Countries that remain outside of the agreement – the outsiders – maximise their individual payoff functions.

We define the payoff function for country *i* as follows:

$$\pi_i = \lambda_i b_i + \rho_i \left(\sum_{j \in N_{-i}} \omega_{i,j} \cdot b_j \right) + \gamma\delta\left(-B^2 + 2\bar{B}B\right) - c_i\left(b_i - a_i\right)^2 \qquad \forall\, i \notin S. \tag{4.5}$$

We assume a quadratic specification for the local costs of conservation where c_i is the parameter of costs for local biodiversity conservation, $c_i > 0$; and a_i is the number of species preserved in each country at no cost, $a_i \geq 0$.

Substituting $B = \Sigma b_j$ in equation (4.5) and splitting the sum we obtain:

$$\pi_i = \lambda_i b_i + \rho_i \left(\sum_{j \in N_{-i}} \omega_{i,j} \cdot b_j \right) + \gamma\delta\left[-\left(b_i + \sum_{j \in N_{-i}} b_j \right)^2 + 2\bar{B}\left(b_i + \sum_{j \in N_{-i}} b_j \right) \right] - c_i\left(b_i - a_i\right)^2 \qquad \forall\, i \notin S. \tag{4.5'}$$

Each country $i = 1, \ldots, n$ maximises its total payoff function with respect to its own conservation level b_i. The equilibrium biodiversity levels are given by:

$$b_i^* = \frac{\lambda_i - 2\gamma\delta\left(B_{-i} - \bar{B}\right) + 2c_i a_i}{2\left(c_i + \gamma\delta\right)} \qquad \forall i \notin S. \tag{4.6}$$

Also, signatories maximise the coalition benefits, leading to the following equilibrium biodiversity levels:

$$b_i^* = \frac{\lambda_i + \sum_{j \in S_{-i}} \rho_j \cdot \omega_{j,i} - 2\sum_{j \in S}(\gamma\delta) \cdot (B_{-i} - \bar{B}) + 2c_i a_i}{2\left[c_i + \sum_{j \in S}(\gamma\delta)\right]} \qquad \forall i \in S, \tag{4.7}$$

where *S* is the set of signatory countries.

Coalition stability

A subgame perfect equilibrium implies that, given the choices in the second stage of the game, i) signatories do not have an incentive to leave coalition *S*, and ii) the outsiders have no incentive to join coalition *S*. A coalition *S* is said to be internally (IS) and externally stable (ES) when:

$$\text{IS:} \qquad \pi_i^*(S) \geq \pi_i^*\left(S \setminus \{i\}\right) \qquad \forall\, i \in S, \tag{4.8}$$

and

$$\text{ES:} \qquad \pi_j^*(S) \geq \pi_j^*\left(S \cup \{j\}\right) \qquad \forall\, j \notin S. \tag{4.9}$$

where $\pi_i^*(S)$ is the payoff of a signatory and $\pi_j^*(S)$ is the payoff of an outsider when coalition S is formed.

For our analysis, we want to study the impact of location and distance between countries on stability and conservation effectiveness of an agreement. In order to do so, we consider n = 12 countries with identical characteristics (i.e. identical cost and benefit parameters and size of biodiversity endowments) but with different locations throughout the circumference of the circle that we use as our spatial structure. In the following section we perform a numerical analysis for different spatial country patterns in our circular structure.

4.3 Numerical analysis

4.3.1 Description of the base model

In order to obtain stability results for our spatial model, we perform a numerical simulation for two scenarios that illustrate different location structures. First, we set a base model where we determine arbitrary values for the different parameters at stake. Second, we proceed to change the value of each of these parameters separately to study the impact of these changes on the size and composition of the stable coalitions. Table 4.1 shows the parameter values for the base model of the two scenarios under analysis.

Table 4.1. Parameter values for the base model

Parameter	Value
Maximum global biodiversity conservation $(\bar{M})$	1
Sum of individual countries' species endowments $(\bar{B})$	6
Benefits of local biodiversity conservation (λ)	0.5
Benefits of regional biodiversity conservation (ρ)	0.05
Weighted value of biodiversity of other countries ($\omega_{i,j}$)	See Appendix
Benefits of global biodiversity conservation (γ)	1
Costs of local biodiversity conservation (c)	1
Number of species preserved per country at no cost (a)	0.05

We consider 12 countries that are symmetric in benefits, costs, and in the size of their biodiversity endowment. The global level of biodiversity is biodiversity is $\bar{M} = 1$. Given the symmetry assumption, each country's biodiversity endowment is $\bar{b} = 0.5$. We assume that the total number of species protected at no cost is 10% of the biodiversity endowment in each country $\bar{b}$, i.e. $a = 0.05$. Also, regional benefits of conservation in the base model are 10% of the local benefits of conservation, i.e. $\rho = 0.05$.

4.3.2 Description of scenarios

As mentioned in the previous section, countries are symmetric in all model parameters except for the weighted parameter $\omega_{i,j}$. This parameter weighs the distance between any two countries in the circumference of the circular spatial structure. In our study we analyse different types of spatial patterns of countries throughout the circumference of the circular structure. The value assigned to the distance weighted parameter per country depends on the scenario at stake. The distance weighted parameter matrices for the scenarios can be found in the Appendix. Below we describe two of these scenarios in detail.

Scenario I: Equidistant countries

The first scenario under analysis considers equidistant countries throughout the circumference of the circular structure. The ED between any two neighbouring countries on the circumference of the circle is of equal size; i.e. $d_{1,2} = d_{2,3} = d_{3,4}$, and so forth:

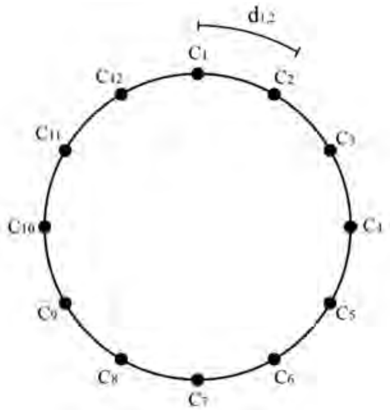


Figure 4.1 Circular structure with equidistant countries

Our assumption implies that the dissimilarity between any two neighbouring sets of species is of the same magnitude.

Scenario II: 'Increasing distances' between countries

For this second scenario we assume that the distance between countries will increase as we move along the spatial plane towards the furthest country possible. For example, if the ED between country 1 (C1) and country 2 (C2) is $d_{1,2} = x$, then the ED between C2 and C3 is $d_{2,3} = 2x$, the ED between C3 and C4 is $d_{3,4} = 3x$, and so forth. The largest distance is that between any two countries that are the farthest away possible from each other: i.e. the ED between C1 and C7, $d_{1,7} = 21x$:

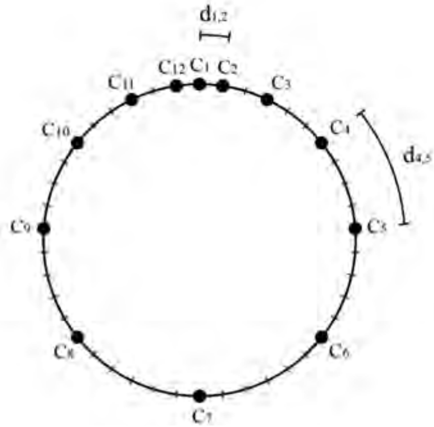


Figure 4.2 Circular structure with increasing distances between countries

Note that since we assume symmetry of countries with respect to the size of their species endowment, our definition of ED is symmetric: $d_{I,J} = I - (I \cap J) = J - (J \cap I) = d_{J,I}$.

4.3.3 Welfare analysis: inclusion of the 'closing the gap index' (CGI)

We are interested in making our numerical appraisal comparable not only with respect to results of parameter changes within scenarios, but also across scenarios. In order to do so, we incorporate in our analysis an effectiveness measure called 'closing the gap index' or CGI (see Eyckmans and Finus 2006). The welfare CGI is defined as:

$$CGI^V = \frac{V^E - V^{NC}}{V^{FC} - V^{NC}}, \tag{4.10}$$

where

V^E is the aggregate payoff of the best coalition in equilibrium

V^{NC} is the aggregate payoff when there is no cooperation

V^{FC} is the aggregate payoff in the social optimum (full cooperation)

Notice that the index satisfies $0 \leq CGI^V \leq 1$.

In order to compare the success of the equilibrium coalition in terms of global biodiversity conservation we also make use of a global conservation index CGI^M. It is constructed analogous to the CGI^V:

$$CGI^M = \frac{M^E - M^{NC}}{M^{FC} - M^{NC}}. \tag{4.11}$$

4.4 Results

After specifying the base model and the two different scenarios of our study, we proceed with the coalition stability analysis. We first calculate results for the base case of each scenario. Then, we change the value of each of the parameters listed in Table 4.1 separately to study the impact of these changes on i) the size and composition of the stable coalitions, ii) global biodiversity conservation and iii) the global payoff from biodiversity conservation.

Table 4.2 presents the results of the numerical analysis for the scenario with equidistant countries. Results show that the maximum size of a stable coalition for our model is $s^* = 2$. For all parameter changes, all possible coalitions of two countries that can be formed in a twelve-player game (i.e. 66 coalitions) are stable. Yet, only those coalitions composed of two neighbouring countries (e.g. C1-C2, C2-C3, etc.) have the best payoff.

For the base model, the value of the CGI^V is 10.8%. This implies that almost 11% of the potential gains from full cooperation can be obtained by means of the formation of a stable agreement of two members. In terms of conservation, the value of the index for the base model is CGI^M = 10.2%. An agreement with a stable coalition of two members will achieve 10% of the total conservation that the grand coalition would achieve in addition to those preserved in the absence of an agreement.

The largest potential gains from cooperation can be obtained when the cost parameter decreases in 10% (from $c = 1$ to $c = 0.9$): 11.8% of the potential gains from the scenario of full cooperation can be reaped when a coalition of two countries is formed. In terms of conservation, the largest gains can be obtained with two separate parameter changes: when the local benefit parameter increases by 20% (from $\lambda = 0.5$ to $\lambda = 0.6$) and when the number of species protected at no cost doubles (from $a = 0.05$ to $a = 0.1$), a total of 11.8% of global conservation under the grand coalition is preserved.

Table 4.3 shows the results for the scenario with increasing distances between countries. As in the previous scenario, the maximum size of a stable coalition for this model is $s^* = 2$. For all parameter changes, there are two coalitions that generate the highest global payoff. These are composed by the two possible combinations of countries that have the smallest ED in the circular structure, i.e. C1-C2 and C1-C12.

In the base model for the scenario with increasing distances between countries, 11.8% of the potential gains from full cooperation can be obtained by the formation of a stable agreement of two members and 11.2% of global conservation under the grand coalition is reached (CGI^V = 11.8% and CGI^M = 11.2%, respectively). Finally, from all parameter changes, both the largest potential gains from cooperation and the largest relative gains in conservation are obtained from a 10% decrease of the cost parameter (CGI^V = 13.3% and CGI^M = 12.7%, respectively).

Table 4.2. Coalition stability and CGI for equidistant countries [a/]

Change in parameter value	Size of stable coalitions (s^*)	Number of stable coalitions (out of a total of 4095)	Global biodiversity under best coalition (number of preserved species and CGI^M in %)	Global payoff under best coalition (absolute value and CGI^V in %)	Remarks
1. BASE MODEL	**2**	**66**	**0.92** **CGI^M= 10.2**	**12.91** **CGI^V= 10.8**	**All possible coalitions of two members are stable. Twelve of these have the best payoff. These are composed of two neighbouring countries: C1-C2, C2-C3, ..., C12-C1.**
2. Higher local benefit parameter $\lambda = 0.6$	2	66	0.96 CGI^M= 11.8	13.84 CGI^V= 11.6	Same as base case.
3. Lower local benefit parameter $\lambda = 0.4$	2	66	0.87 CGI^M= 10.1	11.92 CGI^V= 10.5	Same as base case.
4. Higher regional benefit parameter $\rho = 0.075$	2	66	0.92 CGI^M= 11.1	13.37 CGI^V= 11.1	Same as base case.
5. Lower regional benefit parameter $\rho = 0.025$	2	66	0.92 CGI^M= 9.3	12.46 CGI^V= 10.2	Same as base case.
6. Higher cost parameter $c = 1.1$	2	66	0.89 CGI^M= 9.6	12.48 CGI^V= 10.8	Same as base case.
7. Lower cost parameter $c = 0.9$	2	66	0.94 CGI^M= 11.1	13.36 CGI^V= 11.8	Same as base case.
8. Higher number of species protected at no cost $a = 0.10$	2	66	0.96 CGI^M= 11.8	13.75 CGI^V= 11.0	Same as base case.
9. Zero species protected at no cost $a = 0$	2	66	0.87 CGI^M= 10.1	11.94 CGI^V= 10.7	Same as base case.

a/ Only one parameter is changed at a time. All parameter changes are to be compared with the base model parameters in Table 4.1. The values of the parameter $\omega_{i,j}$ are those showed in the distance weighted parameter matrix for equidistant countries in Table A1 of the Appendix.

Table 4.3. Coalition stability and CGI for increasing distances between countries [a/]

Change in parameter value	**Size of stable coalitions** *(s*)*	**Number of stable coalitions** (out of a total of 4095)	**Global biodiversity under best coalition** (number of preserved species and CGI^M in %)	**Global payoff under best coalition** (absolute value and CGI^V in %)	**Remarks**
1. BASE MODEL	**2**	**63**	**0.92** **CGI^M = 11.2**	**13.14** **CGI^V = 11.8**	**From all possible coalitions of two members (66), 63 are stable. From these, two coalitions have the best payoff. They consist of two countries with the smallest possible ED in the circular structure, namely C1-C2 and C1-C12. Those that are not stable are composed of two countries with the largest possible ED in the circular structure, namely C1-C7, C4-C8 and C6-C10.**
2. Higher local benefit parameter $\lambda = 0.6$	2	64	0.96 CGI^M = 11.8	14.09 CGI^V = 12.7	From all possible coalitions of two members (66), 64 are stable. From these, two coalitions have the best payoff: C1-C2 and C1-C12. Those that are not stable are composed of two countries with the largest possible ED in the circular structure, namely C4-C8 and C6-C10 (for this case C1-C7 is stable).
3. Lower local benefit parameter $\lambda = 0.4$	2	66	0.87 CGI^M = 10.8	12.13 CGI^V = 11.6	All possible coalitions of two members are stable. Two of these have the best payoff: C1-C2 and C1-C12.
4. Higher regional benefit parameter $\rho = 0.075$	2	59	0.92 CGI^M = 12.2	13.72 CGI^V = 12.6	From all possible coalitions of two members (66), 59 are stable. From these, two coalitions have the best payoff: C1-C2 and C1-C12. The remaining seven coalitions that are unstable are composed of two countries that are i) either the farthest possible from each other (e.g. C1-C7), or ii) the combination between C2 and its farthest and second farthest option possible (e.g. C2-C7 and C2-C8), and between C12 and its farthest and second farthest option possible (e.g. C7-C12 and C6-C12).

Table 4.3 (continued). Coalition stability and CGI for increasing distances between countries [a/]

Change in parameter value	Size of stable coalitions (s^*)	Number of stable coalitions (out of a total of 4095)	Global biodiversity under best coalition (number of preserved species and CGI^M in %)	Global payoff under best coalition (absolute value and CGI^V in %)	Remarks
5. Lower regional benefit parameter $\rho = 0.025$	2	66	0.92 $CGI^M = 10.3$	12.58 $CGI^V = 11.3$	All possible coalitions of two members are stable. Two of these have the best payoff: C1-C2 and C1-C12.
6. Higher cost parameter $c = 1.1$	2	66	0.90 $CGI^M = 10.4$	12.70 $CGI^V = 11.3$	All possible coalitions of two members are stable. Two of these have the best payoff: C1-C2 and C1-C12.
7. Lower cost parameter $c = 0.9$	2	66	0.95 $CGI^M = 12.7$	13.61 $CGI^V = 13.3$	All possible coalitions of two members are stable. Two of these have the best payoff: C1-C2 and C1-C12.
8. Higher number of species protected at no cost $a = 0.10$	2	66	0.96 $CGI^M = 11.8$	14.0 $CGI^V = 12.3$	All possible coalitions of two members are stable. Two of these have the best payoff: C1-C2 and C1-C12.
9. Zero species protected at no cost $a = 0$	2	66	0.87 $CGI^M = 10.8$	12.16 $CGI^V = 11.7$	All possible coalitions of two members are stable. Two of these have the best payoff: C1-C2 and C1-C12.

a/ Only one parameter is changed at a time. All parameter changes are to be compared with the base model parameters in Table 4.1. The values of the parameter $\omega_{i,j}$ are those showed in the distance weighted parameter matrix for increasing distances between countries in Table A2 of the Appendix.

Note that not for every parameter change are all the 66 coalitions of two members stable. There are three cases where less coalitions are stable: i) the base model (with 63 stable coalitions), ii) the case with a higher local benefit parameter (with 64 stable coalitions), and iii) the case with a higher regional benefit parameter (with 59 stable coalitions). For the base model, those coalitions that are unstable are composed of any two countries that are the farthest away possible from each other: C1-C7, C4-C8 and C6-C10. These are the only three coalitions with the largest ED in the circular structure ($\omega_{i,j} = 0.04$). This result holds for the higher local benefit parameter λ, with the exception of coalition C1-C7 that is stable for this case.

The latter case, namely the one with the higher regional benefit parameter ρ, is slightly different from the other two. The seven coalitions consisting of two countries that are unstable are structured in two different ways. First, we find the coalitions composed of any two countries that are the farthest away possible: C1-C7, C4-C8 and C6-C10 (with $\omega_{i,j} = 0.04$ between coalition members). Second, we find the coalitions of two countries with a relatively large ED between them (second or fourth farthest distance possible in the circular structure), but also with a large ED between one of the coalition members and the singletons (see Figure 4.2). Those coalitions are: C2-C7 and C7-C12 (with $\omega_{i,j} = 0.08$ between coalition members), and C2-C8 and C6-C12 (with $\omega_{i,j} = 0.24$ between coalition members). The lack of stability of these coalitions is caused by a remoteness effect. First, coalition members protect more than they would under no cooperation in order to maximise their joint benefits. Then, singletons perceive higher payoffs due to the increase in conservation of coalition members. Yet, a coalition member perceives relatively lower regional benefits of conservation when it has a larger ED with respect to the other coalition member and also to the rest of the singletons; i.e. when the country is more remote. The latter effect offsets (part of) the gains from cooperation that the signatory perceives. Under this situation, coalition members have stronger incentives to deviate and therefore the coalition becomes internally unstable. The effect is more pronounced for a higher ρ parameter since this accentuates the remoteness effect. Only coalitions where members are closer to each other and to the singletons remain stable when ρ is increasing, since they can reap larger gains from cooperation.

Inclusion of transfers

Transfers are used as a tool to incentivise participation in an agreement such that larger coalitions may satisfy their internal stability conditions (Pavlova and de Zeeuw 2013). We applied an optimal sharing rule for the outcomes of our spatial biodiversity model to see whether larger stable coalitions could be achieved. We chose an optimal sharing rule because it emphasizes the relevance of individual outside options: the rule guarantees internal stability of a coalition when its payoff (weakly) exceeds the sum of the outside option payoffs (Weikard 2009).

Our calculations show that, for both scenarios, the inclusion of a transfer scheme does not have an impact on the number and size of stable coalitions for any parameter change in both scenarios. The fact that players have identical benefits and cost functions limits the potential impact of transfers on stability. For those parameter settings where all 66 coalitions of two members are stable, the inclusion of transfers has no impact. But also, for the scenario with increasing distances where some of the two-player coalitions are internally unstable, the inclusion of transfers does not help to restore internal stability either. The reason is that, in terms of payoff, either both members of these coalitions are worse off inside the coalition than when acting as singletons, or one is indifferent and the other is worse off. The gains from cooperation are too low to compensate for the remoteness effect.

Remoteness and location of countries in a spatial structure

In order to have a better understanding of the remoteness effect that we observed in the previous scenario and its relation to the location of countries, we set up a spatial pattern on our circular structure capable of illustrating the effects of countries that are clustered. So far we have seen that the maximum size that a coalition can reach under our payoff specification function is $s^* = 2$. With the spatial pattern that we set below we want to study the conditions under which we obtain the best payoff for a stable coalition of two members when there are clusters.

We consider three clusters in our circular structure. The first cluster is composed of two countries (C11 and C12) that have an ED of $d_{11,12} = x$. We name this first cluster 'cluster A'. A second cluster is also composed of two countries (C1-C2), but this time with a larger ED: $d_{1,2} = 2x$ We denote this cluster as 'cluster B'. Although C1 and C2 also form a cluster of two members, the distance between them is larger than that between countries of cluster A.

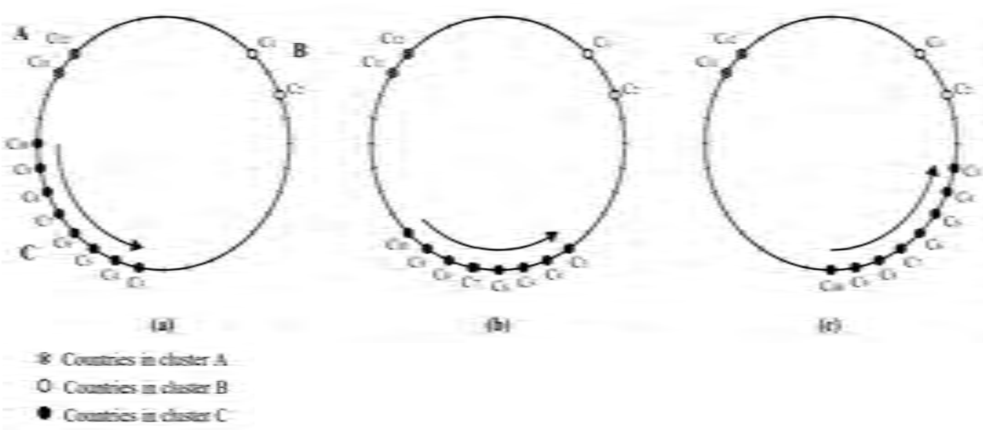


Figure 4.3 Circular structure with clustered countries

Finally, a large cluster that contains all remaining countries (from C3 to C10) is named 'cluster C'. Figure 4.3 (a) above shows how this circular spatial structure looks like.

Notice that in Figure 4.3(a) cluster B is more remote with respect to the rest of the countries. We then proceed to shift the position of the large cluster C from being closer to cluster A to being closer to cluster B. We do so by moving the large cluster C by four positions each time (4*x*) in two separate moments until reaching the spatial pattern represented in Figure 4.3(c). Table 4.4 shows the results of a coalition stability and global payoff for the three spatial patterns of clustered countries for the circular structures presented in Figure 4.3.

Table 4.4 shows that the maximum size of a stable coalition for all structures is $s^* = 2$. Only for the spatial structure in Figure 4.3(a) not every coalition of two members is stable: coalition C2-C8 is internally unstable. One of the members of this coalition is indifferent between remaining part of the coalition or becoming a singleton. However, the other member indeed obtains a higher payoff when deviating from the coalition.

In all structures, the coalition with the best payoff is located within the large cluster C. The largest gains from cooperation under the best coalition of two countries are achieved under the spatial pattern of Figure 4.3(b): 11.6% of the potential gains from the scenario of full cooperation can be reaped when a coalition of two countries is formed. Yet, the highest global payoff in absolute terms for a best coalition is obtained in the spatial pattern of Figure 4.3(a).

In order to get a more general understanding of the role of location on coalition formation, we construct an index of proximity between any two sets of players. For instance, to calculate the distance between cluster A and cluster C in the spatial structure in Figure 4.3(a), we define an index of proximity $\Omega_{A,C}$ between clusters that is measured by the distance weighted parameter matrix:

$$\Omega_{A,C} = \sum_{ieA} \sum_{jeC} \omega_{i,j} \ , \tag{4.12}$$

where *A* and *C* are the sets of countries in clusters A and C, respectively.

We denote an analogous index for cluster B and C as $\Omega_{B,C}$. According to the values of the distance weighted parameter matrix for this structure (see Table A3 in the Appendix) we obtain $\Omega_{A,C} = 6.40$ and $\Omega_{B,C} = 2$. The index is higher for cluster A than for cluster B: the larger the distance between two clusters in terms of ecosystem dissimilarity (ED), the smaller its index of proximity.

We perform a sensitivity analysis where we modified the position of the large cluster C throughout the circumference of the circle to see its effect on global payoff for three coalitions of two members. The three coalitions of two members that we consider are: the coalition composed of countries of cluster A (i.e. C11-C12), ii) the coalition composed of countries of cluster B (i.e. C1-C2), and iii) the coalition with the best payoff for each spatial structure in Figure 4.3. Figure 4.4 shows the results of this comparison.

Table 4.4. Coalition stability and CGI for spatial patterns of clustered countries [a/]

Change in parameter value	**Size of stable coalitions** *(s*)*	**Number of stable coalitions** (out of a total of 4095)	**Global biodiversity under best coalition** (number of preserved species and CGI^M in %)	**Global payoff under best coalition** (absolute value and CGI^V in %)	**Remarks**
Spatial structure 4.3(a)	2	65	0.92 $CGI^M = 10.1$	12.99 $CGI^V = 11.1$	All possible coalitions of two members are stable except for one coalition: C2-C8. Of all the stable coalitions, four have the best payoff: C4-C5, C5-C6, C6-C7, C9-10.
Spatial structure 4.3(b)	2	66	0.92 $CGI^M = 10.1$	12.95 $CGI^V = 11.6$	All possible coalitions of two members are stable. Of all stable coalitions, the one with the highest payoff is C6-C7.
Spatial structure 4.3(c)	2	66	0.92 $CGI^M = 10.1$	12.98 $CGI^V = 11.5$	All possible coalitions of two members are stable. Of all stable coalitions, there are two with the highest payoff: C4-C5 and C6-C7.

a/ Only one parameter is changed at a time. All parameter changes are to be compared with the base model parameters in Table 4.1. The values of the parameter $\omega_{i,j}$ are those showed in the distance weighted parameter matrix for clustered countries in Table A3, A4 and A5 of the Appendix.

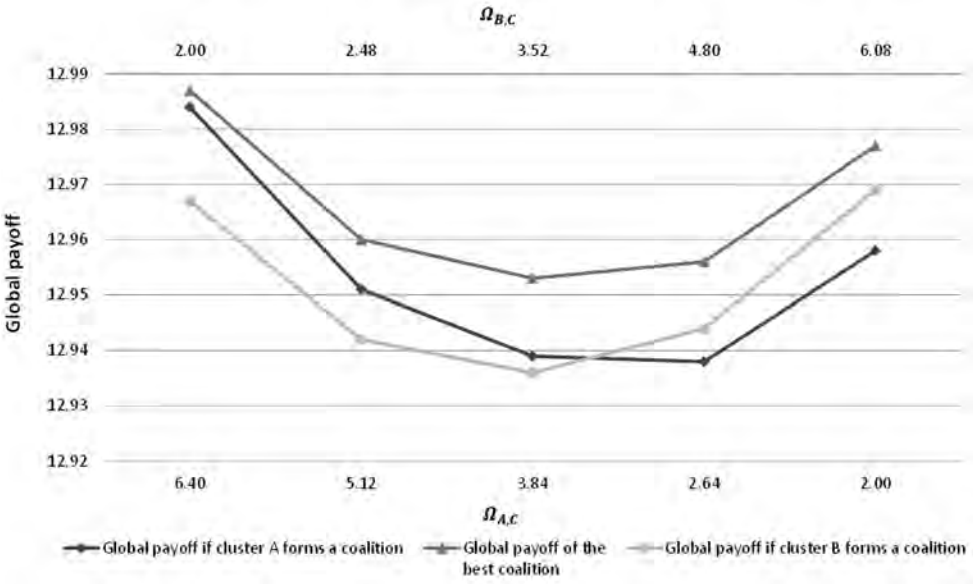


Figure 4.4 Global payoff of three different coalitions of two members for different values of the index of proximity

As the large cluster C moves to the right, the index of proximity between cluster A and C decreases from $\Omega_{A,C}$ = 6.4 to $\Omega_{A,C}$ = 2. Conversely, with the same movement, the index of proximity between clusters increases from $\Omega_{B,C}$ = 2 to $\Omega_{A,C}$ = 6.08. Note that the maximum value of $\Omega_{B,C}$ is lower than the maximum value of $\Omega_{A,C}$. The ED among countries in cluster A is smaller than the ED among countries in cluster B. For this reason the global payoff of the best coalition is higher when cluster C is positioned close to cluster A (i.e. a global payoff of 12.987) as compared to when cluster C is in an equally close position to cluster B (i.e. a global payoff of 12.977).

Benefits from conservation spillovers are higher when coalition members are closer to each other and to the signatories. We conclude that proximity of countries, not only between clusters, but also among members within a cluster, increases overall welfare.

4.5 Conclusions

Our study introduces a general approach to examine the impact of spatial structure on the stability of an IEA for biodiversity conservation. In particular, we analyse the role of location of and distance between countries in a specific circular structure to explain cooperation between neighbouring countries. We also introduce the notion of 'ecosystem dissimilarity' to measure distance between countries with respect to how different the set of species hosted by these countries are. We derive results that we summarise below.

One general finding of our study is that for all spatial patterns that we consider the maximum size of a stable coalition is $s^* = 2$. In the scenario with equidistant countries, all possible coalitions composed of two members are stable, but only those composed of neighbouring countries have the best payoff. Results are robust for all parameter changes that we perform for this scenario.

In the scenario with increasing distances between countries, results vary. For all parameter changes, only two coalitions achieve the highest global payoff. These coalitions are composed by any of the two possible combinations of countries with the smallest ecosystem dissimilarity between them. We also find that under this scenario not all coalitions of two members are stable. The reason is that the internal stability condition is violated due to a remoteness effect. If a coalition member has a relatively large ED with respect to the other coalition member and with respect to the singletons (that is, if it is more remote), it perceives lower regional benefits of conservation. Gains from cooperation are lower and hence the signatory has an incentive to deviate from the coalition. To sum up, higher regional benefits from conservation interfere with coalition stability, and this outcome is more prominent in coalitions composed of countries with relatively larger ED between them.

We constructed an index of proximity to obtain a more general understanding of the role of location on coalition formation. When we considered a scenario with countries grouped in clusters, we found that the global payoff of the coalition is highest when coalition members are closer to each other, and also closer to all other players. Under these circumstances, spillovers from conservation are as large as possible.

We examine the inclusion of a transfer scheme and find that it does not improve the size and number of stable coalitions in any of the scenarios. Furthermore, it does not stabilise those coalitions that are unstable due to a remoteness effect.

We conclude from the abovementioned results that the highest gains from cooperation are achieved when conservation agreements are formed between two countries hosting the most similar set of species. Moreover, gains from a bilateral agreement are enhanced when the set of species shared by country members and singletons is larger. Hence, both distance and remoteness of countries with respect to one another have an impact on their conservation measures and therefore on the global gains from cooperation.

Our results are dismal in terms of the creation of a global international agreement for conservation given that the maximum size a stable agreement can achieve is of two members. But the results suggest that instead of one single international agreement, the alternative of several bilateral agreements – potentially composed of countries with very similar ecosystems – could lead to more effective conservation outcomes.

Our analysis is restricted to symmetric countries in terms of benefits and costs of conservation, as well as on their biodiversity endowment. We adopted this assumption to facilitate the appraisal of the model. Yet, we recognise that it represents a strong simplification of reality and therefore limits the stability analysis. A further desirable extension of the model would be to drop the assumption of symmetric countries to study combined effects of asymmetries in costs and benefits of conservation, species endowment and location. We consider it important to further investigate the role of regional benefits of conservation in the formation of biodiversity agreements among heterogeneous countries. Under those circumstances, transfers could play a significant role in stabilising larger coalitions. Yet, this aspect is outside the scope of the analysis of this study and is left for future research.

4.6 Appendix

Distance weighted parameter matrices for the different spatial structures

For all scenarios we assume n = 12. We then assume that the distance between any country and itself is $d_{I,I}$ = 0; hence, we do not assign a value to the weight parameter of any country i and itself. Also, we consider plausible to assume that there is always at least one common element between the sets of species of any two countries. The larger the ecosystem dissimilarity (ED) between country i and j, the lower the value of the distance weighted parameter $\omega_{i,j}$. For simplicity of the model, we normalise the values of the distance weighted parameter for all three scenarios to be between zero and one, i.e. $0 \leq \omega_{i,j} \leq 1$.

For the scenario of equidistant countries we assume that any two neighbouring countries have the largest distance weight parameter of 0.6 and that those countries that are the farthest away in the circumference of the circle (e.g. C1 and C7) have a distance weight parameter of 0.1. The values between this range vary in 0.1.

Table A1. Distance weighted parameter matrix for equidistant countries [a/]

	C1	C2	C3	C4	C5	C6	C7	C8	C9	C10	C11	C12
C1	-	0.6	0.5	0.4	0.3	0.2	0.1	0.2	0.3	0.4	0.5	0.6
C2	0.6	-	0.6	0.5	0.4	0.3	0.2	0.1	0.2	0.3	0.4	0.5
C3	0.5	0.6	-	0.6	0.5	0.4	0.3	0.2	0.1	0.2	0.3	0.4
C4	0.4	0.5	0.6	-	0.6	0.5	0.4	0.3	0.2	0.1	0.2	0.3
C5	0.3	0.4	0.5	0.6	-	0.6	0.5	0.4	0.3	0.2	0.1	0.2
C6	0.2	0.3	0.4	0.5	0.6	-	0.6	0.5	0.4	0.3	0.2	0.1
C7	0.1	0.2	0.3	0.4	0.5	0.6	-	0.6	0.5	0.4	0.3	0.2
C8	0.2	0.1	0.2	0.3	0.4	0.5	0.6	-	0.6	0.5	0.4	0.3
C9	0.3	0.2	0.1	0.2	0.3	0.4	0.5	0.6	-	0.6	0.5	0.4
C10	0.4	0.3	0.2	0.1	0.2	0.3	0.4	0.5	0.6	-	0.6	0.5
C11	0.5	0.4	0.3	0.2	0.1	0.2	0.3	0.4	0.5	0.6	-	0.6
C12	0.6	0.5	0.4	0.3	0.2	0.1	0.2	0.3	0.4	0.5	0.6	-

a/ The distance between any two neighbours is the same.

For the scenario of increasing distances between countries, the calculation of the distance weighted matrix varies slightly. We first assume that when two countries are as close as possible in this spatial structure, that is when $d_{i,j} = x$, then the distance weighted value is of 0.84 (e.g. C1-C2). As the distance increases by *x*, then the weighted value decreases in 0.04. For instance, the distance between country C1 and C3 is of $d_{1,3} = 3x$, and therefore the distance weighted value in the matrix is of 0.84-(2*0.04)= 0.76. Finally, for those countries that are the farthest away in the circumference of the circle (e.g. C1 and C7), the distance is of $d_{1,7} = 21x$, and therefore the distance weighted value in the matrix is of 0.84-(20*0.04)= 0.04.

Table A2. Distance weighted parameter matrix for increasing distances between countries [a/]

	C1	C2	C3	C4	C5	C6	C7	C8	C9	C10	C11	C12
C1	-	0.84	0.76	0.64	0.48	0.28	0.04	0.28	0.48	0.64	0.76	0.84
C2	0.84	-	0.80	0.68	0.52	0.32	0.08	0.24	0.44	0.60	0.72	0.80
C3	0.76	0.80	-	0.76	0.60	0.40	0.16	0.16	0.36	0.52	0.64	0.72
C4	0.64	0.68	0.76	-	0.72	0.52	0.28	0.04	0.24	0.40	0.52	0.60
C5	0.48	0.52	0.60	0.72	-	0.68	0.44	0.20	0.08	0.24	0.36	0.44
C6	0.28	0.32	0.40	0.52	0.68	-	0.64	0.40	0.20	0.04	0.16	0.24
C7	0.04	0.08	0.16	0.28	0.44	0.64	-	0.64	0.44	0.28	0.16	0.08
C8	0.28	0.24	0.16	0.04	0.20	0.40	0.64	-	0.68	0.52	0.40	0.32
C9	0.48	0.44	0.36	0.24	0.08	0.20	0.44	0.68	-	0.72	0.60	0.52
C10	0.64	0.60	0.52	0.40	0.24	0.04	0.28	0.52	0.72	-	0.76	0.68
C11	0.76	0.72	0.64	0.52	0.36	0.16	0.16	0.40	0.60	0.76	-	0.80
C12	0.84	0.80	0.72	0.60	0.44	0.24	0.08	0.32	0.52	0.68	0.80	-

a/ Distances between countries increase as we move further throughout the circumference of the circular structure. The lowest possible distance weighted value is related to the countries with the largest ecosystem dissimilarity (ED) in the circular structure (e.g. C1-C7).

For the scenario clustered countries, we proceed to follow a similar calculation to the increasing distance scenario. Yet, we set that when $d_{i,j} = x$, then the distance weighted value is of 0.64 (e.g. C11-C12) in Figure 4.3(a). Also for this case, as the distance increases by *x*, then the weighted value decreases in 0.04. For instance, in the same Figure 4.3.(a), the largest possible distance between two countries (e.g. between countries C1 and C6) is of $d_{1,6} = 16x$. Therefore, the distance weighted value in the matrix is of 0.64-(15*0.04)= 0.04. This same calculation is used for all parameter matrices for clustered countries.

Table A3. Distance weighted parameter matrix for clustered countries: spatial structure 4.3(a)

	C1	C2	C3	C4	C5	C6	C7	C8	C9	C10	C11	C12
C1	-	0.60	0.16	0.12	0.08	0.04	0.08	0.12	0.16	0.20	0.32	0.36
C2	0.60	-	0.24	0.20	0.16	0.12	0.08	0.04	0.08	0.12	0.24	0.28
C3	0.16	0.24	-	0.64	0.60	0.56	0.52	0.48	0.44	0.40	0.28	0.24
C4	0.12	0.20	0.64	-	0.64	0.60	0.56	0.52	0.48	0.44	0.32	0.28
C5	0.08	0.16	0.60	0.64	-	0.64	0.60	0.56	0.52	0.48	0.36	0.32
C6	0.04	0.12	0.56	0.60	0.64	-	0.64	0.60	0.56	0.52	0.40	0.36
C7	0.08	0.08	0.52	0.56	0.60	0.64	-	0.64	0.60	0.56	0.44	0.40
C8	0.12	0.04	0.48	0.52	0.56	0.60	0.64	-	0.64	0.60	0.48	0.44
C9	0.16	0.08	0.44	0.48	0.52	0.56	0.60	0.64	-	0.64	0.52	0.48
C10	0.20	0.12	0.40	0.44	0.48	0.52	0.56	0.60	0.64	-	0.56	0.52
C11	0.32	0.24	0.28	0.32	0.36	0.40	0.44	0.48	0.52	0.56	-	0.64
C12	0.36	0.28	0.24	0.28	0.32	0.36	0.40	0.44	0.48	0.52	0.64	-

Table A4. Distance weighted parameter matrix for clustered countries: spatial structure 4.3(b)

	C1	C2	C3	C4	C5	C6	C7	C8	C9	C10	C11	C12
C1	-	0.60	0.32	0.28	0.24	0.20	0.16	0.12	0.08	0.04	0.32	0.36
C2	0.60	-	0.40	0.36	0.32	0.28	0.24	0.20	0.16	0.12	0.24	0.28
C3	0.32	0.40	-	0.64	0.60	0.56	0.52	0.48	0.44	0.40	0.12	0.08
C4	0.28	0.36	0.64	-	0.64	0.60	0.56	0.52	0.48	0.44	0.16	0.12
C5	0.24	0.32	0.60	0.64	-	0.64	0.60	0.56	0.52	0.48	0.20	0.16
C6	0.20	0.28	0.56	0.60	0.64	-	0.64	0.60	0.56	0.52	0.24	0.20
C7	0.16	0.24	0.52	0.56	0.60	0.64	-	0.64	0.60	0.56	0.28	0.24
C8	0.12	0.20	0.48	0.52	0.56	0.60	0.64	-	0.64	0.60	0.32	0.28
C9	0.08	0.16	0.44	0.48	0.52	0.56	0.60	0.64	-	0.64	0.36	0.32
C10	0.04	0.12	0.40	0.44	0.48	0.52	0.56	0.60	0.64	-	0.40	0.36
C11	0.32	0.24	0.12	0.16	0.20	0.24	0.28	0.32	0.36	0.40	-	0.64
C12	0.36	0.28	0.08	0.12	0.16	0.20	0.24	0.28	0.32	0.36	0.64	-

Table A5. Distance weighted parameter matrix for clustered countries: spatial structure 4.3(c)

	C1	C2	C3	C4	C5	C6	C7	C8	C9	C10	C11	C12
C1	-	0.60	0.48	0.44	0.40	0.36	0.32	0.28	0.24	0.20	0.32	0.36
C2	0.60	-	0.56	0.52	0.48	0.44	0.40	0.36	0.32	0.28	0.24	0.28
C3	0.48	0.56	-	0.64	0.60	0.56	0.52	0.48	0.44	0.40	0.12	0.16
C4	0.44	0.52	0.64	-	0.64	0.60	0.56	0.52	0.48	0.44	0.08	0.12
C5	0.40	0.48	0.60	0.64	-	0.64	0.60	0.56	0.52	0.48	0.04	0.08
C6	0.36	0.44	0.56	0.60	0.64	-	0.64	0.60	0.56	0.52	0.08	0.04
C7	0.32	0.40	0.52	0.56	0.60	0.64	-	0.64	0.60	0.56	0.12	0.08
C8	0.28	0.36	0.48	0.52	0.56	0.60	0.64	-	0.64	0.60	0.16	0.12
C9	0.24	0.32	0.44	0.48	0.52	0.56	0.60	0.64	-	0.64	0.20	0.16
C10	0.20	0.28	0.40	0.44	0.48	0.52	0.56	0.60	0.64	-	0.24	0.20
C11	0.32	0.24	0.12	0.08	0.04	0.08	0.12	0.16	0.20	0.24	-	0.64
C12	0.36	0.28	0.16	0.12	0.08	0.04	0.08	0.12	0.16	0.20	0.64	-

Chapter 5

International cooperation for conservation of the wintering habitat of the Golden-winged Warbler: a spatial economic model[10]

We introduce a spatial structure in a game-theoretical framework to analyse coalition stability of an International Environmental Agreement (IEA) for habitat conservation. Specifically, we focus on habitat conservation of wintering areas of a migratory bird species that is facing a sharp, long-term population decline: the Golden-winged Warbler (*Vermivora chrysoptera*). Our model makes use of a spatial setting that describes the incidence of migratory connectivity and the distance between countries (i.c. the United States, Honduras, Nicaragua, Costa Rica, Colombia and Venezuela) and their impacts on conservation efforts and consequent payoffs. We find that for the three spatial scenarios that we analyse, and with the inclusion of transfers, full cooperation is achieved. Under both scenarios of weak and strong migratory connectivity, Honduras and Costa Rica undertake low additional conservation efforts compared to the business as usual scenario due to their high costs of conservation. For all scenarios of our study, the United States transfers part of its payoff to the Latin American countries in order to incentivise conservation and stabilise the coalition. Our results suggest that with the implementation of a transfer scheme, there is scope for an effective conservation agreement between Latin American countries and the United States as long as US households allocate a positive value to the stabilisation of the Golden-winged Warbler population.

[10] This chapter is based on Alvarado-Quesada, I., Elizondo, P., Weikard, H.-P., & van Ierland, E.C. (2015). International cooperation for conservation of the wintering habitat of the Golden-winged Warbler: a spatial economic model. Working paper.

5.1 Introduction

High rates of deforestation and habitat fragmentation in Latin America and the Caribbean represent one of the main threats to the region's biodiversity (UNEP 2010, Sodhi et al. 2011, Blackman et al. 2014). These threats also affect the survival of one of the most sharply declining bird populations in North and Central America: the Golden-winged Warbler (*Vermivora chrysoptera*) (Buehler et al. 2007). Golden-winged Warblers are migratory birds that travel in their non-breeding season (September to April) from North America to Central America and the northern territories of South America. This forest-dependent songbird has experienced a population decline of 2.8% per year for more than forty years (Sauer et al. 2004, Buehler et al. 2007, Confer et al. 2011).

Several reasons are given for the decline of the species' population: hybridisation with the Blue-winged Warbler, loss of breeding habitat, parasitism, and loss of wintering habitat (Gill 2004, Buehler et al. 2007). Conservation of this species has become a main focus of the United States Fish and Wildlife Service (USFWS) and the Cornell Lab's Conservation Science program for more than a decade (Donovan et al. 2002). As a result, the Golden-winged Warbler Working Group was established in 2005. This initiative gathered organisations, academics and individuals to promote the conservation and understanding of the limiting factors of the species. Later on, the Golden-winged Warbler Alliance was created as the work component in the tropical countries (Barker et al. 2008). This alliance is in charge of the development of a conservation plan for wintering grounds, mainly located in Honduras, Nicaragua, Costa Rica, Panamá, Colombia and Venezuela.

Recent attention has been drawn towards this species, not only because of its threatened status, but also because of its charismatic attributes that appeal bird watchers and conservationists. Consequently, a number of studies have been developed on the species' biology, natural history, and conservation status: e.g. Neville et al. (2008), Chandler and King (2011), Confer et al. (2011), Bennett (2012), ABC (2014), and ABC (2014a). Yet, there are still many gaps in the understanding of the species' population dynamics and demographics, especially in the tropical latitudes (Donovan et al. 2002, Confer et al. 2011, Elizondo et al. 2014) that need to be further studied.

This study focuses on the viability of an international environmental agreement (IEA) for conservation of the wintering habitat of the Golden-winged Warbler. We develop a game theoretical model to analyse the incentives of countries to join an agreement for the protection of wintering habitats. In order to calibrate our model, we make use of the data set compiled by the Golden-winged Warbler Alliance together with the American Bird Conservancy (ABC) to create a conservation plan for wintering grounds in the region (ABC 2014, ABC 2014a, Elizondo et al. 2014, Golden-winged Warbler Alliance 2015).

The model elaborates on the methodology of game coalition formation in IEAs that emerged since D'Aspremont et al. (1983), Hoel (1992), Carraro and Siniscalco (1993) and Barrett (1994), and that has extended to recent economic analyses that explicitly address IEAs for conservation: e.g. Punt et al. (2012), Ansink and Bouma (2013), Winands et al. (2013), Alvarado-Quesada and Weikard (2015) and Alvarado-Quesada and Weikard (2015a).

Our model considers both local and regional benefits of conservation of wintering habitat (in addition to local costs of conservation) in a country's decision to undertake conservation efforts. In particular, we relate regional biodiversity benefits to the spatial dimension: we assume that they are dependent on the geographical distance between countries and on the migratory connectivity of the bird species. Migratory connectivity *'describes the movement of individuals between summer and winter populations, including intermediate stopover sites'* (Webster et al. 2002, p.77). For the case of the Golden-winged Warbler, connectivity determines whether bird populations migrating from breeding areas in the North have overlapping wintering habitats in the South that could potentially be considered as substitutes.

For the analysis of our spatial model we follow a game theoretical approach commonly used to study the stability of IEAs. In particular we study a cartel game where countries decide to join or not a conservation agreement based on their net benefits from habitat conservation. We examine stability and effectiveness of a conservation agreement. For a more comprehensive analysis, we consider a transfer scheme.

The structure of this chapter is as follows. Section 5.2 gives a description of the species and its migratory pattern. In Section 5.3 we state the information used to calibrate the parameters of our spatial model. Section 5.4 presents a detailed explanation of the structure of the spatial model for habitat conservation as well as of the different scenarios that we examine. In Section 5.5 we explain our numerical analysis, and in Section 5.6 we present the results of our model. Section 5.7 provides a general discussion of the results and concludes.

5.2 Description of the species

The Golden-winged Warbler (*Vermivora chrysoptera*) is a Neotropical-Nearctic migratory songbird species. These birds have short tails and slim bodies (8-11 g) covered in a silver grey plumage with golden flashes on the head and wings. As insectivores, they look for their food among the foliage by probing into rolled-up leaves with their thin, sharply-pointed bills (Sibley 2000, Sibley 2009, Cornell Lab of Ornithology 2015).

5.2.1 Migratory path

Golden-winged Warblers breed in the northern territories of the American continent, mainly in the States of Wisconsin, Minnesota, Michigan and Pennsylvania. They are long distance migrants: during the non-breeding season, this species occupies territories from the north of Central America to the northern part of South America, i.e. Colombia and Venezuela (Chandler and King 2011, Elizondo et al. 2014). Its mayor abundance is concentrated in Nicaragua, Honduras, Costa Rica and Panama, generally with more abundance in the Caribbean (Rappole et al. 1976, Ridgley and Tudor 1989, Banks et al. 2003). Associations between mortality and long distance travels by birds are known to be substantial (Sillett and Holmes 2002), especially among younger individuals (Moore et al. 2005). It is typically reported that between 30-90% of the individuals of migratory birds will not return to their breeding grounds (Newton 2006).

The migration of Golden-winged Warblers to the South takes place on an annual basis around the month of September, and their return (spring) migration to North America occurs around April. As most migratory birds, their migratory trajectory includes flight and stopover phases. Stopover sites are those where birds pause between migratory flights (Moore et al. 2005), and stopovers can last from a few hours to a few days (Gill 2007). Conditions of stopover sites are such that birds can meet their nutritional requirements in a short period of time, while wintering habitats are those where birds stay for an extended period of time during their non-breeding season (Gill 2007).

5.2.2 Habitat specifics

During their breeding season, Golden-winged Warblers occupy wet, tangled shrubby habitats with some tall trees. In their wintering habitats they are forest-dependent with a large home range (Chandler 2013). Within forests they have specialised microhabitat requirements '*such as hanging dead leaves and vine tangles used as foraging substrates*' (Chandler and King 2011, p.1045), consequently making them more vulnerable to tropical deforestation. Abundance and occupancy appear to be affected more by climate and microhabitat features than by habitat type (Chandler and King 2011). Evidence suggests that both male and female Golden-winged Warblers prefer forests with intermediate levels of epiphytes, in which they are known to forage (Chandler 2013), although segregation of the sexes in different habitat types might occur (Bennett 2015, Golden Winged Warbler Alliance 2015).

The reproductive success of long-distance migratory birds can be influenced by the quality of habitat in their tropical wintering grounds. The loss of high-quality winter habitat may have a negative carry-over effect on individuals during their next breeding season and can lead to a decline of their population abundance (Norris et al. 2004). The protection of wintering habitats enhances the availability of food and shelter requirements for Golden-winged Warblers, and therefore assists in halting the decline of their population.

5.2.3 Migratory connectivity

An important yet still unresolved issue in the study of many migratory birds is their migratory connectivity. Migratory connectivity is *'the degree to which individuals from the same breeding site migrate to the same wintering site'* (Trierweiler et al. 2014, p.1). Technological advances such as satellite transmitters, genetic markers, and stable isotopes measurements have contributed to improve the study of migratory populations in the past years (see e.g. Rushing et al. 2014). However, at the time of this study, there is no sufficient scientific evidence of the strength of migratory connectivity of the Golden-winged Warbler, just as there is no information either for most other bird species. A study that uses geolocators to assess the migratory connectivity of Golden-winged Warblers is currently underway; yet, results are not expected to be available before 2016 (Virginia eBird 2015).

5.3 Materials

To develop our analysis of international cooperation we make use of the information that has been compiled by the ABC as part of the completion of the Golden-winged Warbler Wintering Grounds Conservation Plan (ABC 2015a). The ABC executed an analysis on the threats faced by the Golden-winged Warbler in six countries where the bird is known to occur: Honduras, Nicaragua, Costa Rica, Panama, Colombia and Venezuela (ABC 2014 and ABC 2014a). The assessment resulted in the delineation of focal areas of conservation per country. Such areas vary in size, land use, land ownership and market value. Specialists from each country cooperated with collaborators from the United States to gather data and write reports. A description of the focal areas considered in our study is available in Appendix 5.1 (Table A1).

Next to the information compiled by the ABC, we base our study on two main data sources. First, we used the country factsheets elaborated for the Conservation Plan to obtain general information per focal area such as size, land use, percentage of land devoted to coffee production, and market value of land (when protected and when cleared for other activities). Even though Panama is part of the assessment conducted by the ABC, we do not include Panama in our study because key information to calibrate parameters was either incomplete or missing.

Additionally, we made use of a geodatabase containing an interpolated raster of Golden-winged Warbler's male occupancy of all focal areas (ABC 2015). This information was calculated and based on Chandler's species distribution model (2013) for the over-wintering period (November to February) of the Golden-winged Warbler for the five countries of the study. The model uses systematic point counts in each survey location to account for differential detection probabilities and incorporates negative data.

Focal areas were delineated and refined based on the male occurrence probability and the expert opinions of in-country collaborators. Only male occurrence probability was considered because males and females are showing habitat segregation on the winter range by elevation, canopy height, and rainfall. So far, no survey attempt has yet made an overview to assess female habitats in the winter range. Hence, the accuracy of our study is limited due to the lack of information regarding female occupancy. We acknowledge that different occupancy patterns for female individuals may lead to different results. However, we perform our analysis with the best information available. A follow-up study using the same approach can be conducted in the future when relevant data on female occupancy becomes available.

Occupancy is the outcome of a stochastic process (i.e. $z_i = 1$ if the plot i is occupied, and $z_i = 0$ if the plot i is not occupied by the species) and its expected value is the occurrence probability, $\psi = \Pr(z_i = 1)$ (Royle and Dorazio 2008, p.83). For the calibration of our model, we make use of occurrence probability values.

5.4 A spatial model for habitat conservation

5.4.1 Model description

We make use of a game-theoretical model to assess the viability of an IEA for habitat conservation of the wintering areas of the Golden-winged Warbler. We start by considering our set N composed of $n = 6$ countries: the five Latin American countries with wintering habitat (LAC) – namely Honduras (HON), Nicaragua (NIC), Costa Rica (CR), Colombia (COL), and Venezuela (VEN) – and one country with no wintering habitat but with benefits to be reaped from regional conservation in the wintering range – namely the United States (USA). Habitat conservation in LAC benefits USA since the whole population of Golden-winged Warblers migrates back to North America for the breeding season in spring. Rates of weight gain, departure weights and stopover durations sometimes influence the subsequent survival and reproductive success of individuals in their breeding habitats (Norris et al. 2004, Newton 2006, Harrison et al. 2013).

We follow a game theoretical approach based on the maximisation of the net benefits of conservation of the wintering habitat of the Golden-winged Warbler by coalition members and singletons in a cartel game. To make our framework operational, we consider efforts of conservation e_j as the choice variable of any country $j \in N$. In our model, efforts of conservation reflect how much conservation is carried out per country to protect their wintering habitats. Conservation efforts are measured in number of protected hectares.

We refine the conventional specification of the payoff function that is often used in IEA theory (Carraro and Siniscalco 1993, Barrett 1994). We include general characteristics that are relevant for the appropriate setup of a biodiversity agreement, and also more specific ones that are relevant for the design of a model for conservation of migratory bird species. These characteristics are: i) the inclusion of the occurrence probability of the species in country j, ψ_j, as a function of the conservation efforts of the country, i.e. $\psi = f(e_j)$, and ii) the inclusion of a specific spatial structure to describe the species' expected migratory connectivity. To simplify the analysis we assume that the average occurrence probabilities per country ψ_j are independent of each other.

Costs of conservation of wintering habitat of the Golden-winged Warbler are related to land market values per country and reflect the opportunity costs. As for benefits of conservation, the model accounts for the different scales at which they can be perceived, i.e. at local and regional scales. We describe below the benefits and costs of conservation associated with the payoff functions of the countries.

Local benefits of conservation

Local benefits of conservation refer to those benefits accruing to any country $j \in N$ where the Golden-winged Warbler can be observed. For our study we focus in particular on local, direct benefits.

Direct benefits of conservation are the economic values that humans perceive from the conservation of the species itself. This type of benefits can fall under different categories: use value, option value, existence value and bequest value (Loomis and White 1996, p.198). Some of these values are difficult to quantify because birds provide utility to human beings in a way that is not tradable in the market (Zander et al. 2014). For these cases where there is no price assigned but there is certainly a benefit, economists implement non-market valuation techniques. Contingent valuation studies have been used to estimate the willingness to pay (WTP) for bird species to either avoid their loss, secure a gain in population, or contribute to a conservation plan (see Loomis and White 1996, Brouwer et al. 2008 and Martín-López et al. 2008 for reviews of WTP studies for different species). As yet, no valuation study has been performed to assess anthropogenic benefits associated with the conservation of the Golden-winged Warbler. We therefore approximate the local benefits of habitat conservation by focusing on one of the ecosystem services that are provided by birds, in particular their role in pest reduction in coffee plantations. This implies that our estimates represent a low estimate of the value of the bird.

Recent literature has drawn attention upon birds' provision of ecosystem services (Whelan et al. 2008, Sodhi et al. 2011) and the need to quantify them (Wenny et al. 2011, Green and Elmberg 2014). One of the several ecological functions of birds is their role as pest controllers. This service is mainly provided by insectivore birds by means of their foraging activity (Wenny et al. 2011). A recent study conducted by Karp et al. (2013) in coffee farms in Costa Rica showed that birds can reduce the damages caused by the coffee berry borer (*Hypothenemus hampei*) by a half. Pest control services provided by birds were estimated to save farmers between US$75-US$310 per hectare over a year's harvest. Karp showed that there were five insectivorous bird species found to be borer predators. Two of these species are warblers (i.e. Rufous-capped Warbler and Yellow Warbler), and we assume that the Golden-winged Warbler can also act as a borer consumer species that delivers these types of benefits to coffee plantations (see Table 1 in Karp et al. 2014).

In our model local benefits of conservation are specified as a function of the occurrence probability of the Golden-winged Warbler per country. Consequently, we define the local benefits of habitat conservation as:

$$L_i = \lambda_i \cdot \psi_i \qquad \forall\, i \in N, \tag{5.1}$$

where

L_i are benefits of local habitat conservation for country $i \in N$

λ_i is the parameter for benefits of local habitat conservation for country *i*, $\lambda_i > 0$

ψ_i is the occurrence probability of the Golden-winged Warbler in country *i* as a function of its conservation efforts e_i, where $0 \leq \psi_i \leq 1$

Regional benefits of conservation

In Section 5.2.3 we have addressed the importance of the geographical connections between breeding and non-breeding habitats of migratory bird populations. We also addressed that, up to date, there is no information available regarding the exact dynamics of the migratory connectivity of the Golden-winged Warbler. We recognise the existence of knowledge gaps in the literature with respect to i) the order and number of wintering habitats that the species visit during their winter migration, ii) the migratory connectivity of the population of Golden-winged Warblers; i.e. strong vs. weak connectivity between breeding and non-breeding habitats (Webster et al. 2002), and iii) the impact of wintering habitat loss in country $i \in N$ on the survival of the Golden-winged Warbler in any country $j \in N$, $j \neq i$.

Nevertheless, there seems to be a consensus among experts studying the Golden-winged Warbler that conservation efforts should be implemented at an international scale to reverse the bird's population decline. This consensus is sustained by evidence that states that connectivity between breeding and non-breeding habitats has an impact on the survival of bird species' populations (Harrison et al. 2013, Hallworth et al. 2014, Cooper et al. 2015, Marra et al. 2015). In this study we support this premise and consider this connectivity in our study to evaluate its impact on regional cooperation to preserve wintering habitats of the Golden-winged Warbler. In our model, the connectivity is embedded in the specification of the regional benefits of conservation. Specifically, the regional benefits are measured by a combination of the geographical distance between countries and the degree of connectivity between breeding and non-breeding habitats. Hence, in addition to the local benefits of conservation, we assume that countries also benefit from efforts for the conservation of wintering habitats for this bird in other countries.

Benefits of regional conservation can be perceived as an option value for country *i*. First, if the species is extinct or threatened in country *i*, it could be reintroduced from another country $j \in N$ where the species still survives, allowing for the repopulation of country *i*'s original sites. Second, country *i*'s bird population could use other wintering habitats as overlapping habitats that could be considered as substitutes.

The specification of the regional benefits differs slightly from that of the local benefits. We define regional benefits of habitat conservation of country *i* as a function of two variables: the conservation efforts e_j of any country $j \in N_{-i}$, and their estimated occurrence probability, ψ_j, for all $j \in N_{-i}$. The product of these two variables is an indicator of the size of the population of the bird species in country *j*.

The impact of the estimated occurrence probability ψ_j and the conservation efforts e_j on the regional benefits of habitat conservation of country *i* will be determined by the distance between them ($d_{i,j}$), and their degree of connectivity within the migratory route (i.e. weak or strong connectivity). In order to include these two factors in the payoff function of the model, we define a weight parameter denoted as $\omega_{i,j}$. More precisely, $\omega_{i,j}$ is equal to 1 minus the normalised distance between two countries *i* and *j*. Depending on the degree of connectivity, this parameter is multiplied by an additional factor (see Appendix 5.3 for detailed calculations). Consequently, we define regional benefits of habitat conservation in our model as:

$$R_i = \rho_i \left[\sum_{j \in N_{-i}} \omega_{i,j} \cdot \psi_j \cdot \left(e_j - e_{min} \right) \right] \qquad \forall\, i, j \in N, \tag{5.2}$$

where

R_i are benefits of regional habitat conservation for country $i \in N$

ρ_i is the scaling parameter for regional benefits of country *i* stemming from weighted conservation of other countries, $\rho_i > 0$

$\omega_{i,j}$ is the parameter that weighs conservation of any other country $j \in N_{-i}$ in the regional benefits of habitat conservation of country *i*, by means of the combined impact of distance and degree of connectivity where $0 \leq \omega_{i,j} \leq 1$.

e_j is the effort of conservation in country $j \in N_{-i}$ measured in hectares, where $e_j \geq e_{min}$

e_{min} is the minimum viable conservation (in ha) required for the Golden-winged Warbler to occur

According to expert criteria we assume, as an averaged baseline, that the minimum size of a plot for the Golden-winged Warbler to occur is of 10 ha (see Roth and Lutz 2004 and Martin et al. 2007 for examples of density values in breeding habitats). Therefore, we assume e_{min} = 10 as a constant for all countries. Notice that the weight assigned to other countries' estimated occurrence probabilities and to their conservation efforts is a decreasing function of the distance between any two countries $d_{i,j}$: the larger the distance, the smaller the weight $\omega_{i,j}$ and therefore the smaller the impact of *j*'s habitat conservation on *i*'s regional benefits. Regarding the parameter for regional benefits of country *i*, ρ_i, we calibrate it as a fraction of the parameter for local benefits of conservation λ_i. The only exception is the regional benefit parameter for USA because USA has no local benefits of conservation in wintering habitats. Hence, for USA the calibration is different. It is calibrated as such that it expresses the willingness to pay for the conservation of the bird population. We discuss this further in Section 5.5 and in Appendix 5.2. The complete spatial structure of our model is described in detail in Section 5.4.2.

Local costs of conservation

Costs of wintering habitat conservation of the Golden-winged Warbler are directly related to efforts of conservation per country, e_i. We assume that in the absence of a conservation agreement, countries already undertake a certain level of conservation efforts at no cost. At the same time, such costless conservation efforts are linked to an average occurrence probability per country. We define these costless conservation efforts and occurrence probability values per country as the BAU scenario: e_{BAU_i} and ψ_{BAU_i} (see Table 5.1 in Section 5.5).

We use a quadratic specification for the local costs of conservation where the cost parameter values are derived from the opportunity costs of land conservation per country. We define local costs of conservation as:

$$C_i = c_i \left(e_i - e_{BAU\,i}\right)^2 \qquad \forall\, i \in N, \qquad (5.3)$$

where

C_i are benefits of local habitat conservation for country $i \in N$

c_i is the parameter for costs of local habitat conservation for country i, $c_i > 0$

$e_{BAU\,i}$ is the conservation effort at no cost for country i

Payoff functions

In order to obtain the net benefits derived from the conservation of wintering habitats of the Golden-winged Warbler, we put together equations (5.1), (5.2) and (5.3) and we obtain the following payoff function for country *i*:

$$\pi_i = \lambda_i \psi_i + \rho_i \left[\sum_{j \in N_{-i}} \omega_{i,j} \cdot \psi_j \cdot \left(e_j - e_{min} \right) \right] - c_i \left(e_i - e_{BAU\,i} \right)^2 \qquad \forall i \notin S. \qquad (5.4)$$

We observe in equation (5.4) that benefits of local and regional conservation of wintering habitats of the Golden-winged Warbler are a function of the occurrence probability values of the country itself and of the rest of the countries.

In our model we consider the occurrence probability of the Golden-winged Warbler as a function of the efforts of conservation; i.e. the number of protected hectares from the focal areas of each country. We consider plausible to assume that an increase in wintering habitat conservation will lead to higher occurrence probabilities of the species. *'Occupancy is (...) the outcome of a process that governs how individuals are distributed in space. Therefore, it is necessarily a product of abundance or density and the parameters that govern the dynamics of such processes'* (Royle and Dorazio 2008, p.127). One of the parameters determining abundance, and hence occupancy is size of the available habitat for the species to spend its wintering time.

In order to represent this relation, we make use of a parabolic function. Our function for the occurrence probability of country *i* is then:

$$\psi_i = \delta_i \left[-\left(e_i - e_{min} \right)^2 + 2 e_{max_i} \left(e_i - e_{min} \right) \right] \quad \forall i \in N, \qquad (5.5)$$

where

δ_i is a scaling parameter of the parabolic function, $\delta_i > 0$

e_{max_i} is the maximum conservation efforts possible that can be carried out in country *i*; i.e. country's *i* habitat endowment

For this functional form we assume that the maximum occurrence probability that country i can reach, ψ_{max_i}, is obtained when all the focal areas defined in country i are used for conservation purposes, i.e. when country i's conservation endowment e_{max_i} is fully protected (see Figure 5.1). Furthermore, this specification suggests that each additional protected hectare has a lower marginal impact on the occurrence probability than the unit previously protected.

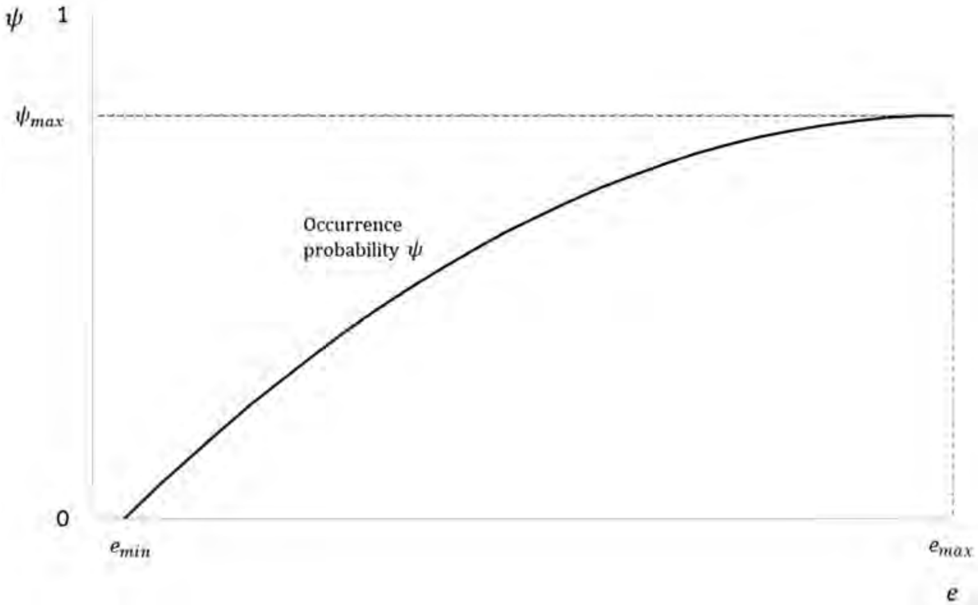


Figure 5.1 Parabolic function of the occurrence probability of the Golden-winged Warbler

Substituting equation (5.5) in the payoff function of country i in equation (5.4) we obtain:

$$\pi_i = \lambda_i\delta_i\left[-\left(e_i - e_{min}\right)^2 + 2e_{max_i}\left(e_i - e_{min}\right)\right] + \rho_i\left[\sum_{j\in N_{-i}} \omega_{i,j}\cdot\left\{\delta_j\left[-\left(e_j - e_{min}\right)^2 + 2e_{max_j}\left(e_j - e_{min}\right)\right]\right\}\cdot\left(e_j - e_{min}\right)\right] - c_i\left(e_i - e_{BAU\,i}\right)^2 \qquad \forall i \notin S. \tag{5.4'}$$

Each country $i = 1,\ldots n$ maximises its total payoff function subject to its own conservation effort e_i. The effort of conservation in equilibrium is given by:

$$e_i^* = \frac{\lambda_i\delta_i\left(e_{min} + e_{max_i}\right) + c_i e_{BAU\,i}}{c_i + \lambda_i\delta_i} \qquad \forall i \notin S. \tag{5.6}$$

Also, signatories maximise the coalition benefits, leading to the following equilibrium effort of conservation:

$$e_i^* = \frac{\left[2(*)\left(e_{min}+e_{max_i}\right)-\lambda_i\delta_i-c_i\right]+\sqrt{\left[2(*)\left(e_{min}+e_{max_i}\right)-\lambda_i\delta_i-c_i\right]^2+3(*)\left[2\lambda_i\delta_i\left(e_{min}+e_{max_i}\right)-e_{min}(*)\left(3e_{min}+4e_{max_i}\right)+2c_ie_{BAU\,i}\right]}}{3(*)} \qquad \forall i \in S, \tag{5.7}$$

where $(*)=\delta_i\left(\sum_{j\in S_{-i}}\omega_{i,j}\cdot\rho_j\right)$ and S is the set of signatory countries. To facilitate the model appraisal, a summary table of variables and parameters of the model can be found in Appendix 5.2 (Table A2).

Coalition stability

Having formally described the specification of benefits and costs of habitat conservation, we consider strategic incentives to cooperate in an IEA for habitat conservation. We consider a two-stage game. In the first stage of the game countries choose to join or not the IEA. In the second stage, countries that join the agreement – the signatories – coordinate their actions to maximise their collective net benefits of habitat conservation.

In order to identify the subgame perfect equilibria of the game, we identify equilibrium membership choices by considering the decisions that countries make in the second stage of the game. We say that in an IEA, a set of member countries *S* is stable if no member country has an incentive to leave the coalition *S*. We also require that the remaining singletons, as outsiders, have no incentive to join the coalition *S*. Formally, the conditions for coalition *S* to be internally (IS) and externally (ES) stable are:

$$\text{IS:} \qquad \pi_i^*(S) \geq \pi_i^*\left(S\setminus\{i\}\right) \qquad \forall i \in S, \tag{5.8}$$

and

$$\text{ES:} \qquad \pi_j^*(S) \geq \pi_j^*\left(S\cup\{j\}\right) \qquad \forall j \notin S. \tag{5.9}$$

where $\pi_i^*(S)$ is the payoff of a signatory and $\pi_j^*(S)$ is the payoff of singleton when coalition *S* is formed.

5.4.2 Spatial structure of the game: description of scenarios

To investigate the stability of an agreement for habitat conservation of a migratory bird species, we need to examine the relation between the breeding habitats and the non-breeding habitats of the Golden-winged Warbler, or in other words, its migratory connectivity, as defined by Webster et al. (2002). The reason is that we are interested in analysing the impact of conservation efforts of wintering habitats in any country *j* on the overall conservation benefits of the rest of the countries with wintering habitats that are also part of the migratory route. To do so, we need to consider whether a country's habitat is a potential substitute for habitats for metapopulations of the bird wintering in other countries. We introduce a spatial structure in the model that allows to examine the potential scenarios of migratory connectivity of the Golden-winged Warbler.

HON, NIC, CR, COL and VEN are the five main countries in which the Golden-winged Warbler spends most of its non-breeding season. Other countries in Latin America have been found to host the bird for shorter periods, i.e. as stopover sites. However, we limit our research by focusing on those five countries with the most relevant wintering sites.

At the present time the dynamics of migration and the migratory connectivity of the Golden-winged Warbler are unknown. Therefore, we consider three scenarios that capture potential types of migratory connectivity between breeding and non-breeding habitats. The different spatial scenarios that we consider depend on i) the location of countries on the migratory route, ii) the geographical distance between them, and iii) the assumptions that we impose with respect to their degree of migratory connectivity. For our approach we adopt a particular specification for our interpretation of distance (i.e. geographical distance) and also for our selected spatial structure (i.e. defined patterns to describe the degree of migratory connectivity). We adopt these two assumptions because together they form the simplest yet suitable structure to examine the spatial aspects for the case of the Golden-winged Warbler. However, the approach of our model is general and would work for arbitrary specifications of distances and spatial structures (see Alvarado-Quesada and Weikard 2015a).

Scenario 1 (S1): Weak migratory connectivity

Weak migratory connectivity occurs when individuals from a breeding population migrate to several different overwintering locations spread through the non-breeding range (Webster et al. 2002). For this scenario we assume that the global population of Golden-winged Warblers migrates to different overwintering locations throughout the LAC countries. We assume under Scenario 1 that wintering habits are substitutes throughout the region. Also, we assume that the weight of regional benefits is determined by the geographical distance between any two countries in *N*: the smaller the distance, the larger the impact of one country's conservation efforts on the regional benefits of the other countries (and vice versa). Hence, $\omega_{i,j}$ is inversely related to the normalised geographical distance between any two countries *i* and *j*.

Scenario 2 (S2): Strong migratory connectivity

Strong migratory connectivity occurs when most *'individuals from one breeding population move to the same non-breeding location to form a non-breeding population'* (Webster et al. 2002, p.78). For this scenario we make an explicit assumption that there are two subpopulations of Golden-winged Warblers: one subpopulation in the breeding habitat that migrates to winter locations in Central America (HON, NIC and CR), and another subpopulation in the breeding habitat that migrates to winter locations in the north of South America (COL and VEN). Therefore we express a strong migratory connectivity for the two subpopulations of Golden-winged warblers. In this scenario, countries obtain lower regional benefits from habitat conservation undertaken in countries belonging to a different geographical cluster, as they cannot act as substitutes of their wintering habitats. Countries belonging to the same geographical cluster obtain the same benefits as those stated for S1. Under Scenario 2 we adjust parameter $\omega_{i,j}$ to reflect the lower value of regional benefits from habitat conservation in countries from a different geographical cluster by multiplying the values of Scenario 1 by a scaling factor of 0.1 (see Table A5 in Appendix 5.3).

Scenario 3 (S3): Strong migratory connectivity in combination with complete habitat loss in one of the countries

Under S3 we assume that there is strong connectivity as in S2 and we assume that the wintering habitats in one of the five countries disappear completely. We consider this separately for all five countries, to assess the impact of complete habitat loss in one country on conservation efforts of the rest of the countries and consequently of the overall benefits of conservation.

The weight parameter matrices for $\omega_{i,j}$ of our connectivity scenarios can be found in Appendix 5.3. With the spatial structure of the model already defined, and given the available information per country, we calibrate our model parameters and proceed to perform a numerical analysis for the different spatial scenarios in the following section.

5.5 Numerical analysis

For our numerical analysis, we set a base model with parameter values that have been calibrated with the available data per country obtained from the ABC's databases. In addition to the standard coalition stability analysis, we allow the inclusion of an optimal transfer scheme to assess its impact on coalition formation. Furthermore, we examine coalition effectiveness by means of a welfare indicator. Table 5.1 shows the parameter values for the base model under analysis. The complete description of the calibration of the parameters is explained in detail in Appendix 5.2.

Table 5.1. Parameter values for the base model

	λ	ρ	c	δ	e_{min}	e_{max}	e_{BAU}	ψ_{BAU}	ψ_{max}
	thousand US$/ year	thousand US$/ ha per year	thousand US$/ ha^2 per year	1/ha^2	ha	ha	ha	—	—
HON	290.086	0.00385	0.0004500	2.31E-12	10	520,082	273,013	0.4831	0.6239
NIC	365.555	0.00385	0.0000169	2.23E-11	10	153,145	92,102	0.4390	0.5219
CR	146.518	0.00385	0.0009209	6.23E-12	10	255,994	143,025	0.3287	0.4082
COL	1,448.790	0.00385	0.0000005	3.43E-13	10	936,805	340,499	0.1790	0.3009
VEN	331.100	0.00385	0.0000040	1.31E-12	10	660,799	108,987	0.1725	0.5699
USA	-	14.21	-	-	-	-	-	-	-

Recall that in Section 5.4.1 we explained that the regional benefit parameter for USA (ρ_{USA}) is calibrated differently from the regional benefit parameters of the LAC. To assess regional benefits from conservation in USA we use results from Reaves et al. (1999) who provide estimates of WTP for species conservation of birds. More specifically, they estimate the WTP per year per US household to improve the chance of survival of the bird species population of the Red-cockaded Woodpecker (*Picoides borealis*) from 50 to 99%, and find a mean value of US$11/year per household. In our scenarios we have used this value as a starting point for the WTP for the protection of the Golden-winged Warbler. Since such estimates are debatable (Brouwer et al. 2008, Martín-López et al. 2008) we conduct a sensitivity analysis and also consider – for comparison – the case of a zero WTP. Using the WTP value of US$11/year per household, we calculate ρ_{USA} = 14.21. Further details of the calibration of ρ_{USA} are explained in Appendix 5.2.

5.5.1 Inclusion of an optimal transfer scheme

Transfers schemes are used to increase participation in an agreement by incentivising countries to join the coalition in a way that larger coalitions may satisfy internal stability conditions (Pavlova and de Zeeuw 2013). One or more members of the agreement are supposed to transfer part of their gains from conservation to other members of the coalition to incentivise membership.

In this game we apply an optimal sharing rule that guarantees that the coalition is stable when the coalition payoff (weakly) exceeds the sum of the outside option payoffs of its members (Weikard 2009). In the context of an international agreement of the open membership kind as ours, it is plausible to assume that no member of the coalition is worse off than as a singleton. The inclusion of transfers generally increases the chances of achieving larger stable biodiversity agreements (Winands et al. 2013).

5.5.2 Welfare indicator: the 'closing the gap index' (CGI)

To examine coalition effectiveness, we incorporate in our analysis a 'closing the gap index' (CGI). This is an indicator of the extent to which a coalition closes the gap between the aggregate payoff (or conservation effort) under no cooperation and the aggregate payoff under full cooperation (see Eyckmans and Finus 2006). The welfare CGI is defined as:

$$CGI^{\pi} = \frac{\pi^{E} - \pi^{NC}}{\pi^{FC} - \pi^{NC}} , \tag{5.10}$$

where

π^{E} is the aggregate payoff of the best coalition in equilibrium

π^{NC} is the aggregate payoff when there is no cooperation

π^{FC} is the aggregate payoff in the social optimum (full cooperation)

Notice that the index satisfies $0 \leq CGI^{\pi} \leq 1$. If CGI is equal to one the gap is fully closed and the sum of the payoff under the best coalition is identical to the global payoff under full cooperation. If CGI is zero the sum of the payoffs is identical to the sum of the payoffs if all players act as singletons.

In order to compare the success of the equilibrium coalition in terms of global conservation efforts of wintering habitat, we also make use of a global conservation index CGI^{E}. It is constructed analogous to the CGI^{π}:

$$CGI^{E} = \frac{E^{E} - E^{NC}}{E^{FC} - E^{NC}} . \tag{5.11}$$

5.6 Results

In reporting the results, we consider different levels of US households' WTP per year for the conservation of the Golden-winged Warbler, from a zero WTP to US$11/year per household. The intermediate values we consider are: US$0.11/year per household and US$1.1/year per household (respectively 1% and 10% of the estimate of Reaves et al. 1999). We do so for two scenarios: weak (S1) and strong (S2) connectivity (see Table 5.2).

We find that for all cases under the two connectivity scenarios without transfers, no stable coalitions exist. For all cases, three countries show higher conservation efforts when acting as singletons as when compared to those in the BAU scenario: NIC, COL and VEN. As for HON and CR, conservation efforts when acting as singletons are equal to those in the BAU scenario, i.e. they do not undertake additional conservation efforts when there is an associated cost.

When transfers are allowed, we do find stable coalitions. The results for conservation efforts and global payoff for the stable coalition with the best payoff for each of the cases are presented below. From Table 5.2 we observe that under both scenarios all countries undertake higher efforts of conservation if the value for ρ_{USA} increases (as a result of higher WTP values). If we compare these values to the maximum values of habitat endowment e_{max} per country in Table 5.1, we find that NIC and COL are protecting their maximum habitat endowment when the WTP value is of US$11/year (under both S1 and S2). On the other hand, the increase in conservation efforts of HON and CR is relatively lower when compared to that of the rest of LAC: when ρ_{USA} varies from 0 to 14.21, HON and CR increase their conservation efforts by only 3% and 1.7%, respectively.

Table 5.2. Conservation efforts per LAC for the best coalition under S1 and S2 for different parameter values of ρ_{USA} [a/]

	Conservation efforts (ha)							
	S1				**S2**			
WTP (US$/year per household)	0	0.11	1.1	11	0	0.11	1.1	11
ρ_{USA}	0	0.14	1.42	14.21	0	0.14	1.42	14.21
HON	**273,013**	**273,105**	**273,837**	**281,220**	**273,013**	**273,100**	**273,832**	**281,215**
NIC	92,131	**94,176**	**110,267**	**153,145**	92,131	**94,056**	**110,151**	**153,145**
CR	**143,025**	**143,052**	**143,265**	**145,404**	**143,025**	**143,051**	**143,264**	**145,402**
COL	341,091	**340,499**	**626,426**	**936,805**	341,091	**340,499**	**622,925**	**936,805**
VEN	109,047	**108,987**	**151,530**	**660,631**	109,047	**108,987**	**150,924**	**660,383**

a/ Numbers in bold refer to members of the stable coalition with the best payoff.

Regarding coalition stability, results in Table 5.3 show that the maximum size of a stable coalition under both spatial structure S1 and S2 is $s^* = 6$ which reflects the grand coalition (i.e. LAC together with USA). For the case when there is a zero WTP per US household, the stable coalition with the best payoff for both scenarios is composed of two countries: HON and CR.

For all cases under scenarios S1 and S2, we observe that global conservation efforts and global payoff increase systematically as ρ_{USA} goes up. Yet, the variations in the global payoff when assuming higher values for ρ_{USA} are relatively larger than those in the global conservation efforts. For instance, when we consider a WTP value per US household of US$11/year, we observe that the conservation efforts under full cooperation lead to a global payoff of almost US$5.8 billion/year. According to our calibration that considers a WTP of US$11/year perceived by US households, if the population of Golden-winged Warblers were stabilised – that is, if 620,000 birds were protected – the total US benefits would be of US$3.8 billion/year (see Appendix 5.2). For such high payoffs under full cooperation the number of protected birds is considerably larger than the one required to stabilise the population. The reason is that our calculation considers that each protected bird has the same value, even beyond what is required to stabilise the population. Note that Reaves et al. (1999) assess a WTP value to improve the chance of survival of the bird population from 50% to 99%, and not the WTP of each additional bird protected. One of the main points of discussion in the valuation literature is the *'insensitivity of WTP values to the magnitude of the proposed level of protection (…) and the absence of decreasing marginal WTP for additional protection'* (Brouwer et al. 2008, p.576). We acknowledge that a value per additional protected bird would have been a better proxy for our study. Yet, given the lack of information regarding the valuation of the Golden-winged Warbler, we considered Reaves et al. (1999)'s WTP value as the best value that we can currently use for our study.

Table 5.3. Global conservation efforts and global payoff for the best coalition under S1 and S2 for different parameter values of ρ_{USA}

		S1			S2		
ρ_{USA}	*WTP* (US$/ year per household)	**Stable coalition with best payoff**	**Global conservation efforts** (ha)	**Global payoff** (thousand US$/year)	**Stable coalition with best payoff**	**Global conservation efforts** (ha)	**Global payoff** (thousand US$/year)
0	*0*	HON+CR	958,307	4,726	HON+CR	958,307	2,768
0.14	*0.11*	LAC+USA	959,820	29,659	LAC+USA	959,693	27,695
1.42	*1.1*	LAC+USA	1,305,325	303,727	LAC+USA	1,301,095	300,625
14.21	*11*	LAC+USA	2,177,205	5,767,549	LAC+USA	2,176,950	5,759,749

Taking this into consideration, we believe that a WTP value of US$1.1/year is a more realistic approximation of the value that US households are willing to pay to stabilise the birds' population. Hence, we use the WTP of US$1.1/year for the rest of our calculations. The detailed results of the scenarios of weak and strong connectivity with transfers, considering a WTP of US$1.1/year per household, are presented below.

Results in Table 5.4 show that full cooperation is achieved under the weak connectivity scenario with transfers included and with a WTP value of US$1.1/year. We find that USA transfers a sum of US$55 million/year to the rest of the countries to stabilise the grand coalition.

As for the model with transfers under spatial structure S2, we observe in Table 5.5 that also the social optimum is achieved as the best coalition is composed of all six players. To stabilise the coalition, USA transfers US$53 million/year to the LAC and end up with revenues of around US$297 million per year.

Table 5.4. Coalition stability and CGI under weak connectivity (S1) with transfers for a WTP value of US$1.1/year per US household

	Conservation efforts under BAU scenario (ha)	**Conservation efforts under best coalition** (ha)	**Payoff under best coalition before transfers** (thousand US$/year)	**Transfers** [a/] (thousand US$/year)	**Payoff under best coalition after transfers** (thousand US$/year)
HON	273,013	273,837	830	-501	1,330
NIC	92,102	110,267	-4,078	-5,797	1,720
CR	143,025	143,265	1,345	-294	1,639
COL	340,499	626,426	-39,622	-40,944	1,322
VEN	108,987	151,530	-5,819	-7,461	1,643
USA	-	-	351,071	54,998	296,073
Number of stable coalitions when transfers are allowed (out of 64)					1
Size of stable coalitions (s^*)					6
Best coalition					**LAC+USA**
Global conservation efforts under best coalition (ha)					1,305,325
CGI^E (%)					**100**
Global payoff under best coalition (thousand US$/year)					303,727
CGI^π (%)					100

a/ Positive transfers imply that countries give away part of their payoff to others, while negative transfers imply that countries receive money from others.

Table 5.5. Coalition stability and CGI under strong connectivity (S2) with transfers for a WTP value of US$1.1/year per US household

	Conservation efforts under BAU scenario (ha)	**Conservation efforts under best coalition** (ha)	**Payoff under best coalition before transfers** (thousand US$/year)	**Transfers** [a/] (thousand US$/year)	**Payoff under best coalition after transfers** (thousand US$/year)
HON	273,013	273,832	265	-401	667
NIC	92,102	110,151	-4,599	-5,646	1,047
CR	143,025	143,264	731	-190	921
COL	340,499	622,925	-39,296	-39,835	538
VEN	108,987	150,924	-6,289	-7,141	852
USA	-	-	349,813	53,213	296,599
Number of stable coalitions when transfers are allowed (out of 64)					1
Size of stable coalitions (s^*)					6
Best coalition					**LAC+USA**
Global conservation efforts under best coalition (ha)					1,301,095
CGI^E (%)					**100**
Global payoff under best coalition (thousand US$/year)					300,625
CGI^π (%)					100

a/ Positive transfers imply that countries give away part of their payoff to others, while negative transfers imply that countries receive money from others.

When we compare the results of these two scenarios, we find higher conservation efforts and higher global payoff in the full cooperative case under weak connectivity. Also, for both scenarios we observe that HON and CR are the countries that undertake the least additional effort with respect to their BAU scenarios. This is a result of the relatively higher costs of conservation of these two countries when compared to those of NIC, COL and VEN. Additional conservation efforts are higher where it is relatively cheaper to protect wintering habitats.

When we study the inclusion of transfers in the scenario of strong connectivity with complete habitat loss in one of the countries (S3), we find that, as in the previous scenarios, the size of the best coalition in all five cases is always $s^* = 6$. From the five cases of complete habitat loss in one of the countries, the full cooperative case that achieves the highest global payoff is the one showing habitat loss in VEN (US$280 million), whereas the case of habitat loss in HON shows the lowest global payoff under full cooperation (US$178 million). In terms of global conservation efforts, the case of habitat loss in NIC achieves the highest conservation (1,190,944 ha), while the case of habitat loss in COL results in the lowest conservation (678,171 ha). Results of the case of habitat loss in COL are presented in Table 5.6. This table is an example of the outcome of complete habitat loss in one of the countries with wintering habitat. The tables with the results on coalition stability for the rest of the cases showing complete habitat loss can be found in Appendix 5.4.

Table 5.6. Coalition stability and CGI under strong connectivity with complete habitat loss in COL (S3) with transfers for a WTP value of US$1.1/year per US household

	Conservation efforts under BAU scenario (ha)	**Conservation efforts under best coalition** (ha)	**Payoff under best coalition before transfers** (thousand US$/year)	**Transfers** [a/] (thousand US$/year)	**Payoff under best coalition after transfers** (thousand US$/year)
HON	273,013	273,832	214	-329	542
NIC	92,102	110,151	-4,653	-5,534	882
CR	143,025	143,264	675	-91	766
COL	-	-	199	-10	209
VEN	108,987	150,924	-6,884	-7,023	138
USA	-	-	233,565	12,986	220,578
Number of stable coalitions when transfers are allowed (out of 64)					1
Size of stable coalitions (s^*)					6
Best coalition					**LAC+USA**
Global conservation efforts under best coalition (ha)					678,171
CGI^E (%)					**100**
Global payoff under best coalition (thousand US$/year)					223,115
CGI^π (%)					100

a/ Positive transfers imply that countries give away part of their payoff to others, while negative transfers imply that countries receive money from others.

In Table 5.6 we observe that although COL cannot undertake any conservation in its territory, it still perceives regional benefits from conservation taking place in the rest of the countries, even before transfers are implemented. As a result, when COL is part of the coalition, the rest of the members of the coalition (except USA) increase their conservation efforts as compared as when COL would be an outsider. Member countries maximise the net benefits of the coalition, and when transfers take place, no country has an incentive to individually deviate from the coalition.

We observe in this case (and in the rest of the cases of complete habitat loss in one country) that USA transfers part of its payoff to the other LAC to stabilise the coalition, including the country with complete habitat loss. The optimal transfer rule that we implement in our game is such that when the coalition payoff (weakly) exceeds the sum of the outside options of the members, these outside options are covered to guarantee stability. If all outside options are covered and there is a residual, then the residual is proportionately distributed among the members of the coalition. In the way that our transfer rule is formulated, COL receives a share of this residual due to the fact that it is a member of the coalition. However, the residual could be distributed in any way and there is no need to transfer money to a country like COL, as it does not require additional incentives to stay in.

5.7 Discussion and conclusions

In this study we develop a coalition stability analysis to examine the viability of an IEA for conservation of the wintering habitat of the Golden-winged Warbler. We do so by calibrating the parameters of our model with available information regarding the current status of wintering areas for the Golden-winged Warbler in LAC. Information to conduct this calibration was obtained from an extended study carried out by the ABC as part of the development of the Golden-winged Warbler Wintering Grounds Conservation Plan.

Our study is unique in that it relates regional biodiversity benefits of countries to the spatial dimension of the wintering habitats of the Golden-winged Warbler based on geographical distance and migratory connectivity. Moreover, it examines the inclusion of a player that has no possibility to undertake conservation efforts within its territory, but that still reaps positive spillovers from conservation in the rest of the countries.

We first set our model in which USA perceives regional benefits of conservation in wintering habitats located in LAC. To calibrate the regional benefit parameter of USA, we use the information of US households having a WTP value of US$11/year to improve the chance of survival of a bird species' population from 50% to 99%. We then perform a sensitivity analysis for different WTP values per US household per year, ranging from a zero WTP to US$11/year per household. We find that, for all three spatial scenarios that we study, and under all WTP values, no stable coalitions are formed in the absence of a transfer scheme. When transfers are allowed, however, several stable coalitions are reached, including the grand coalition.

For the different regional benefit parameter values of USA, and under both scenarios of weak and strong migratory connectivity, we find that the stable coalition with the best payoff reaches the social optimum (i.e. $s^* = 6$). The only exception for both spatial scenarios is when USA does not perceive any benefits from conservation, in which the stable coalition with the best payoff is composed of two countries (HON and CR). Higher regional benefits of conservation of USA lead to higher conservation efforts in all LAC. When the WTP per US household is US$11/year, NIC and COL achieve full protection of their habitat endowment under both S1 and S2.

Global payoff of the stable coalition also increases with higher regional benefit parameter values of USA. The variations of global payoff are considerably larger when compared to variations in conservation efforts. This result is an artifact of the assumption of our model that there is linearity between the number of protected species and the benefits of USA from this conservation. The reason is that our reference WTP value does not represent the willingness of US households to pay for each additional bird protected; instead it represents their willingness to contribute to improve the chance of survival of the bird population as a whole from 50% to 99%. Hence, each additional bird is assumed to have the same value, even after the population has been stabilised. After taking this into consideration, we opted to report in more detail about the WTP value of US$1.1/year per US household in our coalition stability analysis as a more reasonable approximation of the benefits that US households obtain from the Golden-winged Warbler conservation.

For this analysis we find that, in the full cooperative case achieved under the weak connectivity scenario (S1), USA transfers to the LAC to stabilise the coalition amount to US$55 million per year. We know that USA has no wintering habitat to protect. However, benefits from regional conservation perceived by USA – derived from the stabilisation of the bird's population – are high enough for them to cover the outside options of all other coalition members. The total payoff of this coalition is of US$303 million per year, resulting in 1.305 million protected hectares. Under the strong connectivity scenario (S2), both conservation efforts and global payoff for the full cooperative case are lower than those under weak connectivity (1.301 million ha and US$300 million/year, respectively). In both scenarios, HON and CR undertake low additional efforts of conservation with respect to their BAU scenario due to their relatively high costs of conservation. We observe that transfers are mainly allocated to those countries where it is relatively cheaper to protect wintering habitats.

When we examine the cases in which there is complete habitat loss in one of the wintering countries, we find that the stable coalition with the best payoff is also composed of all six countries of the game. When a country with complete habitat loss joins the coalition, the other members of the coalition increase their conservation efforts to jointly maximise the payoff of the coalition.

Without any transfer from USA, countries facing complete habitat loss have no incentive to leave the coalition under full cooperation, as their outside option is lower. Due to an artifact of the transfer rule that we implement in our model, countries with complete habitat loss also receive a transfer from USA because they are part of the coalition. Yet, in reality there is no need to execute this transfer as they do not have incentives to deviate from the coalition.

In general, we conclude that with i) a positive WTP of US households to improve the chance of survival of the population of Golden-winged Warblers and ii) the implementation of a transfer scheme, there is scope for a conservation agreement between LAC and USA to effectively increase conservation efforts.

The model presented in this chapter has been designed to analyse coalition stability for an agreement to preserve wintering habitats of the Golden-winged Warbler. Even though our analysis has focused on one particular species, our model can be used to analyse coalition stability of an IEA for any migratory species that can adjust to the spatial migratory pattern we suggest in the study.

Calibration of the parameters of our model could be furthered improved by acquiring more accurate information on market land values per country, as well as by carrying out additional valuation studies for other type of local benefits e.g. WTP values for the species, ecotourism activities, and bird-friendly agri-environment schemes.

We have assumed independent occurrence probabilities among countries to simplify our analysis. An extension of the model would be to perform a metapopulation assessment where occurrence probabilities among countries are interdependent. Also, this study accounts only for male occurrence probability values due to the lack of information regarding female occupancy. In the future the model can be enriched by including the relevant data regarding female occurrence probability in the winter range. Moreover, although the scenarios defined for our spatial structure are based on hypothesis regarding the migratory connectivity of this species, we hope that in the near future the model can be further adjusted if more empirical data becomes available.

We believe this model to be an effective tool to assess countries' incentives to participate in an IEA to protect the habitat of a migratory species. Further research on the actual dynamics of the migratory connectivity of the species would allow for more robust results to shed light on the main conservation priorities of the region.

5.8 Appendices

Appendix 5.1. Focal areas per country

In total we consider 45 focal areas for the Latin American region (HON= 9, NIC= 8, CR= 16, COL= 8, and VEN= 4). Table A1 shows the size and average occurrence probability per focal area for all five countries.

Table A1. Size and occurrence probability per focal area [a/]

Name of focal area per country	Area (in ha)	Average occurrence probability
HONDURAS		
Sierra de Agalta and El Boquerón	94,698	0.5516
El Carbón	58,663	0.6239
Pico Bonito	115,471	0.4064
Merendón - Water Production Area	37,866	0.2577
La Muralla	26,436	0.5626
Cusuco	19,052	0.2678
Pico Pijol	23,767	0.4299
Botaderos Mountain	108,305	0.5468
Texiguat	35,824	0.4548
Total area	**520,082**	
Average occurrence probability BAU scenario		**0.4831**
NICARAGUA		
Cerro el Arenal	1,504	0.4689
Cerro Datanlí - El Diablo	6,167	0.5219
Cerro Saslaya	66,910	0.4095
Macizo de Peñas Blancas	12,196	0.4803
Cordillera Dipilto y Jalapa	33,309	0.4212
Cerro Kilambé	13,308	0.5217
Yucul	5,886	0.3560
Corredor El Jaguar - Yalí	13,865	0.5032
Total area	**153,145**	
Average occurrence probability BAU scenario		**0.4390**
COSTA RICA		
Monteverde - Pocosol	3,036	0.2807
Monteverde - San Luis	3,523	0.1332
Monteverde - Cedral	2,620	0.3087
Braulio Carrillo - Cinchona - Sarapiquí	18,066	0.2582
Braulio Carrillo - Cinchona - Poás	19,789	0.3793
Braulio Carrillo - Cinchona - Río Cuarto	56,565	0.3766
Turrialba - Guayabo	4,544	0.3882
Turrialba - Cachí - Orosí	3,266	0.3683
Escazú - Acosta (1)	22,698	0.4082

Name of focal area per country	Area (in ha)	Average occurrence probability
Escazú - Acosta (2)	7,508	0.1703
Escazú - Acosta (3)	11,499	0.2392
Escazú - Acosta (4)	2,623	0.3913
Talamanca - Caribe	33,766	0.3874
Talamanca - Coto Brus (1)	48,057	0.2900
Talamanca - Coto Brus (2)	14,838	0.1918
Talamanca - Coto Brus (3)	3,596	0.3848
Total area	**255,994**	
Average occurrence probability BAU scenario		**0.3287**
COLOMBIA		
Magdalena: Sierra Nevada de Santa Marta	132,991	0.3009
Bolívar: Serranía de San Lucas	136,423	0.1699
Antioquia: Jericó-Támesis	31,105	0.2059
Antioquia: Cuenca alta del Río Porci-Municipio Anorí	14,799	0.1957
Antioquia: La Romera-Sabaneta	40,840	0.2179
PNN Los Nevados - Zona de Amortiguación	331,696	0.1379
Santander/Boyacá: Serranía de Los Yarigüíes	107,566	0.1111
Serranía del Perijá	141,385	0.2021
Total area	**936,805**	
Average occurrence probability BAU scenario		**0.1790**
VENEZUELA		
Serranía La Perijá	189,365	0.1889
La Azulita	86,000	0.5699
Altamira	136,113	0.0496
Tachira	249,321	0.0900
Total area	**660,799**	
Average occurrence probability BAU scenario		**0.1725**

a/ The focal areas that we considered for the study where those that appeared on both the country factsheets elaborated for the threat analysis of the Golden-winged Warbler (ABC 2014 and ABC 2014a) and on the spreadsheet for occupancy calculations (ABC 2015). Those focal areas that were defined in only one of the two documents were not considered.

Appendix 5.2. Variables and parameters of the model

Table A2 shows an overview of the variables and parameters of our spatial model for an IEA for conservation of wintering habitats of the Golden-winged Warbler.

Table A2. Summary table of variables and parameters of the model

Parameter	Type	Notation	Unit
Local benefits of habitat conservation	parameter	λ	thousand US$/year
Regional benefits of habitat conservation	parameter	ρ	thousand US$/ha per year
Local costs of habitat conservation	parameter	c	thousand US$/ha^2 per year
Weighted value of habitat conservation in other countries	parameter	$\omega_{i,j}$	—
Scaling parameter of parabolic occurrence probability function	parameter	δ	1/ha^2
Conservation efforts	variable	e	ha
Minimum viable conservation for the Golden-winged Warbler to occur	scalar	e_{min}	ha
Maximum conservation possible (habitat endowment)	parameter	e_{max}	ha
Conservation efforts at no cost	parameter	e_{BAU}	ha
Occurrence probability of the Golden-winged Warbler	variable	ψ	—
Occurrence probability in the BAU scenario	parameter	ψ_{BAU}	—
Maximum occurrence probability (achieved when e_{max} is protected)	parameter	ψ_{max}	—

Below we state the description of the calibration of the main parameters, as well as the sources of information used for such calibrations.

Local benefits of habitat conservation (λ)

We make use of the number of hectares per country allocated to coffee production, and we multiply them by the value saved by coffee farmers due to pest control services in the presence of the Golden-winged Warbler. Karp et al. (2013) state that farmers could save between US$75-US$310 per hectare over a year's harvest due to pest control services of foraging birds. We assume that savings associated to the Golden-winged Warbler correspond to 10% of the average savings per ha over a one-year harvest, i.e. 0.1*[(75+310)/2] = US$19.25/ha per year. We then multiply this value by the number of ha for coffee production per country. As a result, we obtain the parameter values for local benefits of habitat conservation per country (see Table 5.1 in Section 5.5, column 2). This parameter reported in the model is measured in thousand US$/year.

Regional benefits of habitat conservation (ρ)

For simplicity of the model, we consider plausible to assume that regional benefits of habitat conservation are a fraction of local benefits of conservation. Previously we calculated the average savings per hectare over a one-year harvest due to the presence of the Golden-winged Warbler in US$19.25/ha per year. For our study we assume that regional benefits of the LAC correspond to 20% of the local ones, which are equal to 0.2*19.25 = US$3.85/ha per year for all countries (see Table 5.1 in Section 5.5, column 3). The parameter reported in the model is measured in thousand US$/ha per year.

The calibration of the regional parameter for USA is different. USA has no wintering habitat to preserve. Yet, it benefits from the additional birds resulting from the conservation of wintering habitats in LAC. According to PIF Science Committee (2013) the current population of Golden-winged Warblers is estimated in around 410,000 birds, out of which 300,000 breed in USA. We assume that US households assign a value to this global bird population of the Golden-winged Warbler. This population is facing a long-term decline. The population goal established by the Golden-winged Warbler Working Group is to restore the current estimated population of breeding individuals into approximately 620,000 birds, which is similar to the population that existed in the 1980s (Roth et al. 2012). In order to estimate the value given by USA to preserve this additional population of 210,000 birds, we use Reaves et al. (1999)'s WTP estimate as a proxy for our model. Due to the lack of a WTP study related to the Golden-winged Warbler, and of any study that deals with the valuation of a species of the order Passeriformes, we make use of the mean value of a species from the order Piciformes, namely the Red-cockaded Woodpecker (*Picoides borealis*).

Reaves et al. (1999) conducted a study in which they asked US household's representatives for their WTP to improve the chance of survival of the Red-cockaded woodpecker population from 50% to 99%. The mean WTP in US$/year for the three different type of question formats was of US$11/year per household. This is the representative WTP value that we used as a starting point for our study.

Therefore, we assume that US households are willing to pay US$11/year to improve the chance of survival of the bird population of the Golden-winged Warbler from 50% to 99%. According to Roth et al. (2012), increasing the population of the Golden-winged Warbler by about 50% would bring the population to stable numbers. We infer from this information that increasing the bird population to 620,000 birds is equivalent to approaching the chance of survival of the population to a 50%-99%. Hence, with a total of 117 million households in USA (U.S. Census Bureau 2012), we assume that the amount of money that US households are willing to pay to stabilise the population of Golden-winged Warblers is of 11*117.5= US$1.292 billion.

We assume linearity in the relationship between number of birds and benefits in US$/year for USA. Since the benefits perceived from protecting 210,000 birds (the additional number of birds required to stabilise the population) are equal to US$1.292 billion/year, the slope of this linear function is α = (1,292,500,000/210,000) = 6,154.76.

In the absence of a conservation agreement, countries still undertake conservation efforts at no cost, which lead to the BAU occurrence probability values per LAC. We associate these conservation efforts and occurrence probabilities to the 410,000 bird species that are currently known to exist. In line with our previous calculations, the benefits of regional habitat conservation for USA under the BAU scenario (i.e. the protection of 410,000 birds) are equal to 410,000*α = US$2.5 billion/year. If we adjust equation (5.2) to describe the case of USA, we obtain:

$$R_{USA} = \rho_{USA}\left[\sum_{j \in N_{-USA}} \omega_{USA,j} \cdot \psi_j \cdot \left(e_{BAU_j}\right)\right] \qquad \forall\, i, j \in N, \tag{5.2'}$$

First let's take $\sum_{j \in N_{-USA}} \omega_{USA,j} \cdot \psi_j \cdot \left(e_{BAU_j}\right) = [*]$. We stated before that we consider the product of e_j and ψ_j as an indicator of the normalised size of the population of the bird species in country j. Since we consider all countries with wintering habitats in the BAU scenario in equation (5.2'), we assume that [*] is related to the 410,000 existing birds. Calculating the value of [*] with the parameters of our model (see Table A1 in Appendix 5.1), we obtain that [*] = 177,571 ha. Hence, in order to obtain the parameter value of ρ_{USA}, we calculate R_{USA}/ [*] = US$14,211/ha per year. To simplify our model calculations, we take the regional parameter value of USA in thousand US$/ha per year. As a result, ρ_{USA}= 14.21.

Local costs of habitat conservation (c)

To estimate the cost parameter values per country, we use the information of opportunity cost of land protection per focal area from the ABC's factsheets. We associate the hectares related to the BAU scenario per country as being protected at no cost. We then associate the maximum conservation possible per country j, e_{max_j} (in hectares) to the highest net present market value attributed to a hectare in that same country (in US$/ha per year). Through these calculations we obtain the marginal costs of one additional hectare preserved c per country j. This parameter value is measured in thousand US$/ha^2 per year (see Table 5.1 in Section 5.5, column 4). Note that the factsheets from the focal areas of HON had no information regarding the market value of land. For this reason, we based our parameter estimate for HON on experts opinion and assigned it a value between the cost parameter value of CR and NIC (i.e. we assume that opportunity costs of protected land in HON are lower than in CR but higher than in NIC).

Average occurrence probability in the BAU scenario (ψ_{BAU})

Average occurrence probability in the BAU scenario per country j is calculated as follows:

$$\psi_{BAU_j} = \frac{\sum_{k \in J}\left(e_k * \psi_k\right)}{\sum_{k \in J} e_k} \tag{5.12}$$

where J is the set of focal areas within country j (see Table A1 for size and occurrence probability of focal areas per country). Average occurrence probability ψ_{BAU} per country j can be found in Table 5.1 (Section 5.5, column 9).

Conservation efforts at no cost (e_{BAU})

In order to calculate the conservation efforts at zero cost, we first set the parameter values for the parabolic function of occurrence probability. We assume that the maximum conservation possible per country e_{max} is associated to the highest occurrence probability value that was found per country, ψ_{max} (this is the highest occurrence probability value found in the factsheets from the focal areas per country). In other words, we assume that full conservation of wintering habitat in a country leads to the maximum occurrence probability value reported in that country.

Then, together with the scaling parameter δ, we deduct the level of conservation effort associated to the average occurrence probability in the BAU scenario ψ_{BAU}. These conservation efforts are stated in Table 5.1 (Section 5.5, column 8) and are used as a reference throughout our study.

Table A3. Data sources for calibration of the model parameters.

Calibrated parameter	Data used for parameter estimation	Source
Local benefits of habitat conservation (λ) (thousand US$/year)	Value saved by coffee farmers due to pest control service of foraging birds (US$/ha per year)	Karp et al. (2013) Karp et al. (2014)
	Number of ha per country allocated to coffee production (ha)	Country factsheets for ABC's Golden-winged Warbler Wintering Grounds Conservation Plan ABC threat analysis for the Golden-winged Warbler (ABC 2014 and ABC 2014a)
Regional benefits of habitat conservation (ρ) for the five countries with wintering habitats (thousand US$/ha per year)	Share of the value saved by coffee farmers due to pest control service of foraging birds (US$/ha per year)	Karp et al. (2013) Karp et al. (2014)
	WTP value for a representative bird species (US$/year)	Reaves et al. (1999)
Regional benefits of habitat conservation (ρ) for USA (thousand US$/ha per year)	Average occurrence probability values and conservation efforts (ha) for the BAU scenario	Country factsheets for ABC's Golden-winged Warbler Wintering Grounds Conservation Plan ABC threat analysis for the Golden-winged Warbler (ABC 2014 and ABC 2014a)
	WTP value for a representative bird species (US$/year)	Reaves et al. (1999)
	Number of households in USA	U.S. Census Bureau (2012)
	Estimated population of Golden-winged Warblers in USA (birds/year)	Roth et al. (2012) PIF Science Committee (2013)

Calibrated parameter	Data used for parameter estimation	Source
Local costs of habitat conservation (***c***) (thousand US$/ha2 per year)	Maximum conservation possible e_{max} per country (i.e. habitat endowment per country) (ha) Conservation efforts in the BAU scenario (ha) Conservation costs per hectare (if forested and if cleared) (ha) GDP growth rate per country (2010-2014) to calculate NPV of land per country	Country factsheets for ABC's Golden-winged Warbler Wintering Grounds Conservation Plan ABC threat analysis for the Golden-winged Warbler (ABC 2014 and ABC 2014a) World Development Indicators, World Bank (2015)
Weighted value of habitat conservation in other countries $\omega_{i,j}$	Geographical distances (in km) between any two countries from the set of countries *N*. As starting point to calculate distances we used the city that was closest to the conglomerate of focal areas in that country, i.e. San Pedro Sula (HON), Matagalpa (NIC), San José (CR), Bogotá (COL) and Maracaibo (VEN). For USA we use Minnesota as our reference city	Daft Logic Distance Calculator (Google Maps 2015)
Scaling parameter of parabolic occurrence probability function (δ) (1/ha2)	Value obtained when we assume that, in equation (5.5), the conservation efforts *e* per country are equal to the maximum conservation possible e_{max} per country (hence, that the occurrence probability value is at its maximum ψ_{max})	Country factsheets for ABC's Golden-winged Warbler Wintering Grounds Conservation Plan ABC threat analysis for the Golden-winged Warbler (ABC 2014 and ABC 2014a)
Maximum conservation possible (habitat endowment) (e_{max}) (ha)	Sum of area of all focal areas per country (ha)	Country factsheets for ABC's Golden-winged Warbler Wintering Grounds Conservation Plan ABC threat analysis for the Golden-winged Warbler (ABC 2014 and ABC 2014a)
Average occurrence probability in the BAU scenario (ψ_{BAU})	The average occurrence probability per country before any conservation agreement according to current occurrence probability and size of focal areas per country	Country factsheets for ABC's Golden-winged Warbler Wintering Grounds Conservation Plan ABC threat analysis for the Golden-winged Warbler (ABC 2014 and ABC 2014a)
Conservation efforts at no cost (e_{BAU}) (ha)	Conservation efforts calculated to match the average occurrence probability in the BAU scenario, ψ_{BAU} according to the specification of the occurrence probability function (see equation (5.5))	Country factsheets for ABC's Golden-winged Warbler Wintering Grounds Conservation Plan ABC threat analysis for the Golden-winged Warbler (ABC 2014 and ABC 2014a)
Maximum occurrence probability (ψ_{max})	Maximum occurrence probability value registered in a country	Country factsheets for ABC's Golden-winged Warbler Wintering Grounds Conservation Plan ABC threat analysis for the Golden-winged Warbler (ABC 2014 and ABC 2014a)

Appendix 5.3. Spatial structures: weight parameter matrices

We start by assuming that the geographical distance between any country and itself is $d_{i,i} = 0$; hence, we do not assign a value to the weight parameter of any country i and itself. Also, we assume that the larger the geographical distance between country i and j, the lower the value of the weight parameter $\omega_{i,j}$.

For simplicity of the model, we normalise the values of the weight parameters for all three scenarios between zero and one, i.e. $0 < \omega_{i,j} < 1$. In order to do so, we first obtain the distances between countries in kilometres. We divide these values by 1.00E+4 to obtain the values of $d_{i,j}$, and we then proceed to calculate $\omega_{i,j} = 1 - d_{i,j}$. The weight parameter matrix for the weak connectivity scenario (S1) is presented below:

Table A4. Weight parameter matrix for S1 (Weak connectivity)

	HON	NIC	CR	COL	VEN	USA
HON	-	0.9633	0.9246	0.8057	0.8142	0.6470
NIC	0.9633	-	0.9612	0.8410	0.8423	0.6149
CR	0.9246	0.9612	-	0.8751	0.8633	0.5782
COL	0.8057	0.8410	0.8751	-	0.9288	0.4920
VEN	0.8142	0.8423	0.8633	0.9288	-	0.5437
USA	0.6470	0.6149	0.5782	0.4920	0.5437	-

For the strong connectivity scenario (S2), we assume that there are two subpopulations of Golden-winged Warblers: i) those migrating to wintering locations in Central America (HON, NIC and CR); and ii) those migrating to wintering locations in the north of South America (COL and VEN). We then assume that countries that do not belong to the same geographical cluster cannot act as substitutes of wintering habitats. Therefore, benefits related to conservation efforts undertaken in countries from a different cluster are lower than those from countries within the same cluster.

To reflect this case, we first take the weight parameter matrices from (S1), and we proceed to multiply the original values of $\omega_{i,j}$ by 0.1 if i and j belong to different geographical clusters. The resulting weight parameter matrix for the strong connectivity scenario (S2) is:

Table A5. Weight parameter matrix for S2 (Strong connectivity)[a/]

	HON	NIC	CR	COL	VEN	USA
HON	-	0.9633	0.9246	0.0806	0.0814	0.6470
NIC	0.9633	-	0.9612	0.0841	0.0842	0.6149
CR	0.9246	0.9612	-	0.0875	0.0863	0.5782
COL	0.0806	0.0841	0.0875	-	0.9288	0.4920
VEN	0.0814	0.0842	0.0863	0.9288	-	0.5437
USA	0.6470	0.6149	0.5782	0.4920	0.5437	-

a/ Values corresponding to $\omega_{US,j}$ remain equal to the case parameter matrix for S1 because USA is not part of any of the geographical clusters that we consider in the strong connectivity scenario S2.

For the scenario of strong connectivity with complete habitat loss in one of the countries (S3), we use the same matrix from the (S2) case, namely that of Table A5. The difference is that, for each individual case, we assume the loss of habitat by considering no habitat endowment in the country, which leads to no occurrence probability and no conservation efforts in the BAU scenario. If we consider e.g. that HON loses its entire wintering habitat, we express this in the model by assuming $e_{max_{HON}} = 0$, and therefore it leads to $e_{min} = e_{BAU\ HON} = \psi_{max_{HON}}$.

Appendix 5.4. Results on coalition stability for the base model under strong connectivity and habitat loss (S3) of one of the LAC

Table A6. Coalition stability and CGI under strong connectivity with complete habitat loss in HON (S3) with transfers for a WTP value of US$1.1/year per US household

	Conservation efforts under BAU scenario (ha)	**Conservation efforts under best coalition** (ha)	**Payoff under best coalition before transfers** (thousand US$/year)	**Transfers** [a/] (thousand US$/year)	**Payoff under best coalition after transfers** (thousand US$/year)
HON	-	-	427	-143	570
NIC	92,102	110,151	-5,090	-5,625	535
CR	143,025	143,264	259	-157	416
COL	340,499	622,925	-39,338	-39,898	560
VEN	108,987	150,924	-6,331	-7,214	883
USA	-	-	228,048	53,036	175,011
Number of stable coalitions when transfers are allowed (out of 64)				2	
Size of stable coalitions *(s^*)*				2 and 6	
Best coalition				**LAC+USA**	
Global conservation efforts under best coalition (ha)				1,027,263	
CGI^E (%)				**100**	
Global payoff under best coalition (thousand US$/year)				177,975	
CGI^π (%)				100	

a/ Positive transfers imply that countries give away part of their payoff to others, while negative transfers imply that countries receive money from others.

Table A7. Coalition stability and CGI under strong connectivity with complete habitat loss in NIC (S3) with transfers for a WTP value of US$1.1/year per US household

	Conservation efforts under BAU scenario (ha)	**Conservation efforts under best coalition** (ha)	**Payoff under best coalition before transfers** (thousand US$/year)	**Transfers** [a/] (thousand US$/year)	**Payoff under best coalition after transfers** (thousand US$/year)
HON	273,013	273,832	69	-368	437
NIC	-	-	731	-132	863
CR	143,025	143,264	535	-157	692
COL	340,499	622,925	-39,314	-39,837	523
VEN	108,987	150,924	-6,306	-7,118	811
USA	-	-	303,579	47,611	255,968
Number of stable coalitions when transfers are allowed (out of 64)					2
Size of stable coalitions *(s^*)*					2 and 6
Best coalition					**LAC+USA**
Global conservation efforts under best coalition (ha)					1,190,944
CGI^E (%)					**100**
Global payoff under best coalition (thousand US$/year)					259,294
CGI^π (%)					100

a/ Positive transfers imply that countries give away part of their payoff to others, while negative transfers imply that countries receive money from others.

Table A8. Coalition stability and CGI under strong connectivity with complete habitat loss in CR (S3) with transfers for a WTP value of US$1.1/year per US household

	Conservation efforts under BAU scenario (ha)	**Conservation efforts under best coalition** (ha)	**Payoff under best coalition before transfers** (thousand US$/year)	**Transfers** [a/] (thousand US$/year)	**Payoff under best coalition after transfers** (thousand US$/year)
HON	273,013	273,832	97	-384	482
NIC	92,102	110,151	-4,773	-5,637	864
CR	-	-	735	-153	888
COL	340,499	622,925	-39,312	-39,849	537
VEN	108,987	150,924	-6,305	-7,134	830
USA	-	-	311,113	53,158	257,955
Number of stable coalitions when transfers are allowed (out of 64)					1
Size of stable coalitions *(s^*)*					6
Best coalition					**LAC+USA**
Global conservation efforts under best coalition (ha)					1,157,832
CGI^E (%)					**100**
Global payoff under best coalition (thousand US$/year)					261,556
CGI^π (%)					100

a/ Positive transfers imply that countries give away part of their payoff to others, while negative transfers imply that countries receive money from others.

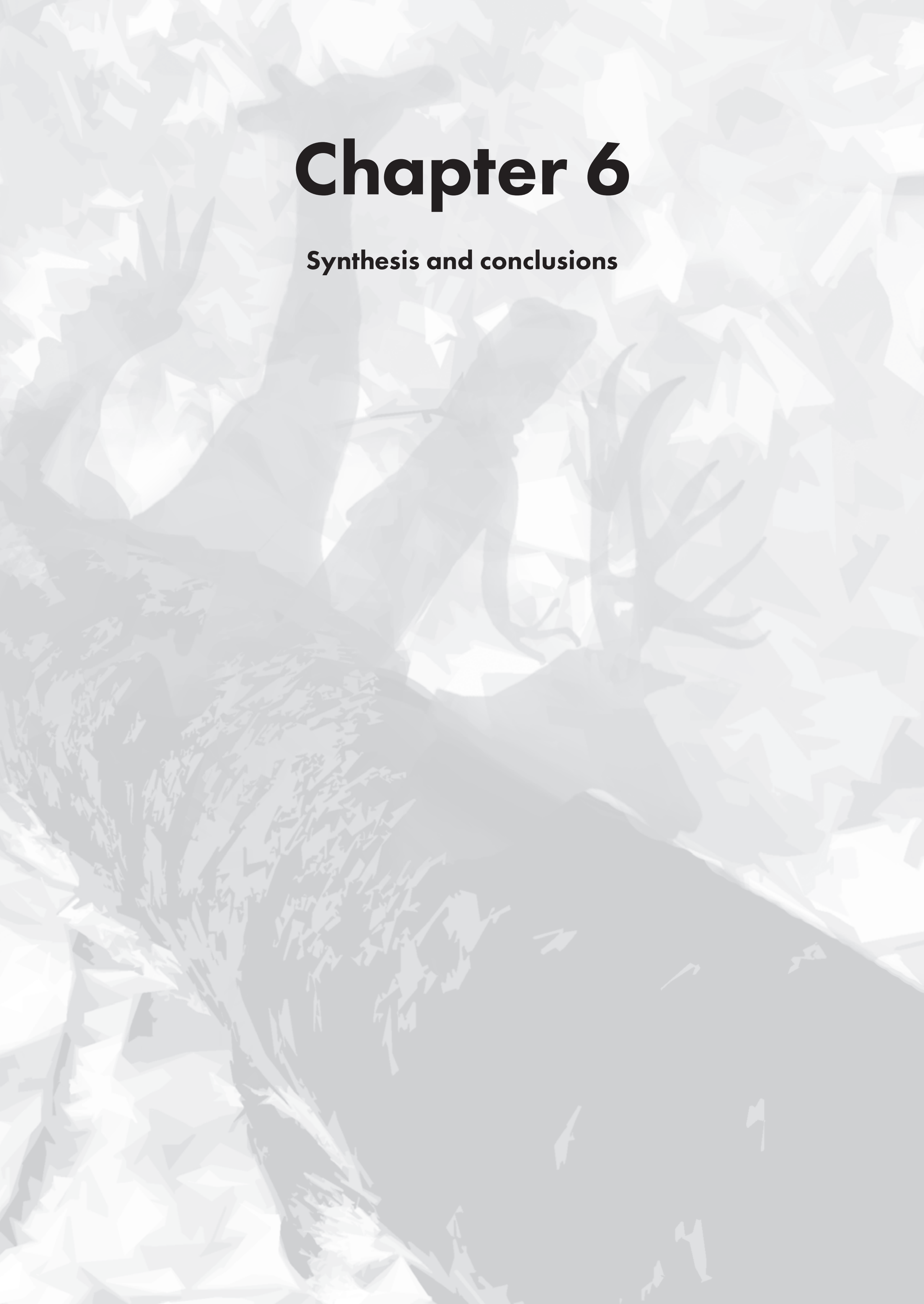
Table A9. Coalition stability and CGI under strong connectivity with complete habitat loss in VEN (S3) with transfers for a WTP value of US$1.1/year per US household

	Conservation efforts under BAU scenario (ha)	**Conservation efforts under best coalition** (ha)	**Payoff under best coalition before transfers** (thousand US$/year)	**Transfers** [a/] (thousand US$/year)	**Payoff under best coalition after transfers** (thousand US$/year)
HON	273,013	273,832	254	-391	645
NIC	92,102	110,151	-4,610	-5,634	1,024
CR	143,025	143,264	720	-178	898
COL	340,499	622,925	-39,421	-39,810	389
VEN	-	-	669	-108	777
USA	-	-	322,947	46,121	276,826
Number of stable coalitions when transfers are allowed (out of 64)					2
Size of stable coalitions *(s^*)*					2 and 6
Best coalition					**LAC+USA**
Global conservation efforts under best coalition (ha)					1,150,171
CGI^E (%)					**100**
Global payoff under best coalition (thousand US$/year)					280,559
CGI^π (%)					100

a/ Positive transfers imply that countries give away part of their payoff to others, while negative transfers imply that countries receive money from others.

Chapter 6

Synthesis and conclusions



This thesis examines the functioning and effectiveness of different economic mechanisms for biodiversity conservation at diverse scales. In Chapters 2-5, I have analysed and discussed various aspects of these mechanisms by using market theory, contract theory and game theory. In this final chapter I first answer the research questions posed in Chapter 1. Then I put the thesis in a wider perspective: Section 6.2 presents overall modelling conclusions and Section 6.3 presents policy conclusions. Section 6.4 provides the limitations of the study and Section 6.5 recommendations for further research.

6.1 Answers to the research questions and overview of findings

Q1. What are the economic conditions under which market-based mechanisms for biodiversity conservation function at the local level and what is their upscaling potential?

In Chapter 2 I present an overview of the economic conditions under which markets for biodiversity are expected to function. These conditions were identified based on both market and contract theory. The economic conditions found to be critical in the analysis of the efficiency of biodiversity markets are: i) clear and enforceable property rights, ii) a sufficient number of buyers and sellers, iii) information completeness, iv) minimisation of transaction costs and v) free entry and exit to the markets.

I performed an efficiency analysis in the light of the abovementioned conditions on a selection of five representative market-based schemes for biodiversity conservation: BioBanking, BushBroker, Conservation Banking, Malua BioBank and Wetland and Stream Mitigation Banking. The analysis shows marked differences between the examined schemes. Older market schemes such as Conservation Banking and Wetland and Stream Mitigation Banking are more consolidated and have a higher market volume as compared to the Australian BioBanking and BushBroker schemes. High entry costs remain an obstacle for the Australian schemes.

A general result from the study is that ensuring long term conservation is a common limitation for all market-based schemes. Uncertainties regarding the availability of funds to cover maintenance costs and monitoring activities undermine the credibility of biodiversity credits. Furthermore, the study shows that none of the market schemes can be easily scaled up to an international level, at least not in the way that they are currently established. I suggest the following measures to overcome the main obstacles hindering the upscaling of biodiversity markets: the standardisation of a biodiversity unit worldwide, the use of remote sensing techniques to standardise monitoring activities, and finally, the creation of a global credit registry for biodiversity credits. This registry would contribute to provide technical support regarding biodiversity credit transactions and measurement, report and verification (MRV) activities, enhancing the credibility of existing and future biodiversity markets.

Q2. What are the key features required to design an IEA for biodiversity conservation?

In Chapter 3 I present a description of three key features that are specific to the study of the formation and stability of an IEA for biodiversity conservation. I first introduce how the analysis of biodiversity management differs from another main global environmental issue that is prominent in the IEA literature: the conventional case of GHG emissions abatement. From this assessment I derive three key characteristics that are specific to biodiversity:

- The uneven distribution of biodiversity among countries. Biodiversity endowments vary among countries in terms of size and composition. Furthermore, these endowments are finite, and the maximum amount of biodiversity that a country can preserve in its territory is limited.
- The mismatch between the scales at which benefits and costs of biodiversity are perceived. Costs of biodiversity are local, but the benefits from conserving biodiversity are perceived at different scales, e.g. local, regional and global. Climate impacts from GHG reductions are perceived globally regardless of where the reductions take place. However, impacts of biodiversity conservation not only offer global benefits, but also more immediate local benefits (e.g. better air quality, health improvements, among others).
- The difficulty in aggregating biodiversity conservation efforts in an additive way. While emission abatement models consider global abatement levels as the sum of countries' levels of abatement, there is no standardised, general accepted measurement to aggregate conservation levels. Biodiversity richness can be very diverse in two protected plots of the same size. Moreover, summing the number of protected species in all countries' set of species can lead to double counting of protected species globally.

I then proceed to take into consideration these three characteristics in the design of a game-theoretical model for biodiversity conservation. As a result, the specification of my model of an IEA for biodiversity conservation includes the following features: i) a hyperbolic cost function to represent the existence of a natural upper bound of conservation per country, ii) the inclusion of local benefits of conservation in addition to the global ones, and iii) a subadditive function for global conservation made operational by using a species count as an approximate measure of biodiversity. I also relax the assumption of symmetric countries that is frequently used in IEAs models.

The study shows that the maximum size of a stable coalition in the model that features local benefits of conservation and hyperbolic costs (with symmetric countries) is of two members. This finding indicates that stable coalitions are smaller in comparison to the ones obtained in models of GHG emission abatement with quadratic cost functions (Barrett 1994, Finus and Rübbelke 2013).

Yet, when the model is extended to the three-feature case that includes subadditivity in the global conservation function, I find that larger stable coalitions can be achieved. This is possible for the case of relatively large local benefits of conservation with comparison to the global ones. Although full cooperation is achieved, results coincide with the paradox of cooperation (Barrett 1994): the gap between the aggregated payoff in the social optimum and the singletons case is small and hence gains from cooperation are small as well.

I then proceed to relax the assumption of symmetry in the study in two separate ways: first by assuming asymmetry in both benefits and costs of conservation, and then by assuming that countries have different natural upper bounds of conservation. An important outcome is that when double-sided asymmetry is allowed and a transfer scheme is implemented, the size of stable coalitions under all parameter changes systematically increases. The study shows that for this case, the inclusion of an optimal transfer rule does not only lead to larger stable coalitions and higher potential gains from cooperation, but also to a different composition of coalitions structures (in terms of country types).

Q3. What role does the inclusion of a spatial structure play in the stability of an IEA for biodiversity conservation?

In Chapter 4 I develop a model for an IEA for biodiversity conservation that considers the effects of the inclusion of an explicit spatial structure. I extend the model from the previous chapter and account not only local and global benefits of biodiversity, but also for regional biodiversity benefits. Regional biodiversity benefits are space-dependent: they are related to the distance between countries as well as to their location in a spatial structure. Since the study is concerned with cooperation between neighbouring countries, I focus on one particular setting capable of describing this type of cooperation in the simplest way: a circular spatial structure in which each country has two neighbours. In the way the model is set up, all countries are identical in costs and benefits of conservation, and also in the size of their biodiversity endowment. The only difference between countries is related to the distance between them and to their location. Furthermore, the study features the introduction of ecosystem dissimilarity (ED) as a measure of distance between countries in terms of how different their sets of species are.

The study shows that the maximum size of a stable coalition in the model with a spatial structure is of two members. These results are robust with respect to the different spatial patterns assessed within the circular structure. When countries are located equidistantly throughout the circumference of the circle, the stable coalitions with the best payoff are those composed of two neighbouring countries. For the spatial pattern of increasing distances among countries, the best global payoff is obtained when the stable coalition is composed of two countries with the smallest possible ED between them.

One main finding of the study is the evidence for a 'remoteness effect' in the increasing distance spatial pattern: one of the signatories of a (two-member) stable coalition perceives relatively lower regional benefits of conservation when it is relatively remote (in terms of ED) with respects to its other coalition member and to the singletons. This remoteness effect offsets part of the gains from cooperation that the signatory perceives. Therefore incentives to deviate from the coalition are higher, resulting in internally unstable coalitions. To sum up, higher regional benefits from conservation interfere with coalition stability, and this outcome is more prominent in coalitions composed of countries with relatively larger ED between them.

The results of a spatial pattern with clustered countries indicate that of all stable coalitions of two members, those with the highest global payoff are the ones with members that are close to each other, but also close to the other countries. Spillovers from conservation are then maximised under these circumstances.

I conclude that the highest gains from cooperation can be attained when two countries hosting the most similar set of species form a conservation agreement. Gains from such an agreement are enhanced when the set of species shared by coalition members and singletons is larger. This outcome sustains that both distance and remoteness of countries with respect to one another impact conservation measures and consequently global gains from cooperation.

Q4. How can an IEA with a spatial structure be applied to habitat conservation of a migratory bird species?

In Chapter 5 I apply a variation of the IEA model for biodiversity conservation developed in Chapter 4 to a case study on habitat conservation for a migratory bird species. In particular, I examine the viability of an environmental agreement for conservation of wintering habitats of one of the most sharply declining bird species in North, Central and part of South America: the Golden-winged Warbler (*Vermivora chrysoptera*). I study the incentives of countries to join an agreement for the protection of wintering habitats by calibrating the game theoretical model with empirical data collected by experts of the ABC for the upcoming Golden-winged Warbler Wintering Grounds Conservation Plan. Moreover, the model includes a spatial structure for the location of wintering habitats that is used to establish the regional benefits for the countries.

In the model I consider five Latin American countries that host the Golden-winged Warbler during its wintering season. Furthermore, I include an additional country with no wintering habitat for the bird, but with positive spillovers from regional conservation in the wintering range: the United States. To calibrate regional benefits perceived by the United States, we make use of a range of WTP values per US household per year to improve the chance of survival of a bird species population from 50% to 99%. I undertake the coalition stability analysis for three different spatial scenarios: weak migratory connectivity, strong migratory connectivity, and strong migratory connectivity with complete habitat loss in one of the wintering countries.

I find that in the absence of a transfer scheme and under all possible WTP values, no stable coalitions are formed under any of the three scenarios. The inclusion of transfers, however, allows for the formation stable coalitions, including the grand coalition. Under both weak and strong connectivity scenarios, the coalition with the best global payoff for the different WTP values is always composed of the six countries in the game. The only exception is when we assume WTP of zero: in this case the best stable coalition is composed of two countries, namely Honduras and Costa Rica.

After conducting a sensitivity analysis, I present the detailed results for the WTP value of US$1.1/year per household as an approximation of the benefits that US households obtain from the Golden-winged Warbler conservation. Results of the analysis considering this value show that in the full cooperative case under the weak connectivity scenario, the United States transfers around US$55 million/year to the Latin American countries with wintering habitat to stabilise the coalition, resulting in a total of 1.305 million protected ha. For the strong connectivity scenario, both efforts of conservation and transfers for the full cooperative case are slightly lower (1.301 million ha and transfers to Latin American countries of US$53 million/year). For both scenarios, Honduras and Costa Rica undertake low additional conservation efforts when compared to their BAU scenario due to their relatively high costs of conservation.

Finally, the inclusion of a country with complete wintering habitat loss in the full cooperative case induces all other countries (except for the United States) to increase their conservation efforts to jointly maximise the global payoff of the coalition. Already without any transfer from the United States, countries facing complete habitat loss have no incentive to leave the coalition under full cooperation.

I conclude from this study that a positive WTP of US households to improve the chance of survival of the population of Golden-winged Warblers, together with the implementation of a transfer scheme, can lead to a conservation agreement between Latin American countries and the United States to effectively increase conservation efforts in wintering habitats.

6.2 Conclusions on scientific approaches and modelling

Biodiversity loss represents a major threat for the livelihoods of current and future generations. Management of this global environmental resource is a complex task that has to be coordinated at different levels of implementation in order to be successful. This thesis contributes to the challenge of policy-makers to undertake efficient conservation strategies by presenting an analysis of the functioning and effectiveness of two economic instruments for biodiversity conservation: market-based mechanisms and IEAs.

With regard to the analysis of the functioning of biodiversity markets presented in Chapter 2, I conclude that defining a set of critical conditions for the efficiency of these markets is useful to obtain an indication of their overall performance. This type of review provides insight on the enforceability and compliance of the mechanisms with respect to the rights established in the agreements between landholders and the respective regulatory body, and therefore, on the credibility associated to their biodiversity credits. This is a useful tool for both buyers and sellers of the credits to make an informed decision. Undertaking this kind of comparative analysis aids to pinpoint both weaknesses of the mechanisms that need to be addressed, as well as aspects that need to be improved to achieve their full consolidation. However, this type of assessment is limited in posing a solution for some serious challenges obstacles of biodiversity markets such as the commitment of conservation in perpetuity.

On the basis of the game-theoretical model for biodiversity conservation in Chapter 3, I conclude that the inclusion of key characteristics of biodiversity in the specification of an IEA model for conservation allows for a higher degree of cooperation when compared to the conventional models of climate change literature. I find that accounting for subadditivity in the global conservation function allows for larger stable coalitions even under the assumption of symmetric countries. Yet, I encounter a common yet dismal result of coalition theory: larger stable coalitions do not achieve much more in terms of conservation when compared to the non-cooperative equilibrium. When a transfer scheme is included in the model with asymmetric countries in terms of their benefits and costs of conservation, the size of the stable coalition increases, but also the composition of the coalition structure changes when compared to the case without transfers.

Regarding the effects of including a spatial structure on the stability of an IEA for biodiversity conservation in Chapter 4, I conclude that under a circular spatial setting, stable agreements are always conformed by two countries that are the closest to each other. For our study this translates into stable coalitions between those two countries with the most similar sets of species. Highest payoffs in a stable biodiversity agreement are attained when member countries are the closest to each other, but also to the other countries in the spatial structure.

From the application of an spatial IEA model for habitat conservation in Chapter 5, I conclude that, when allowing for a transfer scheme, and considering a positive WTP value for the conservation of a bird population, full cooperation can be achieved under the different spatial scenarios. The inclusion of a country in the model that cannot undertake conservation efforts but that benefits from conservation efforts in other countries can lead to the social optimum outcome when a transfer scheme is implemented.

6.3 Policy conclusions

From the work done in this thesis I draw five main policy conclusions. First, I find that there are several obstacles that hinder the upscaling of biodiversity markets: the existence of entry fees, upfront costs of establishing an agreement, the lack of a standardised measure of biodiversity to define tradeable units, the difficulty to ensure conservation in the long run and the lack of enforcement of punishments and actions in the event of non-compliance. In this thesis I suggest that, in order to tackle some of these main difficulties, a global registry of biodiversity credits should be set up. This registry would be a voluntary entity in charge of supporting the MRV of biodiversity credits. To enhance its credibility, I suggest that it should be set up in close collaboration with recognised entities in the biodiversity arena – in both financial and conservation domains – such as the Convention of Biological Diversity, Conservation International, The World Wildlife Fund, and also the World Bank and the Global Environmental Fund. In particular, the Global Environmental Fund, as the main financial mechanism of the CBD, could play a significant role in the creation and management of this registry.

The Aichi Biodiversity Target 3 from the CBD advocates for the development and implementation of economic incentives for the conservation and sustainable use of biodiversity (CBD 2015a). The creation of a global registry under the supervision of reputable institutions would encourage to further develop and improve existing local economic mechanisms for the conservation of biodiversity. Furthermore, such registry would support the goals of other entities in creating sustainable financing schemes for conservation (e.g. The World Bank 2015).

My second main policy conclusion is the critical need for decision makers to explicitly consider asymmetries between countries (in terms of their biodiversity endowment and income) in the design, establishment and enforcement of IEAs for biodiversity conservation. In Chapter 3, I investigate coalition stability in a model for biodiversity conservation and found that, under the presence of asymmetric countries with respect to their benefits and costs of conservation, larger agreements can be attained when a transfer scheme is implemented. The inclusion of a scheme to allow the flow of transfers fosters the possibility of more effective coalitions in terms of global conservation.

This policy conclusion supports the previous recommendation to establish an international registry where biodiversity credits can be traded. Such registry would not only be a good mechanism to increase global conservation, but also to pinpoint where conservation is more effective and what characteristics do potential members of a conservation agreement hold. This registry would also support one of the UN Sustainable Development Goals *'to mobilise and significantly increase financial resources from all sources to conserve and sustainably use biodiversity and ecosystems',* and *'to finance sustainable forest management and provide adequate incentives to developing countries to advance such management, including for conservation and reforestation'* (UN 2015a, p.21).

Thirdly, policy makers must not disregard the inclusion of regional biodiversity conservation in the design and implementation of an IEA. I find in Chapter 4 that global payoffs of conservation are the highest for coalitions in which member countries are as close to each other as possible, but also when they are close to the singletons too. Also, the results of the coalition stability analysis lead to a stable agreement with a maximum of two members. This outcome is robust with respect to the inclusion of a transfer scheme. This result suggests that the alternative of multiple regional agreements, as opposed to a single international one, could potentially lead to more effective conservation outcomes (Asheim et al. 2006). To acknowledge the importance of dealing with biodiversity conservation also from a regional perspective coincides with the scope of the Strategic Plan 2011-2020 of the CBD (CBD 2015a). Although the main activities of this plan are implemented at a national or subnational level, the CBD recognises the relevance of also considering supporting actions at the regional and global levels. These actions are derived not only from establishing regional targets for conservation, but also from considering the participation of '*regional bodies to promote regional biodiversity strategies and the integration of biodiversity into broader initiatives'* (CBD 2010a, p.12).

With regards to my fourth main policy conclusion, I found in Chapter 5 that accounting for regional benefits of habitat conservation of countries that host the same migratory species – regardless of whether they can undertake conservation efforts within their territory or not – benefits the overall outcome of the stable coalition with the best payoff, in terms of number of members in the agreement, and global welfare of conservation. Once again, this is possible under the inclusion of a transfer mechanism in which member countries can incentivise others to stay in the coalition by sharing part of their gains of conservation.

Finally, my last policy conclusion is that the inclusion of transfer schemes as an instrument to incentivise biodiversity conservation can also work as an effective tool to assist in the reduction of inequality. A vast set of literature supports the notion that societies with greater inequality between rich and poor lead to negative results in terms of economic outcomes, social mobility and education, health and trust (some examples are: Bowles 1972, Pickett and Wilkinson 2010, Stiglitz 2012, Piketty 2014). Transfer schemes, as presented in this thesis, can be consider as effective mechanisms for the redistribution of resources (Singer 1975).

Species diversity is crucial for *'sustainable production, poverty eradication, sustainable economic development, hunger eradication, health and other global objectives*' (FAO 2015). Hence, there is an imminent need to use biodiversity goods and services in a sustainable way. In recognising key characteristics in the design for multinational conservation agreements, and in providing insight on the incentives and functioning of economic instruments, this thesis assisted to the global targets related to the development and understanding of economic measures to deal with biodiversity management.

6.4 Limitations of the analysis

In the final selection of biodiversity markets revised in Chapter 2, I included only one biodiversity market of the voluntary type because most of the existing schemes with sufficient available information to conduct the analysis were regulatory markets. Furthermore, in trying to assess different type of credits and operating times, the resulting selected sample of five biodiversity markets concerns only three countries.

Regarding the game-theoretical modelling part of this thesis in Chapters 3-5, I only consider games of the cartel type where only one coalition is formed. This is a limitation of the thesis, in particular for Chapters 4, as one finding suggest that there might be scope for the establishment of effective partial coalitions or bilateral agreements.

When conducting the game-theoretic model of an IEA with an embedded spatial structure examined in Chapter 4, I assume identical countries in their benefits and costs of conservation and in the size of their biodiversity endowment. This poses a limitation on the stability analysis since the assumption of symmetric countries is a strong simplification of reality.

Chapter 5 deals with the construction of a stylised example of a conservation agreement for the wintering habitat of the Golden-winged Warbler. The approximation of the parameter values considered in the model was conducted based on experts opinions as well as on the available information compiled by the ABC. Moreover, we only consider occurrence probability for the male bird population, as information regarding female occupancy is not available. Conclusions should therefore be interpreted within the context of the available information on benefits and costs of conservation of this migratory bird species and the assumptions of its migratory connectivity.

Furthermore, the analysis conducted in this chapter disregards the prioritisation of focal areas for conservation actions within countries elaborated by the Golden Winged Warbler Alliance (Golden-winged Warbler Alliance 2015). This prioritisation was based on the identification of wide goals, a review of main threats, and the inclusion of additional information per focal area. Our model is restricted to the analysis of optimal conservation efforts at a country level and not at a focal area level.

6.5 Recommendations for further research

My recommendations for further research concern two areas, namely i) the improvement of available information and ii) the improvement of models.

6.5.1 Improvement of available information

First, the review and analysis of biodiversity markets could benefit from an extension of the selection of market-based mechanisms. Market schemes of the voluntary type were underrepresented in the analysis presented in this thesis. Also, the assessment would benefit from a more representative sample in terms of location, as our selection was restricted to three countries. Finally, a follow-up examination of the previously selected schemes would shed some light on the evolution of their market volume.

Second, the focus of the case study of the viability of a regional agreement for the conservation of wintering habitats of the Golden-winged Warbler was done at a national scale. This scale of analysis was chosen to present a simple yet insightful analysis on the effects of a spatial structure on coalition formation. Yet, the analysis could be narrowed down to a local level by considering the effectiveness of conservation efforts per focal area instead of per country. Furthermore, research should be stimulated to reduce the knowledge gap on migratory connectivity and valuation of migratory bird species as a means to improve the calibration of our model. In this way, it could become a more effective tool in the provision of accurate information for decision makers regarding the design of agreements for habitat conservation. Finally, the study could be further extended to conduct a metapopulation assessment by relaxing the assumption of independent occurrence probabilities among countries, but this requires more in-depth data collection and analysis on the metapopulation structure.

6.5.2 Improvement of models

The stability analysis conducted in Chapter 4 could be further improved by assuming heterogeneous countries with respect to their benefits and costs of conservation as in Chapter 3. This extension would be particularly valuable as it would consider the inclusion of local, regional, and global benefits of conservation under a setting of asymmetric countries, which would result in a more comprehensive approach. Moreover, it may be interesting for future research to study the alternative of multiple coalitions to assess their stability and effectiveness in dealing with biodiversity conservation.

Relaxing the assumption of independent occurrence probabilities in Chapter 5 would allow to conduct a more comprehensive stability analysis for metapopulations of migratory species. Also, the model would benefit considerably from the inclusion of additional information regarding local benefits of conservation of the bird species, e.g. bird-friendly ecotourism, agri-environmental schemes, among others.

References

Summary

Acknowledgements

About the author

Training and Supervision Plan

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Summary

Biodiversity decline poses significant threats to current and future generations. Although species extinction has been a natural process since the formation of Earth, recent rates of extinction are estimated to be from 100 to 1000 times larger when compared to fossil records. Almost all of the Earth's ecosystems have been dramatically transformed and some of them are being pushed towards critical thresholds that could risk overall livelihoods and wellbeing of human population. Implications of severe biodiversity loss include irreversible alterations of ecosystem services, vulnerability to natural disasters, human health risks, threats to food and energy security, depletion of natural resources and damage to social relations.

There is an urgent need to study and develop efficient conservation instruments that decision makers can implement to halt the ongoing rate of biodiversity loss. However, this is a complex task due to i) the multidimensional nature of biodiversity conservation in terms of the different levels of biological organisation, and also to ii) the diverse geographical scales of concern at stake (from local to global). The objective of this thesis is to examine the functioning and effectiveness of different economic instruments for biodiversity conservation at diverse scales. In order to achieve this objective, different methodological approaches such as market theory, contract theory, and game theory are implemented.

In Chapter 2, I develop an assessment of economic characteristics for biodiversity markets to work efficiently. I first introduce a set of general conditions to guarantee market efficiency. These conditions are derived from market and contract theory. In the light of these conditions, I analyse the efficiency of five selected market schemes for biodiversity conservation that have been implemented in different countries. An assessment of the upscaling potential of the existing markets reveals that obstacles such as the lack of a standardised unit of measurement for biodiversity and the difficulty to ensure long-term conservation make it difficult to scale up any of the selected mechanisms as they are currently performing. I argue that the creation of a global credit registry for biodiversity would facilitate measurement, reporting and verification (MRV) of biodiversity credits to support market-based mechanisms.

In Chapter 3, I present a game-theoretic model for an international environmental agreement (IEA) for biodiversity conservation. I first introduce three key characteristics that differentiate the case of biodiversity conservation from the conventional emission abatement model: the existence of a natural upper bound of conservation per country, the importance of local benefits, and the subadditivity of the global conservation function. Then, I consider asymmetries in benefits and costs of biodiversity conservation, and separately, in the natural upper bound of conservation per country. Results show that there is scope to achieve a higher degree of cooperation in a potential IEA for

biodiversity conservation when subadditivity in the global conservation function is considered. Furthermore, the inclusion of an optimal transfer rule allows not only for larger stable coalitions and higher potential gains of cooperation and conservation, but also for a different composition of coalition structures (in term of country types).

In Chapter 4, I analyse the inclusion of an explicit spatial structure in the modelling of an IEA for conservation. I assess the role of distance and location between countries on coalition formation and overall coalition stability. First, to explain cooperation among neighbouring countries I make use of a specific setting: a circular spatial structure. Furthermore, I employ a notion of distance between countries in terms of their ecosystem dissimilarity: two countries are closer the more species they have in common. I argue that, for the purpose of exploring the stability of conservation agreements, geographical distance may be less important than the dissimilarity of the sets of species that two countries host. Results show that the maximum size of a stable coalition in the model with a spatial structure is of two members. These results are robust with respect to the different spatial patterns assessed within the circular structure. I conclude that the stable coalition with the best global payoff is obtained when stable coalitions are composed of two countries with the smallest possible distance between them. Also, the study shows evidence of a 'remoteness effect'. Highest payoffs in a stable biodiversity agreement are attained when member countries are the closest to each other, but also to the rest of the countries in the spatial structure.

In Chapter 5, the model for an IEA for conservation with an embedded spatial structure is applied to a case study on regional conservation of the non-breeding habitat of the Golden-winged warbler (*Vermivora chrysoptera*). I study the incentives of countries to join an agreement for the protection of wintering habitats by calibrating the game theoretical model with empirical data. Also, I include a spatial setting that best describes specific aspects of the migratory behaviour of the species. Results show that when there is a positive willingness to pay of US households to improve the chance of survival of the population of the Golden-winged Warbler, and when allowing for the implementation of a transfer scheme, there is scope for a stable conservation agreement between the United States and the Latin American countries with wintering habitat of the bird species (i.e. full cooperation). For all scenarios of our study, the United States transfers part of its payoff to the Latin American countries to incentivise conservation and stabilise the coalition.

This thesis has shown the importance of taking into account asymmetries between countries – both in their biodiversity endowments as well as in benefits and costs of conservation activities – in the design and application on economic instruments for biodiversity conservation. Furthermore, the implementation of transfer schemes as instruments to incentivise conservation have the potential to contribute to effective biodiversity management.

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About the author

Irene Alvarado Quesada was born in San José, Costa Rica, on September 30, 1984. She completed her BSc studies in Economics at the Universidad de Costa Rica in 2007. In 2008 she received a full fellowship from the Netherlands Fellowship Programme (NFP-NUFFIC) to pursue her MSc studies at Wageningen University through the International Development Studies programme. During her MSc studies, she completed a major and minor MSc thesis with the Environmental Economics and Natural Resources (ENR) Group at Wageningen University. Both theses were conducted in the Mekong Delta in Vietnam. Her major thesis focused on groundwater management and drought vulnerability in Tra Vinh province, and her minor thesis focused on costs and benefits of flood occurrence in Dong Thap province. Upon completion of her MSc studies in 2010, Irene began her PhD studies on the function and effectiveness of economic instruments for biodiversity conservation at the ENR Group of Wageningen University, under the supervision of prof. Dr Ekko van Ierland and Dr Hans-Peter Weikard. In 2014, she was awarded a research grant from the Latin American and Caribbean Environmental Economics Programme (LACEEP) to conduct a research project on international cooperation for the conservation of the wintering habitat of the Golden-winged Warbler in Latin America. The outcome of this research resulted in one of the chapters of this PhD thesis.

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- *International cooperation for biodiversity conservation when spatial structures matter.* Belpasso International Summer School on Environmental and Resource Economics: 'Spatial Context and Valuing Natural Capital for Conservation Planning', 31 August-6 September 2014, Belpasso, Italy
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