THE TREASURE OF THE COMMONS: VALUING AND MANAGING NATURAL CAPITAL IN COSTA RICA

By

Marcello Hernández-Blanco

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A thesis submitted for the degree of Doctor of Philosophy of The Australian National University

Statement of originality

This is to certify that the content of this thesis is my own work. This thesis has not been submitted for any other academic degree. I certify that the intellectual content of this thesis is the product of my own work and that all the assistance received in preparing this thesis and sources have been acknowledged.

When a paper has been written with one or more co-authors, a statement of contribution is provided for each paper.

Marcello Hernández-Blanco

Statement of Contribution

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Title: Natural Capital and Ecosystem Services

Authors: Marcello Hernández-Blanco and Robert Costanza

Publication outlet: Routledge Handbook of Agricultural Economics

Current status of paper: Published

Contribution to paper: Lead author. I wrote 90% of the paper.

Senior author endorsement: I'm the senior author of this paper.

Marcello Hernández-Blanco Candidate – Print Name	Signature	21/12/18 Date

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Robert Costanza Primary Supervisor – Print Name	Signature	21/12/18 Date
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Title: Future scenarios for the value of Ecosystem Services in Latin America and the Caribbean to 2050

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Title: The economic value of ecosystem services of 7 Ramsar Sites in Costa Rica

Authors: Marcello Hernández-Blanco, Olman Segura-Bonilla, Mary Luz Moreno-Díaz, Edgardo Muñoz-Valenciano

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Title: Economic Valuation of the ecosystem services provided by the mangroves of the Gulf of Nicoya using a hybrid methodology

Authors: Marcello Hernández-Blanco, Robert Costanza, Miguel Cifuentes-Jara

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To Camila, my daughter, my greatest gift.

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Being able to attend to ANU has been a dream come true. I will be eternally grateful with the University for accepting me as a student, for giving me an Australian experience that now is tattooed in me (figuratively and literally speaking). And thank you to the Ministry of Science and Technology from Costa Rica, for providing me the generous scholarship that made all of this possible. Because of them I reached one of my most desired goals.

Abstract

While Costa Rica has been a world leader in ecosystem services research and policy, the value of natural capital and ecosystem services is still often ignored or underestimated. The overarching goal of this dissertation is to address this knowledge gap and to formulate economic incentives and institutions that can adequately incorporate ecosystem services. First, I summarize the state of the art of natural capital and ecosystem services. I describe concepts, how they have evolved in the last two decades, and the most appropriate methods for ecosystem services valuation. Second, I estimate the present and future value of ecosystem services in Latin America and the Caribbean, with a focus on Costa Rica. I estimate that the current ecosystem services value (ESV) of the 33 countries that make up this region to be $US15.3 trillion/year. Modelling four future scenarios, I estimated that there is a potential for ESV to decrease to $8 trillion/year (for the "Fortress World" scenario) or to increase to $19 trillion/year (for the "Great Transition" scenario), a difference of a 47% decrease or a 25% increase. Third, I explore the economic value of wetlands at a national level, estimating the value of ecosystem services of 7 Ramsar Sites. Our results show that the total economic value of ecosystem services from these Ramsar Sites is $3.2 billion/year for 2015, which represents 6% of the country's GDP. Fourth, I focus on the value of ecosystem services of mangroves in the Gulf of Nicoya. A hybrid approach was used, including traditional benefit transfer and expert modified benefit transfer for 11 ecosystem services, and primary studies for 3 ecosystem services (fisheries, climate regulation and coastal protection). Using traditional benefit transfer, the economic value of 11 ecosystem services was estimated at $812 million per year (median=$88 million/year), and the total mean value of the ecosystem services provided by the total extent of mangroves in Costa Rica at $1.5 billion per year (median=$160 million/year). Combining the values of the expert modified benefit transfer with the estimates from the primary studies, the mean total value of the ecosystem services was estimated at $408 million per year (median $86 million/year). Considering the median total value of ecosystem services from mangroves, this represents 0.16% of the GDP in Costa Rica in 2015. Finally, I propose a research and policy agenda to establish a payment for ecosystem services scheme in Costa Rica for marine and coastal ecosystems. I recommend a six-

step approach to create this new PES scheme, called the Blue Fund: 1) Ecosystem assessment, 2) ecosystem services selection and valuation, 3) threats identification and prioritization, 4) creation of funding sources and investment, 5) implementation, and 6) evaluation and adaptation. I illustrate each step with a focus on mangroves and coral reefs.

Publications from this thesis

The following papers of this thesis have been published or are in the process of being published:

Hernández-Blanco, M; Costanza, R (2019). Natural capital and ecosystem services. In *The Routledge handbook of agricultural economics* (pp. 254-268). New York, NY: Routledge.

Hernández-Blanco, M; Costanza, R; Cifuentes-Jara, M (2018). Economic valuation of the ecosystem services provided by the mangroves of the Gulf of Nicoya using a hybrid methodology. Ecosystem Services. Under review.

Hernández-Blanco, M; Segura-Bonilla, O; Moreno-Díaz, M.L.; Muñoz-Valenciano, E (2018). The economic value of ecosystem services of 7 Ramsar Sites in Costa Rica. Wetlands. Under review.

The following papers of this thesis are ready to be submitted to a journal:

Hernández-Blanco, M; Costanza, R; Anderson, S; Kubiszewski, I; Sutton, P. (2019) Future scenarios for the value of Ecosystem Services in Latin America and the Caribbean to 2050

Hernández-Blanco, M (2019) A research and policy agenda for establishing a payment for ecosystem services scheme for the conservation and restoration of marine and coastal ecosystems in Costa Rica. Marine Policy.

The following papers of this thesis were published in the grey literature:

Hernández-Blanco, M; Costanza, R; Cifuentes-Jara, M (2018). Valoración económica de los servicios ecosistémicos provistos por los manglares del Golfo de Nicoya. San José, Costa Rica: Conservación Internacional.

Hernández-Blanco, M; Segura-Bonilla, O; Moreno-Díaz, M.L.; Muñoz-Valenciano, E (2017). Valoración de los servicios ecosistémicos que ofrecen siete de los humedales protegidos de importancia internacional en Costa Rica: Palo Verde, Caribe Noreste, Caño Negro, Gandoca-Manzanillo, Maquenque, Térraba-Sierpe y las Baulas. San José, Costa Rica: PNUD – SINAC.

Note:
Because this thesis is composed of separate (but related) papers, figures and tables are numbered in this way, restarting their numbering with each paper/chapter.

Table of Contents

Introduction: The treasure of the commons – valuing and managing natural capital in Costa Rica

Society's focus on economic growth as the main goal of prosperity has had a socio-economic and environmental impact so significant that some scholars are proposing that humanity has entered in a new geological epoch called the Anthropocene, leaving behind the Holocene (the last 11,700 years of human history) (Steffen et al, 2011). The beginnings of the Anthropocene can be traced back in two different moments of our history: the Industrial Revolution and the middle of the 20th century after World War II. This second period of the beginning of the Anthropocene is called the Great Acceleration, and it is defined by the exponential growth of development indicators such as population, GDP, water consumption, fertilizer consumption and communication, among others (Steffen et al., 2011).

The challenges imposed by the Anthropocene to achieve sustainable development require a new vision of the economy, one in which economy is viewed as a system that is part of a broader one, the Earth System (a single complex system with reasonably well-defined states and transitions between them), instead of viewing nature as just another source of resources. This approach will require recognizing the value of natural capital, which has been commonly ignored, because for many decades, our ecological footprint was low and natural resources were abundant. Therefore, it was difficult to think that we could significantly impact and critically the environment as we actually have done. The field of Ecological Economics addresses this research and policy gap. It is a transdisciplinary field of study that studies the relationships between ecological and economic systems (Costanza & Daly, 1992a), with the goal to represent a new approach to both interconnected systems. It recognizes the need to make economics more aware of ecological dependencies and impacts, and to make ecology more sensitive to economic forces (Costanza, 1989).

Ecological economics is not intended to be an alternative to existing disciplines, it is instead a new lens to view and evaluate sustainability challenges. It views conventional economics as one of many inputs to a broader transdisciplinary framework (Costanza et al., 1991) with the following interdependent goals: 1) present a shared vision of how the world works and

the sustainable development that we aspire to achieve, 2) provide a relevant methodology to assess the new questions and challenges of this shared vision, and 3) design the institutions and instruments needed to implement this vision (Costanza et al., 1991). Costanza and colleagues argue that neoclassical economics is focused mainly on allocation. Distribution is secondarily and scale is left out completely. In contrast, ecological economics address the problems in the opposite order. First, it determines the ecological limits of sustainable scale and creates the polices to assure that the throughput of the economy stays within these limits. Then, it establishes a fair and just distribution of resources through property rights and transfers mechanisms. Finally, mechanisms (including the market) allocate the resources efficiently (Costanza et al., 1991). The reason for addressing scale in the first place is of major importance, from a conceptual point of view, re-defining the foundations of society's current development that supposes that economy can grow indefinitely.

In order to recognize the relation between these two systems, ecology and economics, the concepts of natural capital and ecosystem services emerged more than two decades ago (Gómez-Baggethun & De Groot, 2010; Westman, 1977; Ehrlich & Mooney, 1983). Based on the economics' definition of capital, natural capital can be defined as "a stock of natural resources (i.e. ecosystems) that yield a flow of goods and services (i.e. ecosystem services)". Under this approach, the flow of goods and services are the "natural income", and the stock that yield this flow is "natural capital" (Costanza & Daly 1992, p. 38). Hence, for society to achieve sustainable development (recognizing the limits of our biosphere) it must be focused on the wise use of this income, instead of depleting the stocks, which is the reason of ecosystems' degradation and loss (Prugh et al., 1995).

The research presented here explores in depth the theory and application of the concepts of natural capital and ecosystem services at different spatial scales, with special attention to natural resources considered commons, in order to visualize the benefits they provide to society and hence, the need for its sustainable use, as well as the design of institutions and economic mechanisms to adequately manage them. The end goal is to provide knowledge, from a scientific and policy perspective, to help transform the tragedy of the commons (Hardin, 1968) into *the*

treasure of the commons, a paradigm shift where different sectors of society share the goal of nature conservation and restoration as one of the fundamental requirements to secure its present and future well-being. Elinor Ostrom argued that wealth that is free for all must be valued and managed by the community in order to prevent the tragedy of open access to the commons (Ostrom, 2015). Here, I provide examples of how to turn the commons into wealth that is valued by everyone.

I illustrate the treasure of the commons in Costa Rica, a country widely known for being a biodiversity hotspot. Costa Rica is an economy that has been able to increase the well-being of its citizens, at the same time that it protects natural resources. It has managed a co-evolution between ecology and economics, both intrinsically interdependent, especially in a country that produces the majority of its GDP from ecotourism. Nevertheless, as in most parts of the world, this treasure has been traded for some immediate economic gains, causing the loss and degradation of terrestrial and marine ecosystems. This mainly occurs because these are often undervalued or not valued at all. With this research, I hope to make more visible the value of Costa Rica's most valuable treasure.

Research questions and answers

The overarching research question of this thesis is *how to value and manage natural capital in Costa Rica in order to conserve and enhance its ecosystem services?* This question emerges from the current knowledge gap on ecosystem services valuation in the country, as well as the need to develop new mechanisms to protect and restore them, especially in coastal and marine ecosystems that have not received the same degree of attention as forests have.

To address this overarching question, this thesis is divided in five chapters. Each chapter is a separate paper with a specific research question, but all the papers are interconnected, which will be explained later. Papers are organized from theory to practice, and from a regional scale (i.e. Latin America and the Caribbean) to a local scale (i.e. Gulf of Nicoya in Costa Rica) (Figure 1).

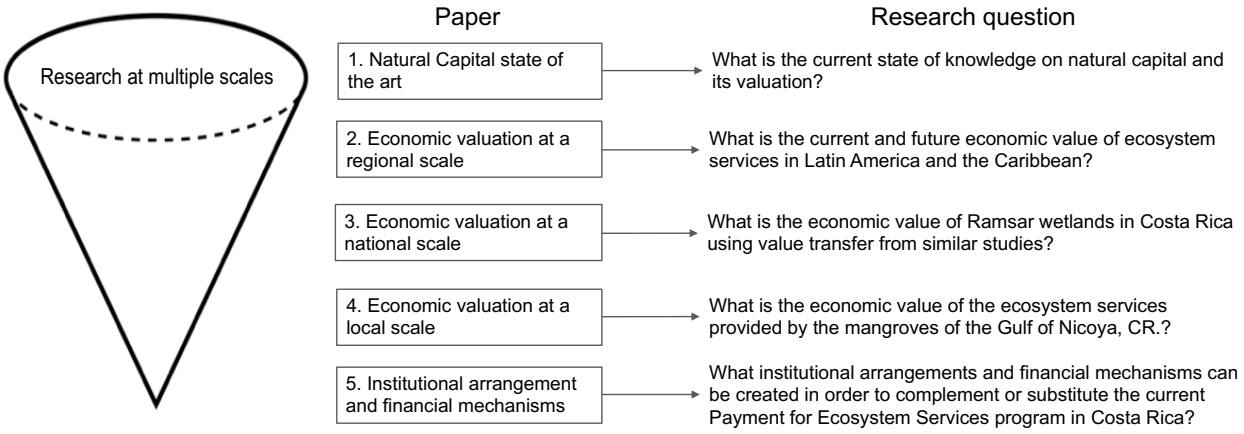

Figure 1. Topics and research questions for each paper of this thesis, organized from a global to a local scale, starting a theoretical description of the topic.

In **Chapter 1,** I conduct a literature review on natural capital and ecosystem services to better understand the state of the art of these two concepts. I begin providing a general description of the history of these concepts, from the introduction of "natural capital" more than three decades ago to the seminal works from Robert Costanza and colleagues and Gretchen Daily in 1997, which sparked an explosion of research and policy interest in ecosystem services. I then explain what scarce resources and their classification are as stock-flow resources and fund-service resources, as well as their classification under the principles of excludability and rivalness.

Building on the economic concept of capital, I provide a definition and classification of natural capital, as well as an explanation of how it interacts with other types of capital (i.e. social, human, and built) in order to provide well-being to society. The concept of ecosystem services is also defined, accompanied by an explanation of the concept of value and valuation to better understand the economic valuation of these services, in addition to a list and description of the most common valuation methods.

Having explained the conceptual basis of natural capital, **Chapter 2**, is the first of the valuation studies that I conducted, starting from a regional scale. This paper builds on the global estimates of the current and future value of ecosystem services by Kubiszewski et al. (2017). I provide an in-depth analysis of the plausible changes in value that ecosystem services can have

under four development scenarios in Latin America and the Caribbean. I first estimate the value of ecosystem services for the 33 countries of the region as $US15.3 trillion/year. Then, using the interactive web tool from the GTI website, "Futures in Motion", Kubiszewski et al (2017) estimated land use change, population, economic activity (GDP) and inequality, among other variables under four scenarios, along with the change in value of ecosystem services from land-cover change and "unit value" change of each ecosystem, due to degradation and restoration. I estimate these changes in more detail for the countries of Latin America and the Caribbean.

The results indicate that there is a potential for ecosystem service value to decrease to $8 trillion/year (for the "Fortress World" scenario) or to increase to $19 trillion/year (for the "Great Transition" scenario), a difference of a 47% decrease or a 25% increase. At a sub-regional scale, the Caribbean would experience a higher degree of change in ecosystem value in the future under three of the four scenarios, ranging from a 35% decrease to a 30% increase. Looking at Costa Rica, which is the focus of this thesis, it is the most affected country in Mesoamerica by the two most negative development scenarios, with a decrease in its ecosystem service value ranging from 28% to 48%

After estimating the value of ecosystem services for the present and future in Latin America and the Caribbean, I explore in **Chapter 3** the economic value of ecosystem services at a national scale by estimating the value of ecosystem services of 7 Ramsar Sites. For this, instead of using fixed global values for each ecosystem, I estimated a value per hectare of 16 land use categories, 9 ecosystems (swamps, ocean, rivers, beaches, scrubland, yolillales, lagoons, forests and mangroves) and 7 human activities (rice paddies, banana plantations, sugar cane, grasslands for cattle production, aquaculture, pineapple fields and oil palm plantations).

After estimating the value of each ecosystem, I found that rivers are the ecosystem that have the highest economic value ($86,090/ha/year), mainly due to their very high value from the prevention of extreme events ($72,655/ha/year). This is followed by beaches ($48,319/ha/year) in which recreation is the ecosystem service with the highest value ($20,889 ha/year) and rainforests ($24,972 ha/year) with the highest value for bioprospecting ($12,145 ha/year). Using the per hectare per year values of each land cover and land use, I applied the benefit transfer method, where I multiplied the total extension of each ecosystem of each Ramsar Site by its per

hectare value. The results show that the total economic value of ecosystem services from the 7 Ramsar Sites is $3.2 billion/year for 2015, a value higher than some sectors of the GDP of Costa Rica, such as agriculture, forestry and fisheries ($2.71 billion/year) and construction ($2.69 billion/year). This makes the relevance of natural capital to society's well-being and to the national economy visible.

The application of the benefit transfer method at a national scale proved to be a useful tool when time and budget are limited, and when the quantity of ecosystems that needs to be valued is high. In order to better understand the application of this and other methods on a local scale, in **Chapter 4,** I conducted a study on the economic value of the ecosystem services provided by mangroves in the Gulf of Nicoya, in Costa Rica. I applied a hybrid "three-tier" method, starting with benefit transfer (as in Chapter 3), but then moving to more detailed and sophisticated methods; first applying an expert modified benefit transfer to calibrate the results from the first method, and then conducting primary studies for 3 ecosystem services (fisheries, climate regulation, and coastal protection).

Using traditional benefit transfer, the economic value of 11 ecosystem services was estimated at $812 million per year (median=$88 million/year), and the total mean value of the ecosystem services provided by the total extent of mangroves in Costa Rica at $1.5 billion per year (median=$160 million/year). Furthermore, with these results, I estimated that Costa Rica has lost on average $1.1 billion per year (median=$120 million per year) from the loss of mangrove cover between 1980 and 2013. Applying the expert modified benefit transfer, I estimated that the mean total value of the mangrove forests of the Gulf of Nicoya is $470 million per year, and a median value of $75 million per year. Finally, combining the values of the expert modified benefit transfer with the estimates from the primary studies, I calculated the mean total value of the ecosystem services from mangrove forests in the Gulf of Nicoya as $408 million per year, and a median total value of $86 million. Considering the median total value of ecosystem services from mangroves in the Gulf of Nicoya, it represents 0.16% of the GDP in Costa Rica in 2015.

After valuing natural capital at the regional, national and local level, in **Chapter 5,** I explore the other key dimension of natural capital management, the institutional arrangements and

financial mechanisms to incorporate its value in decision making. For this, I developed a research and policy agenda to establish a new payment for ecosystem services scheme for the conservation and restoration of marine and coastal ecosystems in Costa Rica, called The Blue Fund. This is a paradigm shift compared to the current PES scheme for forests, not only because this new fund would incorporate several ecosystems, but because all them are owned by the government rather than private land owners.

The agenda consists of six steps: 1) ecosystem assessment, 2) ecosystem services selection and valuation, 3) threats identification and prioritization, 4) creation of funding sources and investment, 5) implementation of conservation and restoration projects, and 6) evaluation and adaptation. I illustrate how to develop each step of the agenda providing detailed examples for mangroves and coral reefs, resulting in the first design of a PES in the country for these two ecosystems. Nevertheless, the process can be applied to any other marine and coastal ecosystem, and even to terrestrial ecosystems, which could also result in the design of a new version of the current, and outdated, PES scheme from FONAFIFO. Moreover, The Blue Fund proposed here can also be a sub-fund of a broader institution and financial mechanism that I have named the Natural Capital Fund, which would include both terrestrial and marine ecosystems, recognizing the linkages between them and the need to increase the productivity of these types of funds by unifying loose initiatives into one fund that can be better managed.

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Natural Capital and Ecosystem Services

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Abstract

The concept of natural capital emerged more than three decades ago and it is defined as "a stock of natural resources (i.e., ecosystems) that yield a flow of goods and services (i.e., ecosystem services)". We present as well a definition of ecosystem services, "the benefits that people obtain from ecosystems", which can be economically valued through a wide variety of methods that we describe, each one depending on the service that needs to be valued. Economic valuation has been criticized perhaps because of a misunderstanding on the concept of value, which we also define here, but we also present some difficulties in conducting these studies. We finish this chapter with a reflection of the approach of natural capital within sustainable development.

Key words: natural capital, ecosystem services, ecological economics, total economic valuation, capital, value

A short history of natural capital and ecosystem services

Gomez and De Groot (2010) state that the concept of natural capital was introduced for the first time in 1973 by Schumacher in his book entitled, "Small is Beautiful: A Study of Economics As If People Mattered" (Gómez-Baggethun & De Groot 2010, p. 108). The term "nature's services" appeared for the first time in the literature in a paper published in *Science* by Walter Westman titled, "How much are nature's services worth?"(Westman 1977). "Ecosystem services" as a synonymous term to "nature's services" was mentioned for the first time in Ehrlich and Ehrlich (1981), and more systematically in Ehrlich & Mooney (1983).

In 1988, Pearce made one of the earliest introductions to the concept of natural capital, stating that "sustainability requires at least a constant stock of natural capital, construed as the

set of all environmental assets" (Pearce 1988). Pearce's goal was to stimulate discussion and research around the topic of sustainability within the field of neoclassical economics. As Akerman states, the concept was then redefined by Costanza and Daly, who brought ecosystem thinking into economic analysis, implying a theoretical change in the understanding of how both ecological and economic systems worked, opening the path for the emerging field of ecological economics (Akerman 2003, p. 443). A more detailed history of ecosystem services focused on its economics roots is provided by Gómez-Baggethun, de Groot, Lomas, & Montes (2010), and L. C. Braat & de Groot (2012), who summarize the history of the concept from the perspective of ecology, economics, and ecological economics.

The year 1997 was a turning point in research and the conceptualization of natural capital and ecosystem services. First, the book *Nature's Services: Societal Dependence on Natural Ecosystems* (Daily 1997) was published, a product of a meeting in October 1995 of Pew Scholars in Conservation and the Environment in New Hampshire, which included scholars such as Jane Lubchenco, Stephen Carpenter, Paul Ehrlich, Gretchen Daily, Hal Mooney, Robert Costanza, and others. Second, during this meeting, Robert Costanza proposed the idea to synthesize all the information being assembled and develop a global assessment of the value of ecosystem services. This was done through a workshop called "The Total Value of the World's Ecosystem Services and Natural Capital," held on 17-21 June 1996 with the financial support of the U.S. NSF-funded National Center for Ecological Analysis and Synthesis (NCEAS) and with the participation of 13 scholars from a range of disciplines. The results were published in *Nature* (Costanza et al. 1997). They provided a "meta-analysis" of all existing studies on seventeen ecosystem services across sixteen biomes that were valued in the range of US$16–54 trillion per year, with an average of US$33 trillion per year, a value significantly higher than gross domestic product (GDP) at the time. These two publications sparked an explosion of research and policy interest in ecosystem services by helping to visualize the dependence that humans have on healthy ecosystems and therefore, the importance of protecting natural capital for human well-being.

Classifying resources: basic principles for natural capital definition

Before analysing the concept of capital (and specifically natural capital), we need to consider some basic definitions that are implicit in it. First, it is important to make a distinction between types of scarce resources, stock-flow and fund-service. On one hand, Georgescu-Roegen defines, in Daly and Farley (2004), a **stock-flow resource** as one that is materially transformed into what it produces, can be used at any rate desired, can be stockpiled, and is used up instead of worn out (e.g., goods such as timber, water, minerals, and fish). On the other hand, a **fund-service resource** is defined as one that cannot be materially transformed into what it produces, can only be used at a given rate, cannot be stockpiled, and is worn out instead of used up (e.g., services such carbon sequestration, erosion control, pollination, and water retention) (Daly & Farley 2004, p. 71).

Second, the classification of resources under the principles of excludability and rivalness is key because it is directly related to the concepts of stock-flow and fund-service. An **excludable resource** is one which its owner can use while simultaneously denying its use to others (the opposite is a **non-excludable resource**). A **rival resource** is when it is consumed or used by one person, it reduces the amount available for everyone else, and a **non-rival resource** is one in which the use by one person does not affect its use by another. In general terms, most stock-flow resources are rival while fund-service resources are non-rival (Daly & Farley 2004, p. 73) (Figure 1).

These definitions frame both the consumption possibilities of resources as well as their governance, which at the end determines their sustainability, key to maintaining the well-being of current and future generations.

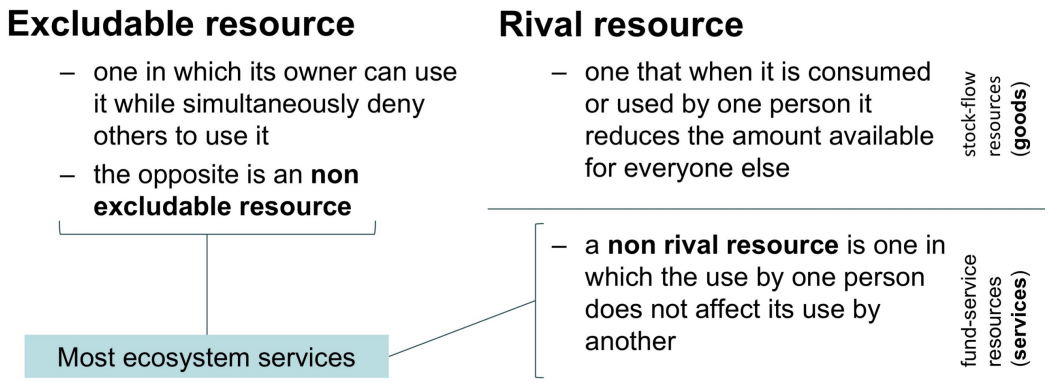

Figure 1. Types of scarce resources. Most ecosystem services are non-excludable and non-rival, which pose a challenge for their sustainable management.

Natural Capital concept

Capital can be defined as a "stock of materials or information that exists at a point in time" (Costanza et al. 1997), or moreover as "a stock of something that yields a flow of useful goods or services" (Costanza et al. 2014, p. 119).

Classical economics identifies three economic factors of production: land, labour, and human-made capital. Neo-classical economics tends to focuses primarily on labour and human made capital in its production functions, omitting land. Corresponding to these three traditional economic factors of production, three types of capital can be defined: natural, human, and manufactured or built capital (Costanza & Daly 1992, p. 38 and T. Prugh et al. 1995, p. 53). Moreover, Ekins (2003) proposes a disaggregation of the capital stock, adding a fourth type of capital, the social capital (Ekins et al. 2003, p. 166). Costanza (2014) states that these four types of capital are necessary to support the economy and its goal of providing human well-being, describing each one of them as follows:

- Natural capital: The natural environment and its biodiversity; it is the planet's stock of natural resources, the ecosystems that provide benefits to people (i.e., ecosystem services).

- Social capital: The web of interpersonal connections, social networks, cultural heritage, traditional knowledge, and trust, and the institutional arrangements, rules, norms, and values that facilitate human interactions and cooperation between people.
- Human capital: Human beings and their attributes, including physical and mental health, knowledge, and other capacities that enable people to be productive members of society.
- Built capital: Buildings, machinery, transportation infrastructure, and all other human artifacts and services (Costanza et al. 2014, pp. 129–130).

Following the definition of capital cited before, natural capital can be defined as "a stock of natural resources (i.e., ecosystems) that yield a flow of goods and services (i.e., ecosystem services)," such as the case of a mangrove forest that provides food and water filtration to communities. Costanza and Daly explain the flow of goods and services as the "natural income" and the stock that yields the flow as the "natural capital" (Costanza & Daly 1992, p. 38). Sustainability (more on this later) is therefore centered in the wise use of income; depleting the stocks is called capital consumption (T. Prugh et al. 1995, p. 51) and is the reason for ecosystems' loss and degradation.

Berkes and Folke state that natural capital and built capital are fundamentally complementary; it is not possible to create built-capital without support from natural capital (Berkes & Folke 1992). Furthermore, it is important to note that natural capital (i.e., ecosystems) cannot provide benefits to people without its interaction with the other three types of capital. Ecosystem services (defined in the next section) do not flow directly from natural capital to human well-being (Costanza et al. 2014, p. 153). Therefore, "ecosystem services refer to the relative contribution of natural capital to the production of various human benefits, in combination with the three other forms of capital" (Figure 2) (Costanza 2012, p. 103).

Perceiving natural capital in insolation from the other forms of capital produces a bias in its management. Often, management of natural capital is the responsibility of the ministries of the environment and does not include other ministries, such as industry, agriculture, or finance. In the private sector, natural capital management is commonly the responsibility of the corporate sustainability department and does not come up in boardrooms (Guerry et al. 2015, p. 7350).

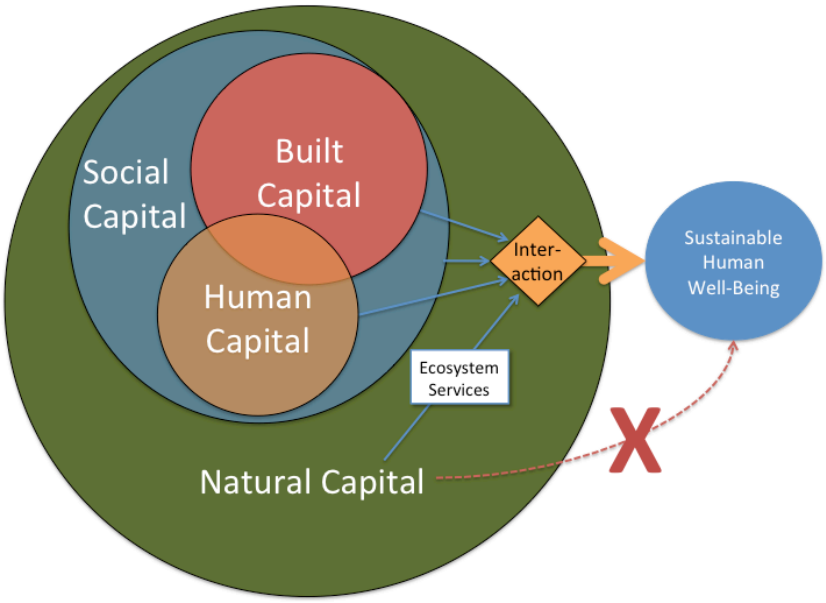

Figure 2. Interaction between social, built, human, and natural capital to contribute to well-being

Types of natural capital

According to Costanza and Daly (1992), there are two broad types of natural capital. The first is renewable natural capital, such as ecosystems, which are active and self-maintaining using solar energy, they are analogous to machines and are subject to entropic depreciation. The second is non-renewable natural capital, such as mineral deposits and fossil fuels, are more passive and they generally do not produce services until extracted. They are analogous to inventories and therefore, are subject to liquidation (Costanza & Daly 1992, p. 38).

Prugh et al (1995) describes a third category of natural capital, a hybrid one that can be called cultivated natural capital, which includes agricultural and aquacultural systems, as well as planted forests, among other things. The main characteristic of this type of natural capital is that its components are not man-made, but they are not completely natural either (T. Prugh et al. 1995, p. 52).

Ecosystem Services concept

Ecosystem services are defined as "the benefits that people obtain from ecosystems" (MEA 2005). A more complete definition of ecosystem services is "the benefits people derive from functioning ecosystems, the ecological characteristics, functions, or processes that directly or indirectly contribute to human well-being" (Costanza et al. 2011).

Although these definitions of ecosystem services are very straightforward, they have been the subject of debate for two decades, and therefore, some clarification is needed. First, it is important to distinguish between ecosystem processes and functions, on the one hand, and ecosystem services on the other. Ecosystem processes and functions refers to biophysical relationships that exist regardless of whether or not humans benefit. The opposite is the case with ecosystem services, which only exist if they contribute to human well-being (Braat 2013).

This human-dependent definition of ecosystem services has led some to argue (Thompson & Barton 1994; McCauley 2006) that the concept represents an anthropocentric, utilitarian, or instrumental view of nature, where it only exists to service humans. Nevertheless, the goal of the concept of ecosystem services is not to be anthropocentric, it is to recognize the dependence of humans on nature for their well-being and their survival, and to visualize *Homo sapiens* as an integral part of the current biosphere. Moreover, instead of implying that humans are the most and only thing that matters, the concept of ecosystem services implies that the whole system is also important, both to humans and to the other species we are interdependent with.

Types of ecosystem services

Pearce (1998) classifies the goods and services that flow from natural capital into four categories: 1) supply of natural resource inputs to the economic production process (e.g., water, genetic diversity, and soil quality), 2) assimilation of waste products and residuals from the economic process, 3) source of direct human welfare though aesthetic and spiritual appreciation of nature, and 4) support systems-biogeochemical cycles and general ecosystem functioning (Pearce 1988).

These four categories were used almost two decades later in the Millennium Ecosystem Assessment under the names of provisioning, regulating, cultural, and supporting services:

- **Provisioning services,** such as timber, water, fiber, and food. A clear example of how these services interact with the other three types of capital is fishing activity, where fish provided to people as food requires fishing boats (built capital), fishermen (human capital), and fishing communities (social capital).
- **Regulating services,** such as pollination, flood control, water regulation, pest control, climate control, water purification, and air quality maintenance. For example, storm protection provided by wetlands (natural capital) to infrastructure such as hotels and houses on the coast (built capital), protecting its residents and other members of the community. Contrary to provisioning services, these services are not marketed.
- **Cultural services** that provide spiritual, recreational, and aesthetic benefits. A recreational benefit requires natural capital such as a waterfall, built capital like a trail and a road, human capital that appreciate the waterfall, and social capital, such as friends and family and the institutions that make the waterfall accessible.
- **Supporting services,** such as photosynthesis, nutrient cycling, and soil formation. These types of services do not require the interaction with human, social, and built capital; they affect human well-being indirectly by maintaining key processes that are necessary for the other three types of services. Using this description of supporting services, some scholars have argued that instead of ecosystem services, they are ecosystem functions. Although this is true, supporting services can be used as a proxy to evaluate services in the other categories if more direct measures are not available (Costanza et al. 2011; MEA 2005)

Costanza et al (1997) identified 17 ecosystem services. Other key reports and initiatives, such as the Millennium Ecosystem Assessment (already mentioned above), The Economics of Ecosystems and Biodiversity (TEEB), and, more recently, the Common International Classification of Ecosystem Services (CICES), have established classifications of ecosystem services in order to

frame and enable discussions, assessments, modelling, and valuation. Table 1 compares these four ecosystem services classification systems, making evident that they are broadly similar.

Table 1. Comparison of four of the main ecosystem services classification systems used worldwide and their differences and similarities (Costanza et al 2017)

	Costanza et al., 1997 (a)	Millennium Ecosystem Assessment, 2005	TEEB, 2010	CICES 4.3 (v. 2013) (b)
Provisioning	Food production (13)	Food	Food	Biomass - nutrition
	Water supply (5)	Fresh water	Water	Water
	Raw materials (14)	Fibre etc.	Raw materials	Biomass – Fibre, energy & other materials
		Ornamental resources	Ornamental resources	
	Genetic resources (15)	Genetic resources	Genetic resources	
		Biochemicals and natural medicines	Medicinal resources	
	X	X	X	Biomass - Mechanical energy
Regulating & Habitat	Gas regulation (1)	Air quality regulation	Air purification	Mediation of gas- & air flows
	Climate regulation (2)	Climate regulation	Climate regulation	Atmospheric Comp. & climate regulation
	Disturbance regulation (storm protection & flood control) (3)	Natural hazard regulation	Disturbance prevention or moderation	Mediation of air and liquid flows
	Water regulation (4) (e.g. natural irrigation & drought prevention)	Water regulation	Regulation of water flows	Mediation of liquid flows
	Waste treatment (9)	Water purification and waste treatment	Waste treatment (esp. water purification)	Mediation of waste, toxics and other nuisances
	Erosion control & sediment retention (8)	Erosion regulation	Erosion prevention	Mediation of mass-flows
	Soil formation (7)	Soil formation *[supporting service]*	Maintaining soil fertility	Maintenance of soil formation and composition
	Pollination (10)	Pollination	Pollination	Life cycle maintenance (incl. pollination)
	Biological control (11)	Regulation of pests & human diseases	Biological control	Maintenance of pest- and disease control
Supporting & Habitat	Nutrient cycling (8)	Nutrient cycling & photosynthesis, primary production,	X	X
	Refugia (12) (nursery, migration habitat)	'Biodiversity'	Lifecycle maintenance (esp. nursery)	Life cycle maintenance, habitat and Gene pool protection
			Gene pool protection	
Cultural	Recreation (16), incl. eco-tourism & outdoor activities	Recreation & eco-tourism	Recreation & eco-tourism	Physical and experiential interactions
	Cultural (17) (incl. aesthetic, artistic, spiritual, education & science)	Aesthetic values	Aesthetic information	
		Cultural diversity	Inspiration for culture, art & design	
		Spirit. & religious val.	Spiritual experience	Spiritual and/or emblematic interactions
		Knowledge systems Educational values	Information for cognitive development	Intellectual and representative interactions

Ecosystem Services valuation

The concept of value

Ecosystem services can be valued through different methods depending on the service, but before explaining these methods it is important to understand the concept of value in this context.

A good start to better comprehend the theory behind the valuation of goods and services is the distinction that Adam Smith made in the 18th century between exchange value and use value, wherein he used the diamond-water paradox to explain it: diamonds have a high exchange value and people are willing to pay a great price depending on the quality of the diamond, but diamonds have low use value because they are mainly useful as jewellery (among other uses that were implemented after Smith). Water, on the other hand, has a low exchange value, which means that people would pay very low prices to consume it, but the use value of water is high since it is a resource we need in order to survive.

Smith used this paradox to dismiss the use value as a basis for exchange value, and he instead formulated a cost of production theory of value based on wages, profit, and rent as the source of exchange value. He suggested a labour theory of exchange value using the beaver-deer example: if it takes twice the labour to kill a beaver than to kill a deer, then one beaver will be sold for as much as two deer. Therefore, when labour is the only scarce factor, services and goods will be "priced" based on the ratio of labour used (Farber et al. 2002). It is worth noting that this point of view of value completely excluded natural capital, perhaps because at the time it was not a scarce resource.

In the 20th century, the "marginal" revolution in value theory was originated through the convergence of related streams of economic thought. Menger stated that the intensity of desire for one additional unit declines with successive units of the good. Exchanging the term "desire for one additional unit" with the term "marginal utility" results in the economic principle of diminishing marginal utility. The marginal utility theory of value is of great importance in the valuation of ecosystem services, because it can be used to measure use values instead of just exchange values, in monetary units (Farber et al. 2002).

The exchange value of goods and services is determined by the willingness to pay (WTP) to obtain them or the willingness to accept (WTA) compensation for losing them. WTP and WTA can be based on marginal changes in the availability of these goods and services, or on larger changes, including their complete absence. Exchange-based values of goods and services are determined by the prices at which they are exchanged. Overall, economists set the value of a good based on want satisfaction and pleasure, meaning that things only have value if they are desired, which is a problematic point of view in valuing natural capital, as explained later. Furthermore, as the good becomes scarcer, the desire increases, and therefore, its value (Farber et al. 2002).

Valuation

As stated before, ecosystem services are the benefits people derive from ecosystems; they are provided by natural capital in combination with built, social, and human capital. The value of ecosystem services is therefore the relative contribution of ecosystems to well-being (Turner et al. 2016). This contribution can be expressed in various units (any units of the four types of capitals), where monetary units are often the most used and convenient since most people understand values in these units. Nevertheless, other units, such as time, energy, and land, can also be used. The selection will depend on which units help to better communicate to different stakeholders in a given decision-making context (Costanza, et al. 2014). Valuation allows a more efficient use of limited funds by identifying where environmental protection and restoration is economically most significant, supporting the determination of the amount of compensation that should be paid for the degradation and/or loss of ecosystem services and improving the financial mechanisms (e.g., incentives) for the conservation and sustainable use of natural capital (e.g., Payment for Ecosystem Services) (De Groot et al. 2012).

The value of ecosystem services can also be estimated by determining the cost to replicate them by artificial means (Costanza et al. 1997), for example how much it would cost to a farmer to pollinate his crops artificially. It is useful to attempt to calculate the impact in human well-being from changes in quantity or quality of natural capital that can occur due to different development decisions (Costanza et al. 1997). Valuation is therefore a tool for evaluating the

trade-offs required to achieve a shared goal, where in the past and in the present these trade-offs have been addressed mainly through marketed goods and services (e.g., fuel or food) using commodity prices, leaving outside the equation other goods and services that currently do not have a price but that contribute equally or even more greatly to well-being (Turner et al. 2016).

Valuing ecosystem services has been criticized as unwise or even impossible because we supposedly cannot put a value on "intangibles" like human life and nature. In reality, we implicitly value these things on a daily basis through, for example, measures to protect human life, such as construction standards for housing and public infrastructure that will require spending more money in order to preserve human lives (Costanza et al. 1997). Therefore, the overall goal is not to put a price tag on nature for exchange purposes, but to visualize the effect of a change in ecosystem services provision to human well-being in terms of a rate of trade off against other things people value (Turner et al. 2003).

Valuation methods

After the identification, quantification, and mapping of ecosystem services for a particular area or scale, there are different types of methods used to conduct a Total Economic Valuation (TEV). These can be divided into revealed preference, stated preference, and non-preference based methods. Revealed preference methods to estimate the benefits from ecosystems are based on market prices, which limits the use of these methods to only a few ecosystem services that are traded in markets (mostly provisioning services) (Turner et al. 2016). Revealed preference methods analyse the choices of people in real world settings and infer the value from those observed choices (Costanza et al. 2011). Non-preference methods recognize the limits of an individual's information about ecosystem services' connection to their well-being and use modelling and other techniques to estimate these connections.

Stated preference methods try to construct pseudo markets through the use of surveys in which people are asked to state their willingness-to-pay for ecosystem services that are not traded in current markets. Therefore, these methods rely on the response of people to hypothetical scenarios (Costanza et al. 2011). Stated preference approaches have limitations because people surveyed often do not completely understand or are not aware of the relation

between healthy ecosystems and human well-being, also because they do not feel comfortable in stating trade-offs for ecosystems in monetary units, and finally because the willingness-to-pay can be significantly different to the real payment when it comes to that point (Turner et al. 2016).

Table 2 summarizes the different methods for ecosystem services valuation using conventional economic valuation and non-monetizing valuation (from Turner et al [2016], which is an adaptation from Farber et al. [2006]).

Table 2. List of methods for ecosystem services valuation.

Conventional economic valuation	Revealed-preference approaches	*Travel cost*: valuations of site-based amenities are implied by the costs people incur to enjoy them (e.g., cleaner recreational lakes).
		Market methods: valuations are directly obtained from what people must be willing to pay for the service or good (e.g., timber harvest)
		Hedonic methods: the value of a service is implied by what people will be willing to pay for the service through purchases in related markets, such as housing markets (e.g., open-space amenities)
		Production approaches: service values are assigned from the impacts of those services on economic outputs (e.g., increased shrimp yields from increased area of wetlands)
	Stated-preference approaches	*Contingent valuation*: people are directly asked their willingness to pay or accept compensation for some change in ecological service (e.g., willingness to pay for cleaner air)
		Conjoint analysis: people are asked to choose or rank different service scenarios or ecological conditions that differ in the mix of those conditions (e.g., choosing between wetlands scenarios with differing levels of flood protection and fishery yields)
	Cost-based approaches	*Replacement cost*: the loss of a natural system service is evaluated in terms of what it would cost to replace that service (e.g., tertiary treatment values of wetlands if the cost of replacement is less than the value society places on tertiary treatment)

		Avoidance cost: a service is valued on the basis of costs avoided, or of the extent to which it allows the avoidance of costly averting behaviours, including mitigation (e.g., clean water reduces costly incidents of diarrhea)
Non-monetizing valuation	---	*Individual index-based methods*, including rating or ranking choice models, expert opinion *Group-based methods*, including voting mechanisms, focus groups, citizen juries, stakeholder analysis

Due to the nature of the service, each ecosystem service can be valuated through one or more particular methods. For each service, the amenability to economic valuation and the transferability across sites will vary from low to high. Table 3 summarizes the set of methods that are appropriate to value each ecosystem service (Turner et al. 2016).

Table 3. Valuation methods for each ecosystem service (Farber et al. 2006).

Ecosystem services		Amenability to economic valuation	Most appropriate method for valuation	Transferability across sites
Provisioning service	Water supply	High	AC, RC, M, TC	Medium
	Food	High	M, P	High
	Raw material	High	M, P	High
	Genetic resources	Low	M, AC	Low
	Medicinal resources	High	AC, RC, P	High
	Ornamental resources	High	AC, RC, H	Medium
Regulating services	Gas regulation	Medium	CV, AC, RC	High
	Climate regulation	Low	CV	High
	Disturbance regulation	High	AC	Medium
	Biological regulation	Medium	AC, P	High
	Water regulation	High	M, AC, RC, H, P, CV	Medium
	Soil retention	Medium	AC, RC, H	Medium
	Waste regulation	High	RC, AC, CV	Medium High
	Nutrient regulation	Medium	AC, CV	Medium
Cultural services	Recreation	High	TC, CV, ranking	Low
	Aesthetics	High	H, CV, TC, ranking	Low

	Science and education	Low	Ranking	High
	Spiritual and historic	Low	CV, ranking	Low

AC=avoided cost, CV=contingent valuation, H=hedonic pricing, M=market pricing, P=production approach, RC=replacement cost, TC=travel cost.

Due to constraints in time and budget, it is often not possible to conduct original/primary studies to value ecosystem services (Wilson & Hoehn 2006; Plummer 2009), which has led to a wider use of secondary data (Richardson, Loomis, Kroeger, & Casey 2015) for this purpose, through valuation techniques such as value/benefit transfer. Although this technique has limitations, it is sometimes the only option to inform policy decisions that require a first approximation to natural capital valuation (Richardson et al. 2015).

In simple terms, value transfer consists in "applying economic value estimates from one location to a similar site in another location" (Plummer 2009). The site where primary data was collected and processed is called the study site, and the site to which this data (i.e., ecosystem services values) is going to be applied is called the policy site (because the values are commonly used for policy decisions such as land use change or the establishment of financial mechanisms) (Plummer 2009). The transfer can be spatial (across different sites, national, or international) or temporal (where the study site and the policy sites are different moments in time) (Navrud & Bergland 2004).

Other authors have proposed the following definitions of value transfer, all of them sharing the core elements of the technique:

- "Transfer of original ecosystem service value estimates from an existing 'study site' or multiple study sites to an unstudied 'policy site' with similar characteristics that is being evaluated." (Richardson et al. 2015)
- "Transposition of monetary environmental values estimated at one site (study site) through market-based or non-market-based economic valuation techniques to another site (policy site)" (Brouwer 2000)

Although the valuation technique is often referred as benefit transfer, Navrud states that the method can also be related to the transfer of damage estimates, and thus a more accurate term would be value transfer (Navrud & Bergland 2004), which is going to be used henceforth.

The aggregation of these methods through a value transfer make the technique useful in academic and policy settings, in which ecosystem services values are not required with a high level of accuracy but they need to be accurate enough to support a project or policy. Nonetheless, they are not suitable when more accurate values are required, in cases such as the calculation of compensation payments for environmental damages (polluter pays principle) (Navrud & Ready 2007).

Difficulties in valuing ecosystem services

Valuing natural capital is far from a perfect science, but is without any doubt a needed one. The following main difficulties when conducting these assessments are identified:

- Marginality: The data used in ecosystem services are "marginal" values rather than aggregated global values; this is because what it is calculated is the value of ecosystem services degradation or loss.
- Double counting: This problem can often occur because many ecosystem services are not complementary, which means that the provision of one is precluded by others.
- Typological issues: These are related to the design and strategy of the valuation assessments, where it is important to distinguish between valuations of the in-situ ecosystem stock and estimates of the value of the flow of goods and services from a given stock.
- Spatial and temporal transfer: These difficulties are specifically for the aggregation method of basic value (or benefit) transfer, including the requirement of good quality studies of similar situations, the potential change of characteristics between time periods, and a failure to assess novel impacts (i.e., thresholds or resilience).
- Distribution of benefits and costs: Developing countries invest high local costs to natural

capital conservation that yield large global benefits, in contrast to developed countries that tend to incur in relatively low local costs that produce lower global benefits.

Natural capital and sustainability

A key point is the understanding of the relation between sustainability and the maintenance of capital stocks. Ekins explains that if sustainability depends on the maintenance of the capital stock, then there are two possibilities: 1) maintaining the total stock of capital, allowing substitutions between its components, or 2) determining whether certain components of capital, mainly natural capital, are non-substitutable. Ekins continues elaborating on these two possibilities by framing them under two types of sustainability: 1) weak sustainability, which considers that natural capital can be replaced completely by built capital under the perception that welfare is not dependent on a specific form of capital, and 2) strong sustainability, which considers complete substitution of natural capital by built capital to be impossible since natural capital provides a unique contribution to welfare, and ultimately, it is the inputs for built capital and the basis of critical life support systems (Ekins et al. 2003, p. 167).

The concept of natural capital, as well as its research and policy implications, becomes relevant more than ever in the current national and global economic growth strategy. In the past (mainly before the Industrial Revolution), we lived in what some scholars call an empty world, empty of humans and their artefacts, full of natural resources. Now, we live in a full world, full of humans and their artefacts, with an increasingly reduced natural environment. In the former world, the limiting factor was built capital, while natural capital and social capital were abundant, in the latter world quite the contrary abounds.

In order to recognize natural capital as a limiting factor, and therefore, its need of conservation and sustainable consumption, a different vision of the interaction between the economic and ecological systems is needed. Fenech et al. propose that, instead of looking at the ecological system as part of the economic system, we need to consider the economy as part of the ecosystem (Fenech et al. 2003, p. 5).

The consideration of the economy as part of the ecosystem acknowledges the limits to growth of the economy since the ecosystem is finite. Costanza and Daly state that growth is related to throughput increase, which is destructive of natural capital, with the negative consequence of having higher costs in the medium and long term than the benefits gained in the short term (Costanza & Daly 1992, p. 43). This cost-benefit analysis for natural capital is often ignored by economic interests, undervaluing natural capital and only recognizing its value when it is lost (Ehrlich et al. 2012 p. 70). Development, on the contrary, means an increase of efficiency and quality improvement, and therefore, does not reduce natural capital (Costanza & Daly 1992, p. 43).

From the natural capital perspective, development under this framework would mean that natural income must be sustainable, which should be at least the case for renewable natural capital. Since non-renewable natural capital is reduced with use, income can be constant only if the total natural capital (renewable natural capital plus non-renewable natural capital) is maintained constant, which implies a certain level of reinvestment of the non-renewable natural capital consumed into the renewable natural capital (Costanza & Daly 1992, p. 43). This is relevant, especially for low income countries, since they have a higher dependency on natural capital both for growth and development (Pearce 1988).

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Future scenarios for the value of ecosystem services in Latin America and the Caribbean to 2050

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Abstract

We explore the implications of four scenarios for the value of ecosystem services provided by terrestrial ecosystems to the year 2050 for Latin America and the Caribbean, based on the Great Transition Initiative scenarios and previous studies at a global scale. We estimated the current ecosystem services value (ESV) of the 33 countries that make up this region to be $US15.3 trillion/year. By modelling the four future scenarios, we estimated that there is a potential for ESV to decrease to $8 trillion/year (for the "Fortress World" scenario) or an increase to $19 trillion/year (for the "Great Transition" scenario), a difference of a 47% decrease or a 25% increase. Our results indicate that adopting appropriate policies could greatly enhance human well-being and sustainability in the region and help to achieve the UN Sustainable Development Goals (SDGs).

Key words: scenario planning, economic value, ecosystem services, natural capital

Introduction

The value of natural capital becomes evident in a region such as Latin America and the Caribbean (LAC)[1], which holds sixty per cent of global terrestrial biodiversity as well as a diverse

[1] Includes 33 countries as defined by the United Nations Environment Programme (UNEP)

marine and freshwater flora and fauna. Six of the most biodiverse countries in the planet are in this region (Brazil, Colombia, Ecuador, Mexico, Peru and Venezuela), including the most biodiverse habitat on Earth, the Amazon rainforest (UNEP-WCMC, 2016). Due to the extension of its area and the historical context of LAC, it is also highly diverse in terms of economy, geography, and policy, which determines the route of development that the region has followed over the last decades.

Data from Steffen et al. (2015), provides a general picture of this development path. In the period from 1750 to 2010, Mesoamerica's population increased 2,157%, with Costa Rica having the highest increase, 13,659%. In the Caribbean, population increased by 4,134%, with Dominican Republic having the highest increase with 8,070%. The population of South America increased by 5,008%, with Argentina being the country with the highest increase, 1,3455%.

Understanding the population's migration to urban settlements is critical as the associated land-cover change has one of the most significant impacts on natural capital. In the same period, 1750 to 2010, Mesoamerica increased its urban population by 51,097%, Costa Rica again with the highest increase, 162,828%. The Caribbean increased by 70,838%, with the greatest change in Dominica, 429,794%. South America shows an increase of urban population of 68,785%, with the highest increase in Brazil, 155,516%.

The same data set describes the increase of economic activity of LAC and its sub-regions, for the period of 1969 to 2010. Mesoamerica increased its GDP by 310% in these five decades, Belize with the highest increase, 833%; the Caribbean increased by 250% its GDP with the highest amount in Dominican Republic, 797%; and South America increased the same indicator by 314% with the highest increase in Chile, 431%.

These indicators show the development path that LAC has followed, which is characterized by examples of success in sustainability, as well as by social and economic challenges. For example, in the period between 1990 and 2014, the total terrestrial area of LAC under protection increased from 8.8% to 23%, a 266% increase (UNEP, 2016). On the other hand, urban areas have been growing in LAC, increasing the urban population by more than 35 million people in five years (2010-2015 period), with a projected increase by 2025 to a total of 597 million persons (UNEP, 2016). This urban development, combined with economic growth and inequity,

is one of the most significant threats to biodiversity in many areas of the region (Pauchard & Barbosa, 2013).

Having broadly described the region's past development and its environmental and social implications, this paper explores through a scenario planning approach, how the economic value of natural capital in LAC might change in the future. We estimate the change of the value of ecosystem services by 2050 under different development scenarios for the region, with the main goal to inform policymakers the consequences that land cover change decisions have on natural capital. This paper is based on the global study on the future value of ecosystem services conducted by Kubiszewski et al. (2017), who estimated that global value of ecosystem services can decline by $51 trillion/year or increase by $30 trillion/year, depending on the development scenario. Other studies have also used data from this global one to produce regional estimates, such as the case of Asia and the Pacific (Kubiszewski et al., 2016).

Scenario planning

Scenario planning or analysis is a structured process of generating future possibilities which have social-economic and environmental implications (Bohensky et al., 2011). Scenarios are narratives that consider how alternate futures may unfold from combinations of highly influential and uncertain drivers, and their interaction with more certain driving forces (O'Brien, 2000). Furthermore, scenarios are not predictive models, forecasts or predictions, rather explorations of *plausible* (not probable) futures (Peterson, Cumming, & Carpenter, 2003).

Scenario planning is based on four assumptions: 1) the future is unlike the past, and is significantly shaped by human choice and action, 2) the future cannot be foreseen, but exploring possible futures can inform present decisions, 3) there are many possible futures, scenarios therefore map within a "possibility space", and 4) scenario development involves both rational analysis and creative thinking (Costanza, 2014).

Although some aspects from the future world created in each scenario can potentially occur, these "fictional" worlds are best viewed as caricatures of reality that allow the public to

learn and take better decisions regarding the factors that are being evaluated (Costanza et al., 2015). The majority of scenarios developed around the world for multiple purposes, fall into a small number of types or "archetypes", which cover topics such as growth, transformation, collapse, and discipline/restraint narratives (Bohensky et al., 2011).

In this study, the four scenarios that we used are a synthesis of prior scenario studies and based around the four "Great Transition Initiative" (GTI) archetypes (Hunt et al., 2012) created by an international network of scientists, using models and regional analyses (Raskin et al., 2002; McGrail, 2011). In general, the driving forces of these scenarios are demographics, considering population growth and urbanization; economics, specially growing markets, regulation and people's preferences; social issues such as inequality and poverty; culture in a globalized world; technological advance; environment, through a global and interconnected vision; and governance, considering a trend towards democratization and decentralization of authority (Raskin et al., 2002).

These are the four scenarios from GTI, as describe in its website (http://www.tellus.org/results/scenarios.html):

- **Market Forces (MF)**: The Market Forces scenario is a story of a market-driven world in the 21st century in which demographic, economic, environmental, and technological trends unfold without major surprises. Continuity, globalization and convergence are key characteristics of world development – institutions gradually adjust without major ruptures, international economic integration proceeds apace and the socioeconomic patterns of poor regions converge slowly toward the development model of the rich regions. Despite economic growth, extreme income disparity between rich and poor countries, and between the rich and poor within countries, remains a critical social trend. Environmental transformation and degradation are a progressively more significant factor in global affairs.
- **Policy Reform (PR)**: The Policy Reform scenario envisions the emergence of strong political will for taking harmonized and rapid action to ensure a successful transition to a more equitable and environmentally resilient future. Rather than a projection into the future, the Policy Reform scenario is a normative scenario constructed as a backcast from the future. It is

designed to achieve a set of future sustainability goals. The analytical task is to identify plausible development pathways for reaching that end-point. Thus, the Policy Reform scenario explores the requirements for simultaneously achieving social and environmental sustainability goals under high economic growth conditions similar to those of Market Forces.

- **Fortress World (FW)**: The Fortress World scenario is a variant of a broader class of Barbarization scenarios, in the hierarchy of the Global Scenario Group (Gallopín et al. 1997). Barbarization scenarios envision the grim possibility that the social, economic and moral underpinnings of civilization deteriorate, as emerging problems overwhelm the coping capacity of both markets and policy reforms. The Fortress World variant of the Barbarization story features an authoritarian response to the threat of breakdown. Ensconced in protected enclaves, elites safeguard their privilege by controlling an impoverished majority and managing critical natural resources, while outside the fortress there is repression, environmental destruction and misery.
- **Great Transition (GT)**: The Great Transition scenario explores visionary solutions to the sustainability challenge, including new socioeconomic arrangements and fundamental changes in values. This scenario depicts a transition to a society that preserves natural systems, provides high levels of welfare through material sufficiency and equitable distribution, and enjoys a strong sense of local solidarity.

The future by 2050 of LAC under these scenarios poses great challenges and opportunities for sustainable development in the region. Taking a more in-depth look at the data on the scenarios from the Great Transition Initiative for Latin America (See Table 1), it shows that population could increase the most, under the Fortress World scenario, going from 557 million people in 2005 to 817 million; and it could increase the least under the Great Transition scenario, reaching 692 million people by 2050. More people in the future can also have a significant impact on poverty, with the most populated scenario (Fortress World) also having the most people with hunger, 81 million, while the Great Transition could have 1 million people under this condition.

These two variables, population and hunger, can be also reflected in the future economic conditions of the region, which present its extremes again under the Fortress World and the

Great Transition scenarios. The Fortress World would produce the lowest economic activity, with a GDP of 15 trillion and a per capita income of $19,000, while the Great Transition scenario presents the highest GDP of all scenarios, 18 trillion, as well as the highest income per capita, $27,000. These figures show some of the socio-economic factors that make LAC under the Great Transition scenario a region with high levels of welfare through material sufficiency and equitable distribution.

The Market Forces scenario would have the highest values of those variables related more to commercial activity (although we can argue that all variables interact with each other in some way), which is the core narrative of this scenario. Agriculture under this scenario has the highest production, with crop outputs and livestock outputs of 2 kilo tonnes and 204 million tonnes respectively. The impact of the high levels of production of these two economic activities, plus other ones in the Market Forces scenario, is reflected in the use of natural resources, which under this scenario are the highest. For example, water use could reach 0.5 trillion cubic meters, the highest of all the four scenarios, and energy demand is the second highest with 91 EJ. A fully market-oriented LAC would also experience the highest CO_2 emissions and the lowest forest cover, which are in part a product of the high agricultural activity.

On the other hand, the Great Transition scenario depicts a LAC with the lowest crop output (2.6 million kt and livestock output (159 million t) which are possible related with this scenario having the lowest population of all four, and the lowest consumption of meat. This as well is reflected in the Great Transition scenario having the lowest water footprint (0.2 trillion m^3) and carbon footprint (0.04 GtC), along with the highest forest cover (798 billions of ha). These environmental indicators show that the Great Transition scenario would represent a true green economy, one in which the levels of GDP and income per capita are the highest, while natural resources are consumed at the lowest level.

Table 1. Social-economic indicators of the four scenarios of the Great Transition Initiative.

Indicator	Market Forces	Policy Reform	Great Transition	Fortress World
Population (10^6 people)	778	739	692	817
Total GDP (10^9 $PPP)	16,539	18,056	18,844	15,630
Income ($PPP per capita)	21,259	24,429	27,215	19,133
Hunger (millions of people)	55	26	1	81

Energy Demand (EJ)	91.31	50.31	38.37	92.60
Crop Output (million kt)	3.17	2.77	2.58	3.04
Livestock Output (million t)	204.59	200.35	159.14	168.36
Forest Area (kha)	574,218	690,607	798,790	581,282
Water Use (10^12 m^3)	0.50	0.24	0.23	0.48
CO2 Emissions (GtC)	1.70	0.41	0.04	1.63
Quality of Development Index	0.44	0.55	0.68	0.38

Source: Great Transition Initiative, 2018

Methods

Land-cover change scenarios

Using the interactive web tool from the GIT website, Futures in Motion (www.tellus.org/results/results_World.html), we estimated land use change (urban, cropland, forest, grassland, desert), population, economic activity (GDP), and inequality, among other variables for the four scenarios described in the previous section to the year 2050.

Because wetlands are not included in the GTI scenarios, we estimated its cover based on past trends loss seen between 1997 and 2011 for the MF and FW scenarios (Costanza et al., 1997, Costanza et al., 2014; Millennium Ecosystem Assessment, 2005), a policy of "no net loss" for the PR scenario, and a wetland restoration policy for the GT scenario based on achieving wetland areas similar to those in 2000 (Costanza et al., 2014; Mitsch & Day Jr, 2006; Gascoigne et al., 2011).

Unit value change scenarios

The change in value of ecosystem services from each land-cover in the four scenarios respectively, was calculated in relation with two factors: 1) change in area covered by each ecosystem type, and 2) change in the "unit value" of each ecosystem (i.e. aggregate value of all the marketed and non-marketed ecosystem services per hectare per year) due to degradation or restoration. Management policies of land and water in each country determine the change of unit values of ecosystem services. These changes were separated out by evaluating the scenarios in two ways: a) using the 2011 unit values estimated by Costanza et al. (2014) and only changing land use, and b) changing both unit values and land use. Moreover, the 2011 unit values of each

ecosystem are averages of values found in studies on natural capital valuation, and they were carefully evaluated by the TEEB initiative (de Groot et al, 2012). Due to the scale of the study, our estimates are a simplification of the reality, but they were sufficient for the purposes of this exploratory exercise.

The unit value changes in each scenario were calculated based on management and policy assumptions in each one of them. Furthermore, these changes also take into account the change in preferences of the people living in each scenario. For example, in the Fortress World scenario, it is assumed that society will follow a development path based on inequality and unsustainable use of natural resources, and therefore,, unit values of ecosystems would decrease by 20%, and in the opposite case, in the Great Transition scenario, in which society achieve sustainable development, unit values would increase by 20%. These assumed percentages were based in a general way on the estimates from the Bateman et al. (2013) study of six future scenarios for the United Kingdom; they were used here as an illustration on how each development path described in each scenario have plausible changes on the value of natural capital, and therefore, can be applied in any region of the world. The following assumptions were made for each scenario:

- **Market Forces:** decrease in consideration of the environmental and non-market factors resulting in an average *10 per cent reduction in unit values* from their 2011 levels. In this scenario, climate change has not been dealt with.
- **Fortress World**: *significant* decrease in consideration of environmental and non-market factors resulting in an average *20 per cent reduction in unit values* from their 2011 levels. In this scenario, climate change has accelerated.
- **Policy Reform**: slight improvement from 2011 policies and management leading to *no significant change in unit values* from their 2011 estimates. In this scenario, climate change has been moderated.
- **Great Transition**: *significant* increase in consideration of environmental and non-market factors resulting in an average *20 per cent increase in unit values* from their 2011 levels. In this scenario, climate change has been addressed.

Mapping land-cover change

The spatial data of the change of land-cover for each scenario was created via a loose coupling with the scenario projection modelling. Each scenario was modelled to generate a change in land-cover at a 1km^2 resolution for the following types: urban, wetland, cropland, forest, grassland, and desert. A modified version of the GlobCov (Global Land Cover map) from the European Space Agency was used as the original base data. In each scenario, land-cover increased or decreased according to the percentage change indicated in the previous sub-section, and these changes were adjacent to the existing original extent of that land-cover. Precedence for these land-cover changes occurred in the following order: urban, wetland, cropland, forest, rangeland/grassland, and desert. This precedence worked in such a way that all previous land-cover transitions are excluded from subsequent conversion (e.g. cropland cannot replace urban or wetlands).

Results

Values in 2011

The total terrestrial ecosystem service value (ESV) in 2011 of Latin America and the Caribbean is USD $15.3 trillion/year (Table 2). As expected, Brazil had the largest ESV, USD $6.8 trillion/year, due to its size and extensive rain forest cover. Argentina and Bolivia, although following Brazil in ESV in the region, have less than a third of the value with USD $2.2 and $1.3 trillion/year, respectively. Mexico is the country with the highest ESV in Mesoamerica, with a value of USD $849 billion/year, accounting for 72% of the ESV of this sub-region; while in the Caribbean Cuba has the highest, USD $68 billion/year.

Looking at the region through a lens of ecosystem services value per area per year, South America has the highest ESV of the three regions, USD $7900/ha/year. Nevertheless, at a country level, the Caribbean have the top 3 countries with the highest ESV per area per year of the region, The Bahamas ($23 thousand/ha/year), Saint Vincent and the Grenadines ($20 thousand/ha/year)

and Antigua and Barbuda ($18 thousand/ha/year). In Mesoamerica, Costa Rica has the highest ESV per hectare at USD $8 thousand/ha/year.

Differences in values of ecosystem services per hectare are due to varying land-use management practices and policies in the countries and heterogeneity distribution of ecosystem services across the region. For example, the “weight" that certain land-covers or ecosystems have on the ESV is evident in countries such as Brazil, in which forest accounts for half of the land-cover of the country (Figure 1a), but they provide a third of the ESV. Furthermore, while tidal marshes and mangroves cover only 1% of the territory, they provide 31% of the countries ESV (Figure 1b), this is because these ecosystems are the most valuable of all assessed, USD $194000/ha, a very high value compared to ecosystems such as tropical forests that are valued in USD $5400/ha (Costanza et al., 2014). In Brazil, out of the 7408km of its coastline, 6786km contain mangrove forests (Schaeffer-Novelli, Cintrón-Molero, Soares, & De-Rosa, 2000), which provide a wide arrange of ecosystem services (Estrada, Soares, Fernadez, & de Almeida, 2015).

In the Bahamas, forests cover 30% of the territory, but account for only 5% of its ESV. This is also another clear example on how tidal marshes and mangroves play a key role in the provision of ecosystem services in the country. Here, these ecosystems cover 10% of the territory (Figure 2a), but they constitute 81% of the ESV of the Bahamas (Figure 2b). Despite the high contribution of mangroves to the country’s ESV, they are currently threated by several stressors such as coastal development, mainly in New Providence and Grand Bahama (Buchan, 2000).

A similar case happens with Costa Rica, where tidal marshes and mangroves cover 2% of the country (Figure 3a) but are 41% of its ESV (Figure 3b). Forest also provide a significant portion of the ESV of the country, 28%, with a forest cover of 43%. Costa Rica is known for his pioneering Payment for Ecosystem Services scheme that has played a key role in stopping deforestation by paying private land owners for the services that these ecosystems provide to society (Porras, Barton, Miranda, & Chacón-Cascante, 2013; Farley & Costanza, 2010; Pagiola, 2008); our results show that PES programs have a high potential on other ecosystems such as coastal ones.

These three countries provide a sound justification for ecosystems that provide highly valuable goods and services to society and therefore, the necessity to its conservation and promotion of financial mechanisms that are based in cost-benefit analysis.

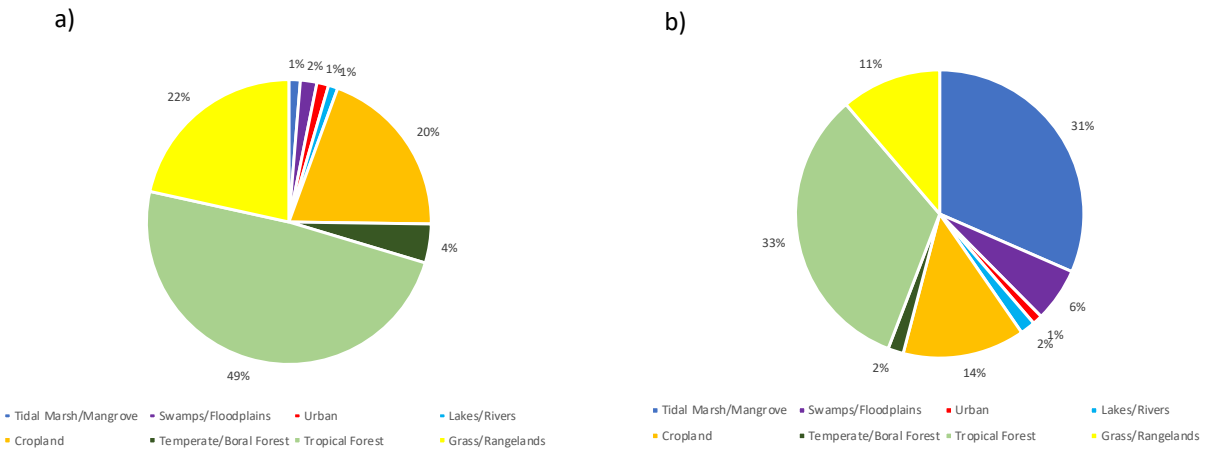

Figure 1. a) Land-cover distribution of Brazil in 2011. b) Value of ecosystem services for each land-cover in Brazil in 2011

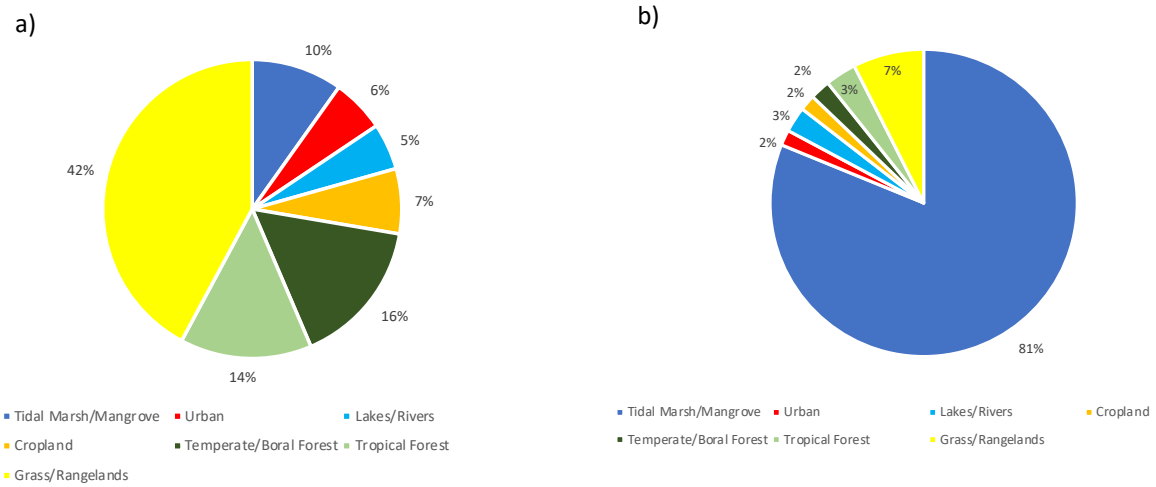

Figure 2. a) Land-cover distribution of Bahamas in 2011. b) Value of ecosystem services for each land-cover in Bahamas in 2011

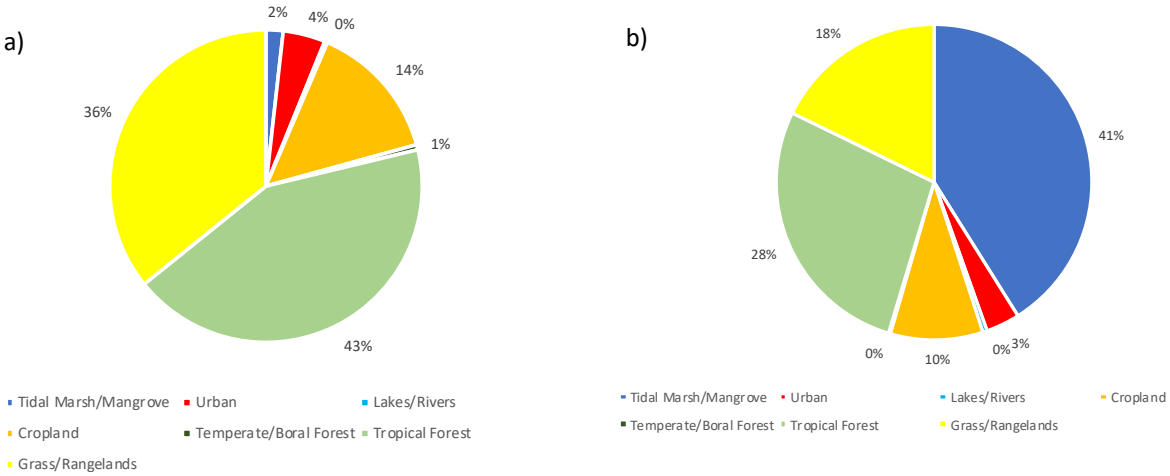

Figure 3. a) Land-cover distribution of Costa Rica in 2011. b) Value of ecosystem services for each land-cover in Costa Rica in 2011

Table 2. Terrestrial values of ecosystem services in Latin America and the Caribbean for 2011 and for 2050 under 4 scenarios.

Country	Area (km2)	GDP, PPP (2011 Million$)	ESV_2011 (Million$/yr)	S1_MF (Million$/yr)	MF % change from 2011	S2_FW (Million$/yr)	FW % change from 2011	S3_PR (Million$/yr)	PR % change from 2011	S4_GT (Million$/yr)	GT % change from 2011
Mesoamerica											
Belize	22,211	2,587	11,647	10,352	-11%	9,268	-20%	11,618	0%	13,840	19%
Costa Rica	51,410	60,138	42,444	30,740	-28%	22,144	-48%	42,672	1%	51,343	21%
El Salvador	20,680	45,998	14,953	11,058	-26%	8,850	-41%	15,061	1%	18,217	22%
Guatemala	109,691	102,318	58,364	51,519	-12%	45,974	-21%	58,853	1%	70,241	20%
Honduras	112,866	33,791	66,954	54,006	-19%	46,800	-30%	66,974	0%	80,364	20%
Mexico	1,965,721	1,893,303	848,935	763,625	-10%	676,614	-20%	859,273	1%	1,019,572	20%
Nicaragua	128,867	24,529	87,309	71,065	-19%	59,578	-32%	87,279	0%	104,884	20%
Panama	74,595	60,793	51,622	38,148	-26%	31,843	-38%	51,673	0%	62,196	20%
Total Mesoamerica	2,486,041	2,223,457	1,182,228	1,030,513	-13%	901,071	-24%	1,193,404	1%	1,420,657	20%
South America											
Argentina	2,787,501		2,212,877	1,418,025	-36%	935,071	-58%	2,194,339	-1%	2,698,339	22%
Bolivia	1,092,700	56,424	1,294,751	652,015	-50%	405,007	-69%	1,310,242	1%	1,639,570	27%
Brazil	8,523,524	2,973,856	6,768,369	4,726,633	-30%	3,717,035	-45%	6,868,298	1%	8,461,479	25%
Chile	745,770	348,602	298,938	177,484	-41%	158,005	-47%	284,881	-5%	390,255	31%
Colombia	1,142,733	533,513	717,015	538,452	-25%	468,230	-35%	740,988	3%	934,161	30%
Ecuador	257,031	150,664	160,915	120,877	-25%	105,843	-34%	163,455	2%	201,541	25%
Guyana	211,967	4,594	182,562	110,337	-40%	88,824	-51%	191,707	5%	250,956	37%
Paraguay	400,675	47,233	496,869	380,381	-23%	251,496	-49%	497,670	0%	599,140	21%
Peru	1,299,044	308,865	922,717	556,076	-40%	448,138	-51%	942,175	2%	1,202,038	30%
Suriname	145,973	7,914	141,562	83,839	-41%	64,152	-55%	145,858	3%	185,120	31%
Uruguay	178,378	60,619	125,929	88,071	-30%	67,292	-47%	126,284	0%	152,939	21%
Venezuela	916,774	500,326	691,372	460,285	-33%	371,038	-46%	715,163	3%	902,459	31%
Total South America	17,702,070	4,992,610	14,013,876.8	9,312,476	-34%	7,080,130	-49%	14,181,059	1%	17,617,998	26%
Caribbean											
Antigua and Barbuda	537	1,762	984.6	810.7	-18%	669.9	-32%	990.6	1%	1,144	16%

Bahamas, The	12,204	8,312	28,623	13,698	-52%	10,216	-64%	28,647	0%	35,302	23%
Barbados	448	4,322	322	298.7	-7%	216	-33%	329	2%	389	21%
Cuba	109,710	214,296	68,757	55,242	-20%	46,182	-33%	69,358	1%	82,987	21%
Dominica	778	728	586	428	-27%	357	-39%	563	-4%	717	22%
Dominican Republic	48,634	114,065	26,451	23,842	-10%	21,450	-19%	26,686	1%	31,803	20%
Grenada	349	1,179	288.8	264	-9%	237	-18%	293	2%	348	20%
Haiti	27,322	15,849	15,837	14,189	-10%	12,662	-20%	16,025	1%	19,111	21%
Jamaica	11,094	22,898	6,156	5,498	-11%	4,989	-19%	6,247	1%	7,396	20%
Saint Kitts and Nevis	198	1,090	201	153	-24%	138	-31%	170	-16%	243	21%
Saint Lucia	637	1,889	537	486	-10%	438	-19%	543	1%	606	13%
Saint Vincent and the Grenadines	343	1,079	692	197	-72%	148	-79%	678	-2%	852	23%
Trinidad and Tobago	5,038	39,730	6,016	3,377	-44%	2,286	-62%	6,246	4%	7,995	33%
Total Caribbean	217,292	427,199	155,453	118,485	-24%	99,988	-36%	156,775	1%	188,893	22%
Total Latin America & The Caribbean	**20,405,403**	**7,643,266**	**15,351,558**	**10,461,474**	**-32%**	**8,081,190**	**-47%**	**15,531,239**	**1%**	**$ 19,227,547**	**25%**

Future Values of Ecosystem Services

After 100 years from the Great Acceleration, in a world fully embedded in the Anthropocene, land-cover in Latin America and the Caribbean could change substantially under the four development scenarios that we assessed (Figures 4-8). With land-cover change, values on ecosystem services of the region will decline the most under the Fortress World scenario with a 47% decrease. The Market Forces scenario also results in a significant decline of ESV in the region with a 32% decrease. The Policy Reform scenario would result in almost the same ESV as it is in the present, with only a 1% increase, while under the Great Transition scenario the ESV of the region would increase in 25% (Table 2).

At the sub-regional level, despite having the smallest area of the sub-regions analyzed, the Caribbean will experience the most change in ESV in the future under three of the four scenarios, decreasing 35% under the FW scenario, and increasing 3% and 30% under the PR and GT scenarios respectively. Furthermore, South America can experience a decrease of 49% of its ESV under the FW scenario, the highest decrease of all sub-regions.

At the country level, Saint Vincent and the Grenadines show the greatest potential ESV loss among the countries in the FW scenario with a decrease of 79%. This is a decrease of USD $545 million/year since the 2011 base value, which is equal to losing approximately half of the country's GDP (USD $1 billion in 2011). Saint Vincent and the Grenadines has already the challenge of forest management in the face of increasing demands for land intended for housing and agriculture; in addition to the threats that climate change poses on Small Islands Developing Stats like this, such as coastal erosion, droughts, floods and forest fires (Ministry of Health Wellness, 2013; UNEP, 2010).

In Mesoamerica, Costa Rica is the most affected under the MF and FW scenarios with a decrease in the ESV of 28% and 48% respectively, and in South America, Bolivia is the country with the highest decrease of ESV, also under these two scenarios, with a loss of 50% and 69% respectively. Under the PR scenario, the majority of the countries show little change in their ESV except for Saint Kitts and Nevis which experiences a 16% decline. The GT scenario shows a similar increase of ESV among Mesoamerican countries (between 19% and 22%), and the highest

increase occurs in the Caribbean with an improvement of 37% in Guyana. The Caribbean is the region that will experience the greatest volatility (Figure 9).

Figure 4. Land-cover map of Latin America and the Caribbean in 2010

Figure 5. Land-cover map of Latin America and the Caribbean in 2050 under the MF scenario.

Figure 6. Land-cover map of Latin America and the Caribbean in 2050 under the FW scenario.

Figure 7. Land-cover map of Latin America and the Caribbean in 2050 under the PR scenario.

Figure 8. Land-cover map of Latin America and the Caribbean in 2050 under the GT scenario.

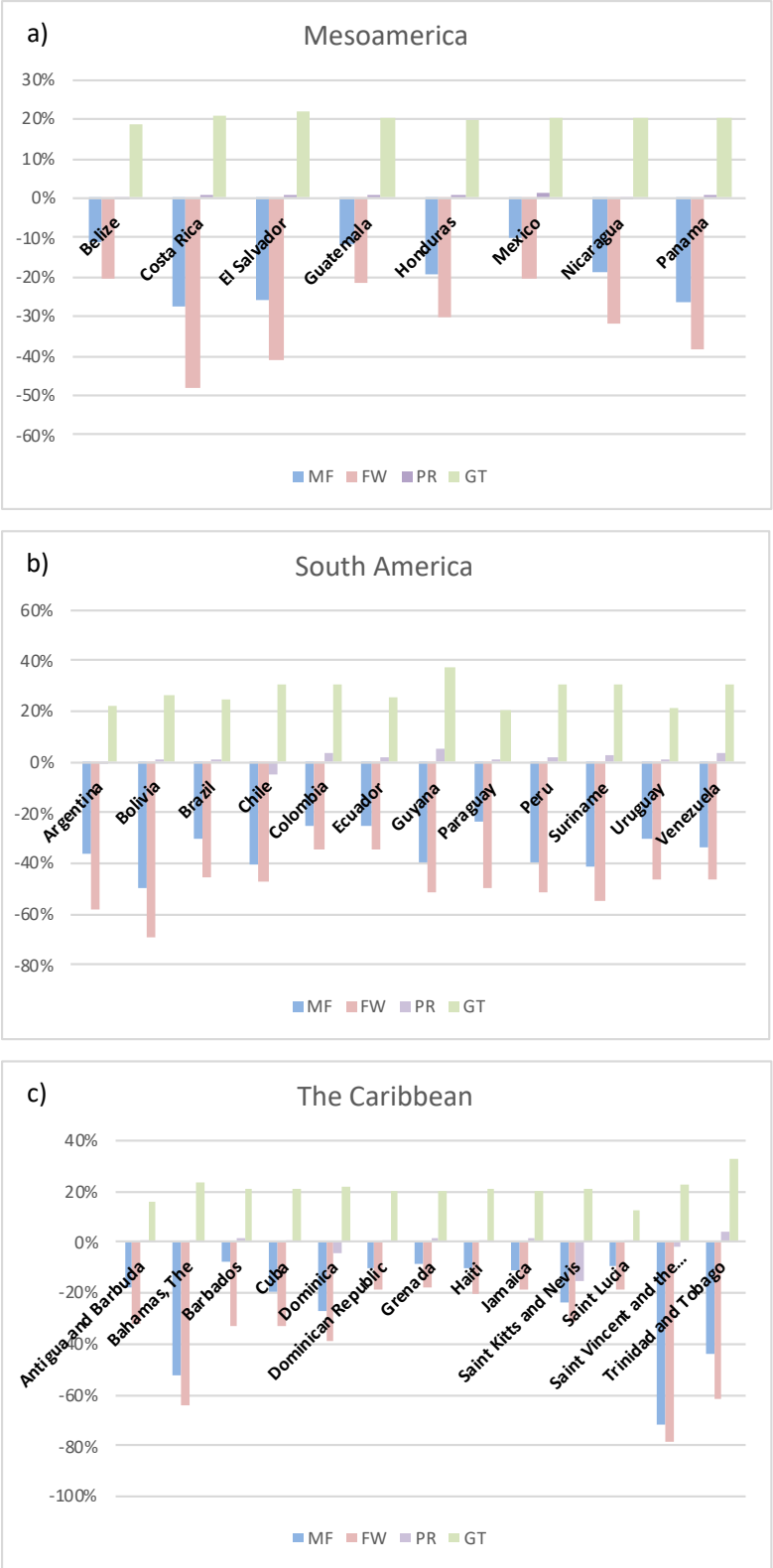

Figure 9. Percentage of change of the total ecosystem services value in each scenario for Mesoamerica (a), South America (b) and the Caribbean (c).

These results should be interpreted considering the following limitations and caveats. First, as in any other study on scenario analysis, scenarios are a simplification of plausible complex futures, and therefore, they are not predictions. Second, the value of ecosystem services for each biome is assumed to be constant over space, it is the potential supply of services from an ecosystem, but it is not related to the location and intensity of the demand from beneficiaries. This can be addressed in future studies by assessing specific services from which their beneficiaries are known or modelled. Finally, in our analysis and results of the change of land-cover and its ESV, some could argue the role that scarcity plays on their value, meaning that a loss of ecosystem services could raise their value since they would be scarcer, as in the case with marketed goods. Nevertheless, the majority of ecosystem services that we evaluated are non-rival, non-excludable, and non-marketed public or common property goods and services, which means that their unit values may not be affected significantly by relative scarcity from reduced area as much as by population demand. We assumed here that changes in supply are the major factor and the unit values will change mainly as a function of management policies and ecosystem health and condition that these imply (Kubiszewski et al., 2017).

Discussion

As described before, LAC is one of the most biodiverse regions of the world, but also struggles with high rates of poverty (Wodon & Ayres, 2000) and other social challenges. This makes both the environment and the communities that depend on it for their livelihoods very vulnerable to changes as the plausible ones calculated in this study, under different development scenarios. Moreover, it seems that in the current development path, the region is following similarly to MF, with a population and economic activity growing but at the expense of significant social and environmental impacts.

This tendency is shown in reports such as the GEO-6 Regional Assessment for Latin America and the Caribbean, which found that the region has a strong reliance on primary products and natural resources, both accounting for 50% of all good exports, and in the case of

South America this is even more prominent due to extra-regional demands for agricultural (e.g. coffee, soybean and meat) and mineral (e.g. ores and metals) resources. Furthermore, international tourism receipts in the Caribbean were 45% of total exports, more than twice the amount earned by Mesoamerica, and 9 times greater than South America. The report concludes that although the rate of conversion of natural systems has begun to slow, the overall rate of loss of ecosystems remains high (UNEP, 2016).

Reaching a development as the one described in the GT scenario will require an integral and perhaps escalated approach, finding solutions for the most urgent social problems (e.g. extreme poverty and inequality) that constitutes the bases to address the economic challenges, both being the pillars towards environmental sustainability. The GT scenario, and to some extent the PR scenario, take into consideration many of the 17 Sustainable Development Goals agreed by all the UN member states in 2015 as part of the 2030 Agenda for Sustainable Development (United Nations, 2015).

The key challenge on maintaining and enhancing ecosystem services is the development of strategies that reduce the negative environmental impacts of land use across multiple services and scales while maintaining social and economic benefits, balancing short-term and long-term needs (Foley et al., 2005), at the same time that "tipping points" are considered (Galaz, 2014). A well-known example that this is possible is the case of Costa Rica, a country with no army, with 5% of the planet's biodiversity, electricity produced by more than 95% renewable energy, and that has not only stopped deforestation rates, but it has also reverted it, having more forest every year at the same time its GDP is growing, with incomes per capita that are double than what they used to be three decades ago, and providing universal access to health care and education (Stiglitz, 2018).

Bold development decisions in Costa Rica, such as eliminating the army, or switching to an economy based more on nature conservation rather than the former economic strategy based on agriculture, required decision makers to imagine a future that, a few decades ago, seemed not plausible or too different from the development path other countries were following. Nevertheless, current policy in the great majority of countries around the world, is based on the

wrong assumption that the future will be similar to the present, making policies obsolete and unadaptable to unforeseeable surprises.

Scenario planning exercises on natural capital as the one presented here, help decision makers to develop policies under a shared goal of environmental sustainability, making evident the intrinsic relation between economic development and nature conservation, enhancement and restoration. Conducting this type of studies will provide policy makers a clear picture of plausible changes in economic benefits from healthy ecosystems according to different development paths, as well as to identify which land covers could be protected or restored in order to get the highest economic gains (which should be done in combination with ecological and biophysical assessments), representing a unique opportunity to produce cost-benefit analysis in the present and future.

Conclusion

This study is the first of its kind for Latin America and the Caribbean, providing values of the ecosystem services for all 33 countries of the region for the present and the future. Our estimates show how different management options, and development priorities, can have a significant impact on land-cover and its ecosystem services. The ecosystem service value of Latin America and the Caribbean range from USD $19 trillion under the GT scenario to USD $8 trillion under the FW, a difference equivalent to a 145% of the region's GDP in 2011.

These results are a first approximation to the present and future value of natural capital. Further research on this should take into consideration key factors such as the non-linear behavior of drivers of change and its associated tipping points, the participation of different sectors of society at local scale that can provide new visions of plausible futures that were not taken in consideration at the global and regional scale, and the role of communication using novel approaches such as visual arts and science fiction narratives in order to engage a wider public.

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The economic value of ecosystem services of 7 Ramsar Sites in Costa Rica

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Abstract

Ramsar Sites provide a wide range of ecosystem services, from food, raw materials, recreation, carbon sequestration, water purification and medicines, among many others. Nevertheless, these benefits are often undervalued or not valued at all, causing a rapid decline and degradation of these ecosystems. We calculated the economic value of ecosystem services from seven Ramsar Sites in Costa Rica. We used value transfer to calculate the economic value of those services that are not in the market, and direct market valuation for those that are in the market (i.e. agriculture). Our results show that the total economic value of ecosystem services from these Ramsar Sites is $3.2 billion/year for 2015, which represents 6% of the country's GDP.

Key words: wetlands; Ramsar Sites; economic value; ecosystem services; natural capital

Introduction

The Ramsar Convention, also called the Wetlands Convention, is the intergovernmental treaty that provides the framework for the conservation and wise use of wetlands and their resources. It was adopted in 1971, and it has been signed for approximately 90% of the United Nations member states.

The Convention defines wetlands as "areas of marsh, fen, peatland or water, whether natural or artificial, permanent or temporary, with water that is static or flowing, fresh, brackish or salt, including areas of marine water the depth of which at low tide does not exceed six meters" (Ramsar, 1971). Contracting Parties of the Convention are committed to identify and designate their most significant wetlands as Ramsar Sites, which need to meet a list of criteria such as to contain representative, rare or unique wetlands types.

Costa Rica is part of the Convention since 1991, and currently have twelve Ramsar Sites (also known as Protected Wetlands of International Importance) distributed all over the country: National Wildlife Mixed Refuge Caño Negro, Palo Verde, Caribe Noreste, Gandoca-Manzanillo, Cocos Island National Park, Potrero Grande Mangrove, Respingue Lagoon, Las Baulas, Talamanca Peatlands, Arenal Reservoir and National Wildlife Mixed Refuge Maquenque.

This study estimated the value of the following seven Ramsar Sites (Figure 1):

1. *Palo Verde*. It is a complex of permanent and seasonal wetlands, formed by a group of swamps, marshes, lagoons, rivers and streams from the basin of the Tempisque river. These wetlands play an important role in the conservation of several ecosystems, is one of the few areas where there are remnants of Tropical Dry Forest in Costa Rica. This wetland is one of the most important in Central America for nesting, resting and wintering for more than 60 species of migratory aquatic birds, as well as a habitat for resident endangered species. Neighboring communities depend on the water that is obtained in some of the wetlands for their activities such as livestock.
2. *National Wildlife Mixed Refuge Caño Negro*. It is the largest natural lagoon in the north of Costa Rica, surrounded by sectors that are flooded annually by the overflow of the rivers Frío, Mónico y Caño Negro, generating swamps and flooded forest of yolillo palms. This wetland is part of a series of wetlands between Nicaragua and Costa Rica with a high importance for animals such as migratory birds; and it is also the habitat of the tropical gar (*Atractosteus tropicus*), considered a living fossil and at extinction risk. Caño Negro is vital in maintaining the environmental quality of the north of the country,

mainly because its role in regulating floods. Additionally, the Caño Negro lagoon is ancient site for the Maleku indigenous community.

3. *Las Baulas*. Covered 80% by mangrove forests, it has five species of Pacific mangroves with individuals of 25-30 meters high and 45cm of diameter. This wetland is classified as marine, estuarine and palustrine. The coastal area is constituted by Playa Grande, one of the most important nesting sites for the leatherback turtle (*Dermochelys coriacea*). Tourism is the principal economic activity in Las Baulas, with recreation activities such as aquatic sports, boat tours, birdwatching, hiking and the spawning of leatherback turtles.

4. *Terraba-Sierpe*. Considered one of the largest wetlands in Central America, it is made up of five types of wetlands: swamp, fluvial, lacustrine, estuarine and coastal marine. The vegetation of the mangrove forest of Terraba-Sierpe plays a fundamental role in flood regulation, coastal erosion and storm protection. It is the habitat of numerous species of birds, fishes, mammals, mollusks and reptiles. The main economic activity in this wetland is the extraction of piangua (a type of mollusk), fishing, intensive agriculture, and more recently tourism.

5. *Gandoca-Manzanillo*. It includes in its marine part, shallow water areas, areas of underwater vegetation and coral reefs; the coasts are variable with areas of rocky cliffs, areas of sandy beaches of reef origin and mineral material. In the terrestrial part, there are red mangroves (*Rhizophora mangle*), adjacent swamps, flooded forests such as "yolillales" (yolillo palms), tidal creeks and the Gandoca Lagoon. This wetland is habitat of many resident and migratory birds, it has the only coastal lagoon in the south Caribbean, which is a vital ecosystem for the survival of the manatee (*Trichechus manatus*), and its beaches are a nesting site for marine turtles. The Gandoca Lagoon is an important ecosystem for fish larvae breeding, and the Punta Mona swamp works as a water reservoir for the communities in the south-east of the country. The main activities in this wetland are tourism, banana production, subsistence crops, basic grains and artisanal fishing.

6. *Caribe Noreste*. This wetland contains marine-coastal ecosystems, including shallow waters on the coast, long sandy beaches, river mouths and lagoons; in the terrestrial part, it has flooded ecosystems such as flooded forests, herbaceous swamps, wooded swamps and swamps dominated by palms, as well as an extensive system of channels, rivers and lagoons that feed and connect to the wetland. Caribe Noreste is one of the main migratory routes for a wide variety of birds, a reproduction and feeding ground for the main species of fish for subsistence, it is one of the key areas for reproduction and feeding of the manatee (*Trichechus manatus*), and its beaches are a nesting site for four species of marine turtles. The main economic activities on this Ramsar Site are agriculture, cattle and tourism.

7. *National Wildlife Mixed Refuge Maquenque*. It is formed by a series of lagoon complexes that are representative from the north zone of Costa Rica, mainly wetlands that are part of a very humid tropical forest and of a hydrologic system that is characterized by three large rivers (San Juan, San Carlos y Sarapiquí). Maquenque also contains swamps and flooded forest of yolillo palms. These wetlands influence the regional climate regulation, determining a unique habitat for many species such as the jaguar (*Panthera onca*), manatee (*Trichechus manatus*), tropical gar (*Atractosteus tropicus*), mountain almond tree (*Dipteryx panamensis*) and the green macaw (*Ara ambiguus*). The rivers and lagoons are used for transport, tourism, recreation and fishing. Other economic activities include agriculture and cattle (Proyecto Humedales, 2015).

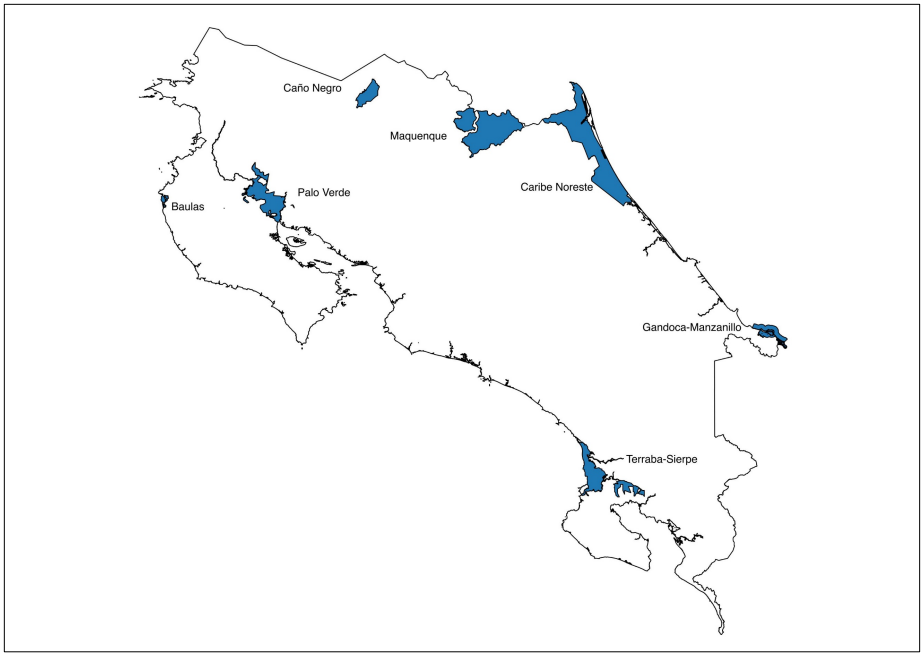

Figure 1. Ramsar Sites evaluated in this study.

Wetlands provide many ecosystem services to society, such as food, fresh water, genetic materials, climate regulation, pollination, erosion regulation, education, recreation and soil formation, among others (Millennium Ecosystem Assessment, 2005). These services are often underpinned by a combination of ecosystem functions that originates inside and outside the boundaries of the wetlands, such as in the case of the hydrology of wetlands which is determined by ecological and physical features of the wetland itself and that of its catchment where it is located (Russi et al. 2013).

Ecosystem services from wetlands benefit many different stakeholders that can lead to conflicting interests and/or over-exploitation of some services (often provisioning services) at the expense of others (regulating services and cultural services), which makes the recognition of the value of wetlands essential for a more balanced decision-making and therefore, a more sustainable use (De Groot, Stuip, Finlayson & Davidson 2006).

The value of wetlands can be expressed in different units, including biophysical units such as the carbon dioxide sequestered, the kilograms of pasture that it provides for cattle ranching, or the number of species of commercial interest that depend on them. Nevertheless, this value is often expressed in economic units (which are related to biophysical units) such as dollars per

hectare or dollars per unit of production of the service, in order to be able to compare it with other activities and products from the market, especially for cost-benefit analysis for decision makers. Economic valuation of wetlands aims to quantify the benefit (marketed and non-marketed) that people obtain from these ecosystems, which allows to compare them with other sectors of the economy when investments are assessed, policies are designed, activities are planned, or decisions regarding land and water are made (Millennium Ecosystem Assessment, 2005). Therefore, the goal of valuation under this framework, is the allocation of wetland resources to improve human well-being (Barbier, Acreman, & Knowler, 1997). Globally, the value of ecosystem services from wetlands has been estimated in $14 trillion annually (De Groot et al., 2006).

Ramsar wetlands have been valued economically in several locations worldwide. Sharma et al. estimated in $16 million/year the value of the first Ramsar Site in Nepal, the Koshi Tappu Wildlife Reserve (Sharma, Rasul, & Chettri, 2015). Uddin et al. valued the provisioning and cultural ecosystem services of the Sundarbans Reserve Forest in Bangladesh, the world largest mangroves covering 6000km^2, at $744,000 and $42,000 per year (Uddin, van Steveninck, Stuip, & Shah, 2013). Also in Bangladesh, Sun et al. conducted a biophysical assessment in the Tanguar Haor wetland, in which they calculated the change in provisional services (rice production and food supply) and biodiversity services (mainly birds populations using the Shannon-Wiener Index) due to anthropogenic impacts such as land use, population growth and economic development (Sun, Zhen, & Miah, 2017). Other studies have focused on the impact of designating a wetland as a Ramsar Site. Barau et al, for example, examine the ecological and social-economical trade-offs in terms of ecosystem services in the Pulau Kukup Ramsar Site in Malaysia (Barau & Stringer, 2015).

In this study, we conduct a first approximation of the value of ecosystem services of seven of the twelve Ramsar Sites in Costa Rica, the first study of its kind in the country, with the goal of communicating to the Ministry of Environment and Energy and its Wetlands Project the importance of wetlands conservation not only because of its high biodiversity, but also because of the many beneficiaries that depend on them for their livelihoods and welfare.

Methods

Because we assessed Ramsar Sites, which includes several ecosystems, and not only wetlands *per se*, the first step was to determine the extension of each ecosystem in each Site. The Wetlands Project, from the Ministry of Environment and Energy of Costa Rica, was able to provide this information for each Site, with a total of 16 land use categories, 9 ecosystems (swamps, ocean, rivers, beaches, scrubland, yolillales, lagoons, forests and mangroves) and 7 human activities (rice paddies, banana plantations, sugar cane, grasslands for cattle production, aquaculture, pineapple fields and oil palm plantations) (Table 1 and Figure 2).

Table 1. Extension of ecosystems and human activities of each Ramsar Site.

	Palo Verde	Baulas	Caño Negro	Caribe Noreste	Gandoca-Manzanillo	Maquenque	Terraba-Sierpe
Swamps	12864	39	3081	67	0	231	1442
Ocean	0	694	0	0	6085	0	0
Rivers	2479	64	127	1848	46	95	742
Beaches	0	42	0	9	3	16	72
Scrubland	31	0	3	2128	0	0	747
Yolillal	0	0	922	23303	1246	259	2585
Lagoons	367	4	1226	1295	0	91	20
Forest	11947	93	3387	40016	2985	30828	1687
Mangroves	860	404	0	28	15	0	12222
Rice paddies	43	0	0	0	0	0	55
Banana plantations	0	0	0	6	0	0	0
Sugar cane	2	0	0	0	0	0	0
Grasslands	278	2	509	3449	123	4820	1103
Aquaculture	38	0	0	0	0	0	263
Pineapple fields	0	0	0	0	0	229	0
Oil palm plantation	0	0	0	0	0	0	13

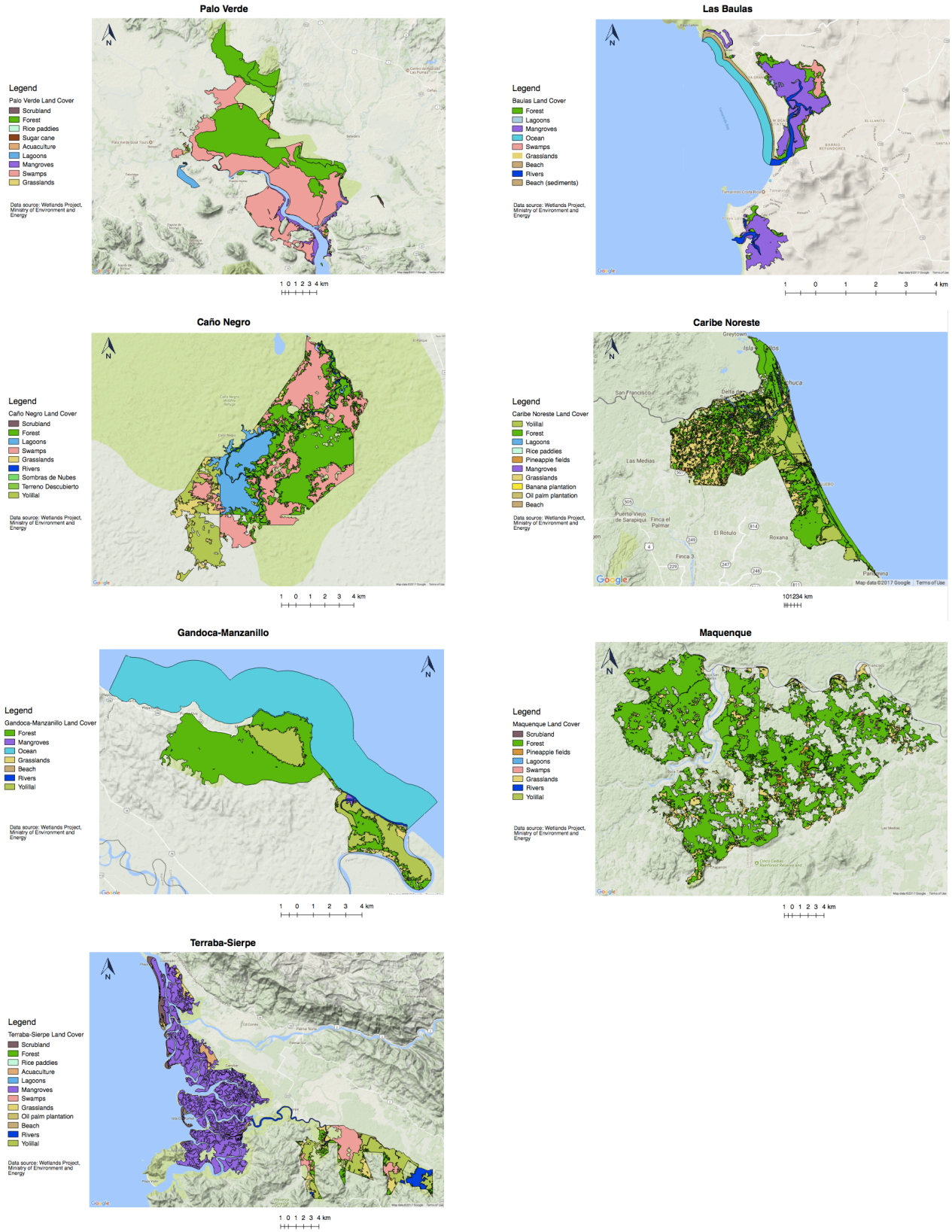

Figure 2. Maps of each Ramsar Site evaluated, showing the distribution of ecosystems and human activities.

Once the land cover for each Site is classified and measured (in hectares), the economic valuation process can start, with the end goal of determining the value (estimated in dollars/ha/year) of each ecosystem[2]. It is worth noting that ecological integrity or health of ecosystems (both related to natural variability and anthropogenic impacts) was not considered due to the extensive geographical scope of this study, but it could be considered in future studies (perhaps with a smaller area of study) in order to reflect how this can have an effect on the quantity and quality of ecosystem services provided to society.

The value of ecosystem services is the relative contribution of ecosystems to well-being (Turner et al. 2016), which can be expressed in different units (any units from the social, built, human or natural capital). For this study, the units are expressed in monetary terms (international dollars per hectare per year) since the goal is to communicate stakeholders (primarily the government of Costa Rica) the economic value of Ramsar Sites for a better management of these areas and with the future objective on developing financial mechanisms that could enhance its conservation (e.g. Payment for Ecosystem Services).

To calculate the economic value of ecosystem services (Total Economic Value, TEV) from the Ramsar Sites, we used unit benefit transfer, applying an average value taken from several study sites to the policy site, which is preferable than only transferring a single point estimate since an average value from several studies will produce a more accurate outcome (either because there are many suitable studies to take information from or because there are not suitable studies and therefore, partially cancelling out biases in individual studies) (Richardson, Loomis, Kroeger, & Casey, 2015). The application of this technology has been widely used in wetlands assessments (Brander et al., 2012) (Clarke, Harlow, Scott, & Phillips, 2015), (Crespin & Simonetti, 2016), (da Silva, Everard, & Shore, 2014), (De Groot et al., 2012), (Gandarillas, Jiang, & Irvine, 2016), (Ghermandi, Sheela, & Justus, 2016), (Ghermandi, 2017), (Kubiszewski, Costanza, Dorji, Thoennes, & Tshering, 2013).

In total, we took values from 304 estimates. Because we used estimates from different countries (hence different currencies) and from different years, we converted all estimates into 2015 U.S. international dollars per hectare per year, first by applying the Consumer Price Index

[2] Here we use ecosystem interchangeably with land cover

to express all values in 2015 values and then the Purchasing Power Parity index to convert values into international dollars by considering the cost of goods in the different countries from where we extracted the data. Once we did this, we were able to calculate the minimum, maximum and mean values of each ecosystem service from each ecosystem.

Results

Table 2 lists the ecosystem services valuated per ecosystem, as well as the number of estimates that we used for each ecosystem and the minimum, maximum, mean and median values per ecosystem service that we calculated.

Table 2. Value per hectare per year of each ecosystem service assessed.

	Number of estimates	Min	Max	Mean	Median
Swamps					
Provisioning services					
Food	7	4.25	559.29	128.40	59.34
Medicines	2	27.90	79.89	53.90	53.90
Raw materials	5	41.02	268.59	137.12	138.23
Drinking water	3	123.08	4415.27	2154.67	1925.64
Water sources	1			2568.79	
Total	18	196.25	5323.04	5042.88	2177.11
Regulation services					
Climate regulation	3	99.96	832.34	360.90	150.41
Erosion control	2	141.61	3901.82	2021.71	2021.71
Pollination	1			27.40	
Biological control	1			413.28	
Soil fertilization	2	49.96	413.28	231.62	141.61
Waste treatment	2	93.90	1127.88	610.89	141.61
Total	11	385.43	6275.31	3665.79	2455.33
Cultural services					

Inspiration	1			949.40	
Recreation/tourism	2	18.50	351.44	184.97	184.97
Total	3	18.50	351.44	1134.37	184.97
Ocean					
Provisioning service					
Food	1			209.91	
Genetic resources	1			60.76	
Total	2			270.67	
Regulation services					
Prevention of extreme events	1			5.69	
Total	1			5.69	
Cultural services					
Recreation/tourism	2	35.83	3093.48	1564.65	1564.65
Total	2	35.83	3093.48	1564.65	1564.65
Rivers					
Provisioning service					
Food	5	86.12	838.84	435.14	430.60
Medicines	4	30.81	11964.72	3150.08	302.38
Raw materials	7	1.46	1690.20	440.45	258.36
Genetic resources	7	0.72	3359.66	566.29	172.24
Drinking water	4	172.24	3950.71	1246.04	430.60
Total	27	291.35	21804.13	5837.99	1594.19
Regulation services					
Prevention of extreme events	1			72655.71	
Total	1			72655.71	
Cultural services					
Recreation/tourism	5	4.40	35792.80	7596.83	887.61
Total	5	4.40	35792.80	7596.83	887.61
Beaches					
Provisioning service					
Food	3	126.13	9203.41	3313.29	610.32

Raw materials	3	160.39	781.85	497.96	551.65
Genetic resources	4	101.86	22636.21	5817.18	265.33
Drinking water	3	76.16	535.34	254.30	151.39
Total	13	464.54	33156.81	9882.73	1578.68
Regulation services					
Climate regulation	2	124.74	2370.43	1247.59	1247.59
Water purification	1			15.72	
Prevention of extreme events	3	717.12	19421.72	9973.81	9782.59
Waste treatment	1			3357.44	
Total	7	841.87	21792.15	14594.56	11030.17
Cultural services					
Recreation/tourism	3	106.39	39926.01	20889.54	22636.21
Total	3	106.39	39926.01	20889.54	22636.21
Scrublands					
Regulating services					
Carbon sequestration	2	132.07	583.52	357.80	357.80
Erosion prevention	1			5.38	
Prevention of extreme events	1			22.28	
Total	4	132.07	583.52	385.46	357.80
Yolillal					
Provisioning service					
Raw materials	1			99.99	
Total	1			99.99	
Lagoons					
Provisioning service					
Food	1			353.83	
Drinking water	2	106.31	4588.31	2347.31	2347.31
Total	3	106.31	4588.31	2701.14	2347.31
Regulation services					
Water purification	1			598.02	
Total	1			598.02	

Cultural services					
Recreation/tourism	1			2330.64	
Total	1			2330.64	
Rainforest					
Provisioning service					
Energy	2	0.09	6985.68	3492.88	3492.88
Food	21	0.18	4596.36	659.27	98.59
Genetic resources	4	16.41	283.04	87.03	24.33
Medical/Bioprospecting	16	1.26	171528.38	12145.53	122.71
Timber	7	49.71	859.05	289.30	118.70
Fuelwood and charcoal	4	81.68	299.30	189.70	188.91
Fodder	2	185.48	289.91	237.69	237.69
Other raw materials	11	6.52	8678.68	1746.12	440.86
Water	2	9.72	23.93	16.83	16.83
Total	69	351.06	193544.33	18864.34	4741.51
Regulation services					
Water regulation/flows	3	2.47	53.05	34.30	47.37
Air quality	1			15.52	
Biological control	1			14.21	
Climate regulation	10	15.29	9889.32	1468.30	402.56
Erosion prevention	13	16.41	5621.72	1548.36	554.80
Protection against extreme events	5	10.58	710.89	160.62	12.37
Pollination	4	8.10	324.83	194.21	221.95
Water purification	4	0.89	1406.69	534.42	365.04
Soil detoxification	1			11.47	
Genetic pool (Biodiversity protection)	15	0.54	3119.20	277.66	22.26
Nursery	1			19.34	
Soil fertility	4	2.25	87.64	32.89	8.77
Total	62	56.52	21213.34	4311.28	1635.12
Cultural services					
Recreation/tourism	20	2.35	26378.08	1796.76	93.82

Total	20	2.35	26378.08	1796.76	93.82
Dry forest					
Provisioning service					
Food	2	0.09	0.82	0.46	0.46
Total	2	0.09	0.82	0.46	0.46
Regulation services					
Pollination	1			34.30	
Gene pool (biodiversity		9.05	19.66	14.35	14.35
protection)	2				
Total	3	9.05	19.66	48.65	14.35
Mangroves					
Provisioning service					
Food	11	1.04	22804.15	2763.22	189.25
Medical/Bioprospecting	3	10.25	733.58	258.22	30.81
Timber	5	52.26	789.12	259.29	136.61
Fuel wood	3	10.59	176.05	71.85	28.92
Other raw material	3	1.35	712.63	238.48	1.46
Total	25	75.50	25215.53	3591.06	387.05
Regulation services					
Climate regulation	3	11.24	2427.56	884.00	213.21
Erosion prevention	2	685.54	3121.73	1903.63	1903.63
Biodiversity protection	3	14.80	16724.78	5608.50	85.91
Nursery	3	152.36	1210.30	548.34	282.36
Protection against extreme		179.81	27637.69	7308.49	1804.91
events	6				
Total	17	1043.74	51122.06	16252.96	4290.02
Cultural services					
Recreation/tourism	3	52.21	944.29	353.70	64.60
Total	3	52.21	944.29	353.70	64.60

Ramsar Sites also cover non-natural land covers (i.e. agriculture), and therefore, it was needed to value these activities as well in order to have a total economic value of the Sites. To do this, we did a direct market valuation for each product based on income per hectare per year data published in the literature (Table 3).

Table 3. Values of agricultural products that are produced in some of the Ramsar Sites.

Product	Income per hectare per year[1]	Data source
Rice	$6,993.34 (minimum) $7,209.76 (maximum)	(CONARROZ, 2015)
Banana	$25.278 $40,504	(CORBANA, 2011)
	$22759	(Camacho, 2011)
Sugar cane	$3,782	(Kempkes, 1998)
Grasslands[2]	$2,936/ha	(Morales, Acuña, & Cruz, 2003)
Aquaculture (marine shrimp)	$14,680 (minimum) $29,361 (maximum)	(Valverde & Alfaro, 2014)
Pineapple	$23,105	(Carazo & Aravena, 2016)
Oil palm	$2,957	(Beggs & Moore, 2013)

[1] 2015 international dollars
[2] The value of grasslands is calculated based on the price of hay for cattle.

With the values of each ecosystem service per ecosystem, as well as the agricultural products, we multiplied them by the number of hectares of each ecosystem.

The total ecosystem service value per ecosystem varies considerably. On a per hectare basis, rivers ($86,090/ha/year) have the highest value, mainly because their very high value of prevention of extreme events ($72,655/ha/year), followed by beaches ($48,319/ha/year) in which recreation is the ecosystem service with a highest value ($20,889 ha/year) and rainforests ($24,972 ha/year) with the highest value for bioprospection ($12,145 ha/year) (Table 4).

It is important to state that values of ecosystem services were calculated depending on the information available, meaning that the total value of ecosystem services for an ecosystem

might be considerably different to another one due to the estimates used (i.e. if the total ecosystem services for an ecosystem is very low in comparison to another one, this might be because there was not many economic information available for this ecosystem, therefore less estimates to sum up).

Table 4. Mean value, on a per hectare per year basis, of each ecosystem

Ecosystem	Mean Value
Swamps	9,843.04
Ocean	1,841.02
Rivers	86,090.53
Beaches	48,319.03
Scrublands	385.46
Yolillal	99.99
Lagoons	5,629.80
Rainforest	24,972.38
Dry forest	49.11
Mangroves	20,197.72

Caribe Noreste is the Ramsar Site with the highest mean Total Economic Value (TEV), its extension is 55% forest, which provides 85% of its TEV. In second place is Palo Verde, where swamps are the ecosystem with the highest area (44%), but forests have the highest contribution to its TEV (45%) followed by rivers (32%) which have a small extension though (9%). In third place is Maquenque, covered mostly by forests (84%) which provides 96% of its TEV. Baulas is the Site with the lowest TEV and extension, mangroves are the ecosystem with the highest economic value, equivalent to 48% of its TEV (Table 5 and Figure 3).

Furthermore, if we estimate the TEV per hectare of each Ramsar Site, we can see a different picture. Palo Verde is the Site with the highest value ($22,823 per ha/year), followed by Maquenque ($21,884 per ha/year) and Terraba-Sierpe ($18,168 per ha/year).

Table 5. Total Economic Value of the seven Ramsar Sites

Ramsar Site	Area (ha)	Min TEV	Max TEV	Mean TEV	Median TEV
Palo Verde	28,909	$16,076,551	$3,247,759,991	$659,798,323	$151,200,094
Baulas	1,342	$638,355	$64,527,289	$19,711,438	$5,713,388
Caño Negro	9,255	$3,405,855	$866,485,899	$134,331,258	$39,951,894
Caribe Noreste	72,149	$17,591,194	$9,767,096,046	$1,180,794,418	$268,291,876
Gandoca-Manzanillo	10,503	$1,477,059	$742,748,594	$90,638,954	$29,145,765
Maquenque	36,569	$12,815,762	$7,442,391,860	$800,287,908	$201,038,773
Terraba-Sierpe	20,951	$20,542,013	$1,427,770,803	$380,654,569	$80,977,593
Total	179,678	$72,546,790	$23,558,780,483	$3,266,216,868	$776,319,383

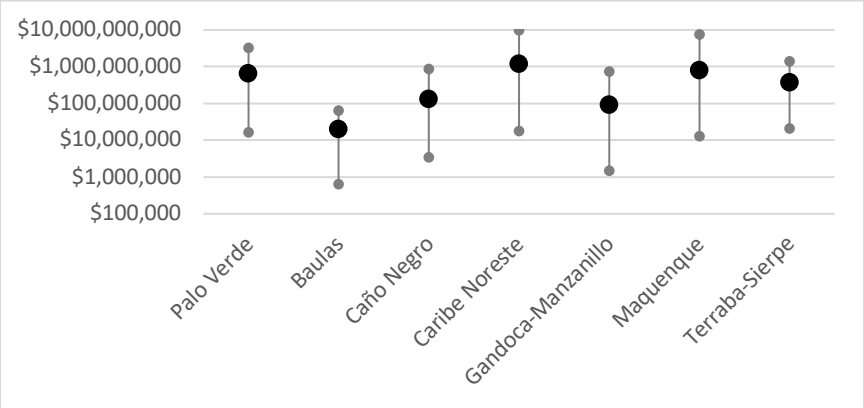

Figure 3. Range of total economic values per Ramsar Site

The mean TEV of all Ramsar Sites evaluated is $3.2 billions per year, which represents 6% of the country's GDP. These estimates are a first approximation, probably underestimates of the real value, but serves as a communication strategy for policy makers and other relevant stakeholders that are in charge of natural capital management. Our estimates should be used as a starting point for future studies to come, especially primary studies for each ecosystem service in each Ramsar Site.

Discussion

This study is the first economic valuation of Ramsar Sites at this level in Costa Rica (at least that we know of); we were able to show that healthy wetlands are essential for the well-being of its beneficiaries and the economy of Costa Rica. Unfortunately, often this value is not recognized leading to the degradation and conversion of these ecosystems to other land uses that the market can capture their value (e.g. urbanization, agriculture, tourist developments).

Recent policies in Costa Rica emphasize the importance of wetland valuation such as the one that we did. The government launched this year the National Wetlands Policy for 2017-2030, which recognizes the need to value the ecosystem services from wetlands, as well as the need to design and implement policies and programs that can promote wetlands conservation based on its value. Therefore, our study constitutes a direct input for the goals of the Wetlands Policy on economic valuation of natural capital for its sustainable management.

Going beyond unit value transfer

For achieving these results, we used the "simplest" form of transfer that can be applied, often due to budget and time constraints, as explained in the methodology section. One limitation from this methodology is that it assumes that the total extent of the ecosystem provides services, which is actually the potential supply of services, and therefore, demand is not taken into account, which should consider the location of the beneficiaries as well as the quantity of benefits they demand. To address this limitation, if budget and time constraints are still significant, and therefore, primary studies cannot be conducted, the next step would be to apply more complex transfer methodologies, such as a Value Function Transfer or Meta-Regression Analysis Function Transfer.

Value Function Transfer

If a higher level of accuracy is needed, or there is a need to enhance the quality of the transfer due to differences in the ecosystem services or the population between the study site and the policy site, then the value function is the next best type of transfer to use, in which a willingness to pay (WTP) function that was constructed in the study site can be applied to the policy site, with new parameters in order to determine the value of the ecosystem service.

Richardson et al explain the example of a WTP function in which the WTP depends on the quantity and/or quality of the ecosystem services, as well as from socioeconomic variables of the population that benefits from those services (Richardson et al., 2015). As it is always the case, the quality of the function transfer depends on the quality of the primary research in which the function is based (Johnston & Rosenberger, 2010).

In order to work well, according to Ready and Navrud, this type of transfer needs to comply with the following conditions: 1) sufficient variation at the study site in the attributes of the ecosystem service, 2) sufficient variation at the study site in the attributes of the beneficiaries/population, 3) the attributes of the ecosystem service and the population at the policy site are within the range of values estimated at the study site , and 4) preferences for the ecosystem service are similar in both sites (Ready & Navrud, 2005).

Nevertheless, as with the case of unit value transfer, the function transfer has some limitations or challenges such as the need of knowledge of the values of the independent variables for the policy site, and furthermore the assumption that the statistical relationship is the same between the dependent and independent variables at both sites (Richardson et al., 2015).

Meta-Regression Analysis Function Transfer

The next level of complexity, and therefore, a higher quality of the results, is the use of meta-regression analyses functions, used specially when existent studies that match the policy context are unavailable (Richardson et al., 2015).

Wilson and Hoehn defines this methodology as a “statistical technique for synthesizing the results of several existing non market valuation studies by estimating relationships between

control variables (methodology used, sample demographic characteristics, characteristics of the good) and monetary values estimated across multiple studies" (Wilson & Hoehn, 2006). A value function is generated through the combination of the results from numerous original valuation studies that contain both information from the good and the user population (Ready & Navrud, 2005). In other words, as Navrud explains, "instead of transferring the benefit function from one valuation study, results from several valuation studies can be combined in a meta-analysis to estimate one common benefit function" (Stale Navrud & Olvar Bergland, 2004).

Richardson et al. states that in general terms, the steps of the methodology for conducting a meta-regression analysis function are:

1. Collecting the ecosystem services valuation studies.
2. Coding those studies in terms of WTP per unit, characteristics of the study site, methodological aspects of the study (e.g. valuation methodology used) and demographics of the population.
3. Calculating the regression model using the WTP per unit as the dependent variable and the study site characteristics, the methodological aspects and socioeconomic variables as independent variables (Richardson et al., 2015).

New conservation strategies for wetlands

Our results are an important input for the development of new conservation efforts in the country. The Payment for Ecosystem Services program of Costa Rica has been one of its most successful conservation strategies, used in many countries as the base to develop their own financial mechanisms with the goal of forest protection with the help of the private sector. The program has been running for two decades now, and we think it is time to evolve towards a new scheme that is not only focused on forests, but also in other ecosystems such as wetlands. This new mechanism would have new challenges, mainly because wetlands are not only one ecosystem, and most of all because the majority of them are public property. Therefore, establishing this new scheme, in a form of a PES program or in other formats, such as a Common

Asset Trust (Farley, Costanza, Flomenhoft, & Kirk, 2015), will require the development of new legislation, institutional frameworks, and national and international financial mechanisms.

Our results can support the development of these financial mechanisms for wetlands conservation, by providing a list of ecosystem services that can yield the highest social-economic and environmental profit, helping this way to determine benefit/cost ratios of nature conservation (Balmford et al., 2002), which would help the decision makers establish new policies based on sound information. In the specific case of creating a PES scheme for wetlands, our results should not be used as a price, they should be used to inform buyers the benefits they are receiving from ecosystems, it is therefore an indicator of the social profit of investing in natural capital. Moreover, it is worth noting also that our results should not be used to set fines to polluters from environmental damages, since the error from results from value transfer can be significantly high.

Conclusion

Although this is an initial estimate of the ecosystem services from Ramsar Site using simple benefit transfer techniques, it is the first of its kind for Ramsar Sites in Costa Rica. We assessed not only wetlands but other ecosystems such as rain forests and dry forests, which allowed to estimate the value of a wide variety of ecosystems for Costa Rica that can be used in future studies of ecosystem services valuation. The high economic values of the benefits that these ecosystems provide to society demonstrates the need for their conservation not only as part of the environmental agenda, but also as part of the social and economic one.

This study can be used to develop financial mechanisms for wetlands and other ecosystems conservation and restoration, such as new schemes of Payment for Ecosystem Services in Costa Rica, which right now are only focused on forests. The use of the value that we estimated here can be used to estimate the social benefit of PES schemes by expressing how much we are paying to land owners versus how much the ecosystem is providing to them and to society as a whole, both nationally and internationally. Nevertheless, the majority of the

ecosystem services assessed are public goods, and therefore, a new type of PES scheme or other financial mechanisms need to be developed in order to incorporate common/public property rights.

Having these estimates as a baseline, the next step is to conduct primary studies in Costa Rica for Ramsar Sites, taking into consideration the real beneficiaries in each site instead of the potential ones, as well as calculating values based on biophysical data from the specific site. In order to conduct this research, more efforts on multidisciplinary research need to be done, in all phases, but specially in research design, allowing the flow of information needed through the many and different steps that a primary economic valuation encompass.

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Economic valuation of the ecosystem services provided by the mangroves of the Gulf of Nicoya using a hybrid methodology

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Abstract

Mangrove forests have been the most studied estuarine ecosystem in the Gulf of Nicoya, its total cover here is 20,739 ha. Because of the public good nature of many of the mangrove's ecosystem services, markets for them do not exist and there is limited potential to manage them with conventional markets. And because of the difficulties in estimating the value of these non-marketed services, mangroves are often undervalued in cost-benefit analysis of conservation versus commercial land uses, causing their degradation and loss. We applied a hybrid approach to estimate the value of ecosystem services from mangrove forests in the Gulf of Nicoya, using traditional benefit transfer and expert modified benefit transfer for 11 ecosystem services, and primary studies for 3 ecosystem services (fisheries, climate regulation and coastal protection) including the use of modeling with INVEST in combination with benefit transfer. Using traditional benefit transfer we estimated the economic value of 11 ecosystem services of these mangroves in $812 million per year (median=$88 million/year), and the total mean value of the ecosystem services provided by the total extent of mangroves in Costa Rica as $1.5 billion per year (median=$160 million/year). By applying the expert modified benefit transfer, we estimated that the mean total value of the mangrove forests of the Gulf of Nicoya is $470 million per year, and a median value of $75 million per year. Combining the values of the expert modified benefit transfer with the estimates from the primary studies, we calculated the mean total value of the ecosystem services assessed from mangrove forests in the Gulf of Nicoya in $408 million per year, and a median total value of $86 million. Considering the median total value of ecosystem services from mangroves, it represents 0.16% of the GDP in Costa Rica in 2015.

Key words: mangroves; economic value; ecosystem services; natural capital

Introduction

The Gulf of Nicoya is located in the north-western part of the Pacific coast of Costa Rica. It is one of the most important ecological and geographical systems in the country. The Tempisque, Barranca and Grande de Tárcoles rivers drain into it creating a highly productive estuary (Kappelle, 2016). It represents one of the largest estuaries in Central America, with a surface area of 1,530 km^2 (Fernández, Alvarado, & Nielsen, 2006).

The shape of the Gulf and its bathymetry determines three main regions: (1) the internal and shallow region, which extends from a line between the tip of the Puntarenas peninsula and San Lucas Island to the mouth of Tempisque River; (2) the middle region, with limits to the south with a line from Negritos Islands at the west coast to Puerto Caldera at the east coast; and (3) the lower region, from Islas Negritos to the south at a line that crosses the gulf from Ballena Bay to Herradura Bay (Rivera, 2018). The inner part of the gulf is bordered mostly by mangrove forests, tidal flats and sandy beaches (Kappelle, 2016).

Mangrove forests in the Gulf of Nicoya are favored by the interaction between the fresh water runoff from the rivers that flow into the gulf and salt water incoming from the sea. The surge and the high quantity of sediments and nutrients that are deposited in this area by the inflowing rivers and tides support mangrove productivity. The main species of mangroves in the gulf are *Rhizophora mangle*, *Rhizophora racemosa*, *Avicennia germinans*, *Avicennia bicolor*, and *Laguncularia racemosa* (Proyecto Golfos, 2012a). Mangrove forests have been the most studied estuarine ecosystem in the Gulf of Nicoya (Kappelle, 2016). The total mangrove cover in the Gulf is 20,739 ha (Rivera, 2018), which is under the management of three Conservation Areas from SINAC: Arenal Tempisque Conservation Area (ACAT), Central Pacific Conservation Area (ACOPAC), and Tempisque Conservation Area (ACT) (Figure 1).

One of the most recent estimates of the total area of mangroves in Costa Rica in 2013 was 36,250 ha (Programa REDD/CCAD-GIZ - SINAC, 2015). FAO had previously estimated

mangrove extent in the country in 2000 at 41,840 ha and in 1980 at 63,400 ha (FAO, 2007a). Although different techniques have been applied to calculate the total mangrove cover in the country, based on this data, Costa Rica has lost approximately 27,150 ha or 43% of the original mangrove area between 1980 and 2013; an annual loss rate of 1.3%.

Mangroves in Costa Rica, as in other parts of the world, have been impacted significantly by human activities such as the extraction of forest products like tannic acid (extracted from the bark of *Rhizophora*), charcoal, and construction materials, among others. Between 1960 and 1980, mangroves in some areas (especially in the Gulf of Nicoya) were drained and turned into rice fields, salt ponds, agricultural fields and shrimp ponds (FAO, 2005c). On the Pacific coast, mangroves have also been used as dump sites or have been filled (Kappelle, 2016). However, mangroves in Costa Rica are now mostly protected and their loss has slowed (Zamora-Trejos & Cortés, 2009).

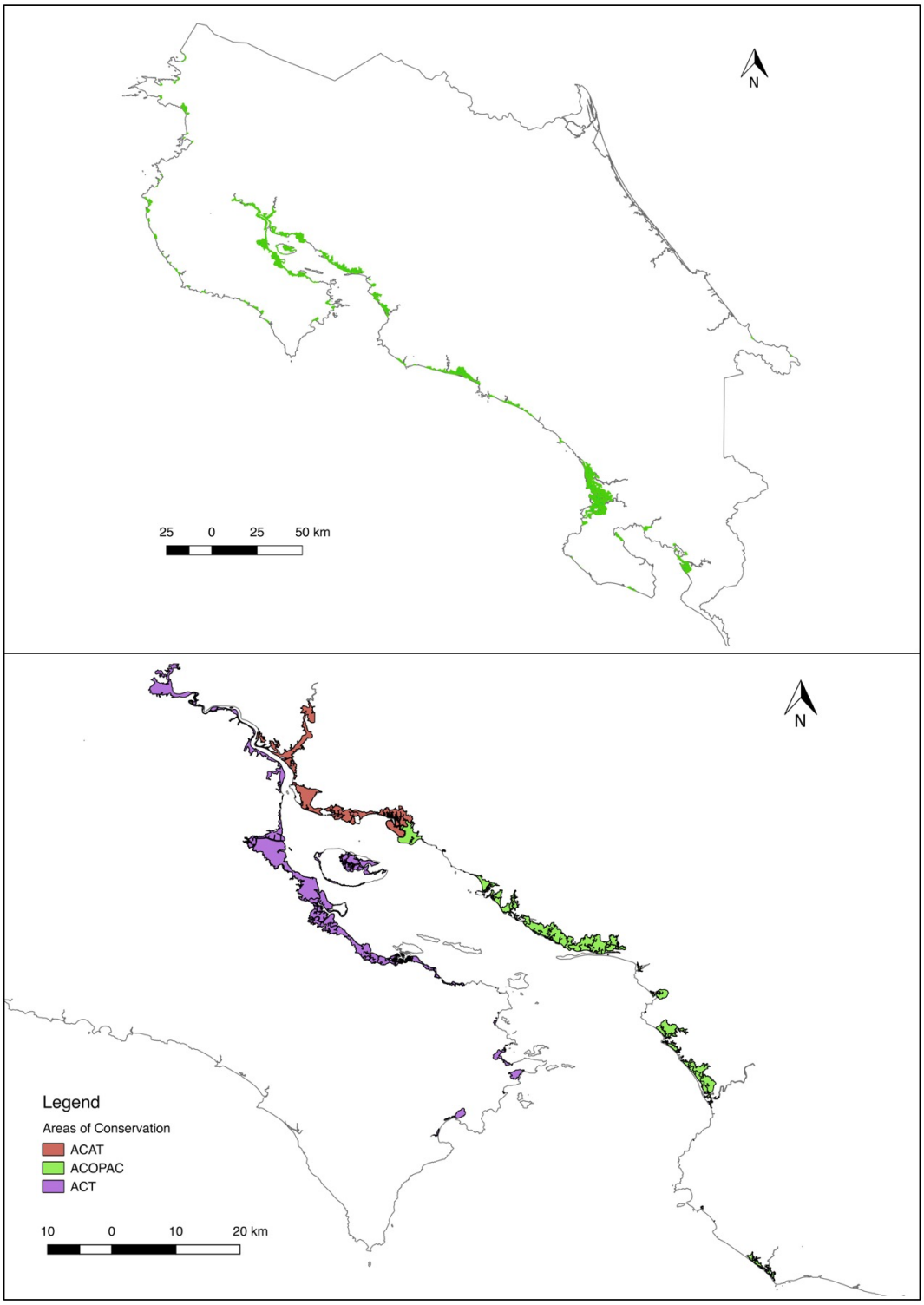

Figure 1. Mangrove cover of Costa Rica in 2013 (above) and the mangrove cover in the Gulf of Nicoya, indicating the Areas of Conservation from SINAC (below)
Source: Programa REDD/CCAD-GIZ - SINAC, 2015; Rivera, 2018

Economic value of ecosystem services from mangroves

Ecosystem services are defined as "the benefits people derive from functioning ecosystems, the ecological characteristics, functions, or processes that directly or indirectly contribute to human well-being" (Costanza et al., 1997; Costanza et al., 2011; Millennium Ecosystem Assessment, 2005a). Mangroves are known for providing many ecosystem services such as food, raw materials, climate regulation, pollution control, coastal protection, recreational opportunities and spiritual experiences, among many others (Millennium Ecosystem Assessment, 2005b; Russi et al., 2013).

The Millennium Ecosystem Assessment determined that the most frequent or significant services provided by mangroves forests are food, such as the production of fish and invertebrates; raw materials, such as timber and fuel; biological regulation, such as the interactions between different trophic levels; pollution control and detoxification, mainly though retention, recovery and removal of excess nutrients and pollutants; retention of soils, and storm protection (Millennium Ecosystem Assessment, 2005b). We summarize in Table 1 the list of ecosystem services that mangrove forests provide according to different authors.

Many of these ecosystem services have the characteristics of "public goods" (Brander et al., 2012). A public good exists when goods (or services) are non-rival (one individual may benefit from the existence of an environmental attribute and this does not reduce the benefit another individual can receive from that same attribute) and non-excludable (it is difficult or impossible to exclude individuals from benefiting). This is in contrast to private goods, which are both rival and excludable (Barbier, Acreman, & Knowler, 1997, Costanza, 2008).

Markets work best with private, (rival and excludable) goods and services. Because of the public good nature of many of the mangrove's ecosystem services (especially regulating and cultural services), markets for them do not exist and there is limited potential to manage them with conventional markets (Brander et al., 2012). And because of the difficulties in estimating the value of these non-marketed services, mangroves are often undervalued in cost-benefit analysis of conservation versus commercial land uses (Salem & Mercer, 2012, Acharya, 2002), causing their degradation and loss globally as previously mentioned. Therefore, valuing the multiple

ecosystem services that mangroves provide to society is necessary for a sustainable management and protection of these ecosystems. Also, a combination of market and non-market-based institutions is necessary for their effective and sustainable management.

Table 1. List of ecosystem services provided by mangroves according to several studies.

	MEA, 2005	Russi et al, 2013 (TEEB)	Quoc Tuan Vo et al, 2012	Salem & Mercer, 2012	Mehvar et al, 2018	Moberg & Ronnback, 2003	Mukherjee et al, 2014	Barbier et al, 2011	Spalding, 2010
Provisioning	Food	Food	Food	Commercial fishing and hunting	Fisheries	Seafood, honey, sugar, fruits, alcohol, vinegar	Fisheries (food), fisheries (aquaculture), honey	Food	Fisheries
	Fibber, timber, fuel	Raw materials	Wood products (timber, fibber, fuel)	Harvesting of natural materials	Raw materials	Tannins, lime, timber, thatch, firewood, fur, animal fodder	Wood, timber, fodder	Raw materials	Timer/fuelwood
	X	Ornamental resources	X	X	X	X	X	X	X
	Biochemical products	Medicinal resources	Medicines	X	X	Traditional medicine	Pharmaceuticals	X	Medicines
	Genetic materials	Genetic resources	Genetic materials	X	X	Genetic resources	X	X	X
	X	(Fresh) water supply	X	Improved water quality	Water filtration	Water catchment and groundwater recharge	X	X	X
	X	X	X	X	X	Aquarium industry products	X	X	X
	X	X	X	X	X	Sustaining the livelihood of coastal communities	X	X	X

	X	X	X	X	X	Habitat for indigenous people	X	X	X
	X	X	X	Energy resources	X	X	Energy resources	X	X
Regulating	Climate regulation	Climate regulation	Carbon storage	Reduced global warming	Carbon sequestration	Carbon dioxide sink	Carbon sequestration	Carbon sequestration	Carbon sequestration
	X	X	Microclimate regulation	X	X	X	X	X	X
	Biological regulation	Biological control	X	X	X	X	X	X	X
	Pollution control and detoxification	Waste treatment/ water purification	Air pollution reduction	Waste disposal	Contaminant storage and detoxification	Trap sediments and pollutants	Pollution abatement, environmental risk Indicator	Water purification	Biofiltration
	Erosion protection	Erosion prevention	X	X	Shoreline stabilization and erosion control	Erosion control	X	Erosion control	Erosion control
	Natural hazards	Moderation of extreme events	X	Storm protection	Storm protection and wave attenuation	Storm protection	Coastal protection	Coastal protection	Coastal protection
	X	X	X	Flood protection	Flood control	Flood protection	X	X	X
	X	Regulation of water flows	Watershed protection	X	Regulation of water flow	Interrupts fresh water discharge	Protection from salt intrusion and/or sedimentation	X	X
	X	Influence on air quality	X	X	Oxygen production	Oxygen production	X	X	X

	X	X	X	X	X	Nutrient filter	Water bioremediation		X
	X	Pollination	X	X	X	X	X	X	X
Cultural	Spiritual and inspirational	Inspiration for culture, art & design / Spiritual experience	Cultural uses	X	Artistic value	Cultural, spiritual and artistic values	X	X	X
	Recreational	Recreation/ tourism opportunities	Recreational uses	Recreation, tourism. Recreational fishing and hunting.	X	Support recreation	Ecotourism and recreation	Tourism, recreation	Recreation
	X	X	X	Appreciation of species existence	X	X	X	X	X
	Aesthetic	Aesthetic information	X	X	Aesthetic	X	Aesthetic value	X	X
	Educational	Cognitive information (education & science)	Educational uses	X	Educational opportunities	Educational and scientific information	X	Education, and research	X
	X	X	X	Existence, bequest, option values	X	X	X	X	Non-material values
Supporting	Biodiversity	Lifecycle maintenance (a.k.a. biodiversity)	Biodiversity	X	Nursery and habitat for fishes and other marine species	Nursery, feeding and breeding ground. Maintenance of biodiversity	Fisheries (nursery)	Maintenance of fisheries (nursing)	Biodiversity

	Soil formation	Maintenance of soil fertility	X	X	X	Top soil formation	X	X	X
	Nutrient cycling	Nutrient cycling	Nutrient cycling	X	X	Export of organic matter	X	X	X

In this context, the term “valuation” is understood as the relative contribution of a good or service to sustainable well-being (Costanza et al. 2014). The valuation of ecosystem services helps decision makers understand their value to society and the cost of their loss or the benefit of their conservation (Mukherjee et al., 2014 ; Himes-Cornell, Pendleton, & Atiyah, 2018; Acharya, 2002; Brander et al., 2012). The value of ecosystem services is therefore the relative contribution of natural capital in interaction with built, social, and human capital, to sustainable human well-being (Costanza et al., 2014, Turner et al., 2016).

Economic value is often defined in strict economic terms as aggregate willingness-to-pay (WTP) for the stream of services or to accept compensation for their loss. We think that such a definition is far too narrow when valuing ecosystems and their services. Since ecosystem services are the direct and indirect contributions to sustainable human well-being (which is more than the aggregate of individual, self-assessed welfare), it also depends on the welfare of the community or society, and on the sustainability of the ecological life support system (i.e., natural capital). Also, individual humans do not adequately perceive all the things that contribute to their well-being.

It is worth noting that conventional WTP-based valuation should be applied in addition to, and not as a replacement for broader social and ecological valuations (Costanza et al., 2017; De Groot, Stuip, Finlayson, & Davidson, 2006). Market prices or pseudo-market prices from surveys often exist for provisioning services (e.g. food, water, raw materials) and for a few cultural services (e.g. recreation/tourism), but do not exist for most regulating and supporting services.

Estimating the value of ecosystem services from mangroves can be used in at least 3 ways: (1) raising awareness on how the narrow economic paradigm dominates government decisions and the need for parallel ways of measuring natural resources as a complementary conservation strategy; (2) comparing commercial and conservation alternatives based on criteria such as net present value and cost-benefit ratios that give adequate weight to non-markets goods and services; and (3) calculating the amount that a commercial developer might need to pay as compensation for an environmental impact on mangroves (Lal, 2003).

According to Mehvar et al. (2018), the value of ecosystem services can be measured in three ways: 1) Total Economic Value (TEV), that refers to the value of a specific ecosystem service over the entire area covered by an ecosystem during a defined period; 2) average value of an ecosystem per unit of area or time; and (3) marginal value which is the additional value obtained or lost by an incremental change in a provision of a specific ecosystem service (Mehvar et al., 2018). TEV can be disaggregated into use values and non-use values. Use values are composed of direct use, indirect use and option values (Barbier et al., 1997). Direct use values are consumptive and non-consumptive uses that require a physical interaction with the ecosystem, such as outputs of fish, fuel and recreation. Indirect use values are regulatory ecological functions that lead to indirect benefits such as climate regulation, coastal protection and erosion control. Non-use values refer to existence and bequest values of mangroves (Salem & Mercer, 2012).

There are a range of different methodologies to estimate the value of ecosystem services from mangroves. The following table lists the most commonly used valuation methods for each of these ecosystem services:

Table 2. Methods for economic valuation of ecosystem services from mangroves.

Ecosystem services from mangroves	Commonly used valuation methods
Food	MP, P
Raw materials	MP, P
Medicines	AC, RC, P
Genetic materials	M, AC
Climate regulation	AC, R, PC
Biological regulation	AC, P
Pollution control and detoxification	RC, AC, CV, P
Coastal protection / Erosion protection	RC, AC, P
Recreation/tourism	TC, CV, H, P
Education	CV, benefits arise through education program expenditures
Biodiversity	CV, P
Option values	CV, P

MP=Market Price method, P=Production Systems Modeling, RC=Replacement Cost method, AC=Avoided Cost method, CV=Contingent Valuation method, TC=Travel Cost method, H=Hedonic method
Source: Turner et al., 2016; Salem & Mercer, 2012; Lal, 2003; Mehvar et al., 2018

The list of studies on economic valuation is extensive. Spalding provides a series of economic values for ecosystem services from global, regional and national studies (Spalding, 2010). The author argues that total economic values of mangroves ranges from $13,819/ha/year (UNEP/GPA, 2003) to $22,526/ha/year (Chong, 2006). Salem and Mercer (2012) also provide a good summary of valuation studies on mangroves, in which they cite mean values in US$ per hectare per year for fisheries (23,613), forestry (38,115), coastal protection (3,116), recreation and tourism (37,927), nutrient retention (44), carbon sequestration (967), non-use (17,373), biodiversity (52), water and air purification/waste assimilation (4,748) and traditional uses (114), based on 149 observations (Salem & Mercer, 2012). Other lists of value estimates can be found in Russi et al., (2013); Lal, (2003); De Groot et al., (2012); Vo, Künzer, Vo, Moder, & Oppelt, (2012); Mukherjee et al., (2014); Mehvar et al., (2018); and Barbier et al., (2011).

Himes-Cornell et al. (2018) provide the most recent review on mangroves valuation, which was made for 2007-2016. The authors found that most valuation studies are from Asia (53%) and Africa (14%), while Central and South America account only for 6%. Furthermore, the authors state that valuation studies often value only a small number of services, ranging from 1.8 services per study in North America to 4.9 in Africa. The services that are more commonly valued are food, raw materials, climate regulation, coastal protection, waste treatment, maintenance of life cycle of migratory species and opportunities for recreation and tourism (Himes-Cornell et al., 2018). Other authors agree with Himes-Cornell in that fisheries and coastal protection are among the most frequently valued ecosystem services (Mehvar et al., 2018), while studies on biodiversity are very scarce (Vegh, Jungwiwattanaporn, Pendleton, & Murray, 2014).

In Costa Rica there is only one study that we know about on mangroves at the Gulf of Nicoya (Arguedas-Marín, 2015), that estimated the value of the extraction of mollusks ($175 – 280/ha/year) and carbon sequestration ($15 – 38/ha/year) . Other similar studies include those on wetlands, specially from Ramsar Sites (Hernández-Blanco et al, 2017), and the Terraba-Sierpe wetland in the south Pacific (Barton, 1995; Earth Economics, 2010; and Sánchez et al., 2013).

This study is the first of its kind in Costa Rica, using a hybrid approach never applied in Costa Rica until now. Our method includes calculating the value of three ecosystem services of the Gulf of Nicoya using a detailed and innovative approach, based on primary and secondary

information, as well as ecological models. Furthermore, we expand upon that estimation and provide an economic value of the ecosystem services for the total cover of mangroves in the country, with the goal of showing decision makers the ecological and social-economic importance of protecting these endangered ecosystems.

Methods

We applied a hybrid approach to estimate the value of ecosystem services from mangrove forests in the Gulf of Nicoya, which means that we conducted both secondary and primary studies. Due to restrictions in time and budget, sometimes it is not possible to conduct primary studies to value ecosystem services (Wilson & Hoehn 2006; Plummer 2009) and, therefore, secondary data must be used.

Because of the limited biophysical-economical information available in Costa Rica required for natural capital studies on mangroves, and the limited time and resources available for this study, we used the benefit transfer method as one of our main approaches. This method consists of "applying economic value estimates from one location to a similar site in another location" (Plummer 2009). Applying a mean or median value taken from several study sites (origin of the primary studies) to the policy site (area been valued) is preferable than only transferring a single point estimate since a mean value from several studies will probably produce a more accurate outcome.

Following this approach, we extracted 67 estimates for 11 ecosystem services from mangroves in per hectare per year units from the Ecosystem Services Valuation Database (ESVD) from TEEB (Van der Ploeg & de Groot, 2010) (see References section in this paper for the list of studies from which these values where extracted). We chose studies that share similar characteristics to Costa Rica, especially location (i.e. tropical countries) and studies that provide values of flows instead of capital.

Because we used estimates from different countries (hence different currencies) and from different years, we converted all estimates into 2015 international dollars[3] per hectare per year, first by applying the Consumer Price Index to express all values in 2015 values and then the Purchasing Power Parity index to convert values into international dollars by considering the cost of goods in the different countries from where we extracted the data. We finally calculated the minimum, maximum, mean and median values of each ecosystem service. Standardizing values as described allow us to compare directly among studies and increases the strength of our inferences.

We also used these values to make a first approximation of the value of the ecosystem services provided by the total cover of mangroves in Costa Rica. In this case, we used the spatial data from the most recent national forest inventory developed by the government of Costa Rica, which estimated the national area of mangroves as 36,250 ha (Programa REDD/CCAD-GIZ - SINAC, 2015). These values were then multiplied by the per ha per year values of mangroves to obtain total value estimates.

To overcome some of the limitations of the benefit transfer method, in July 2018, we conducted a workshop with experts from the government, the academy and NGO's to determine which ecosystem services from our list of 11 are in reality provided by mangroves in the Gulf of Nicoya, as well as to define where these services are benefiting people. Having calculated the area of provision of ecosystem services according to the experts, we multiplied it by its relevant per ha per year value.

Once we estimated the value of ecosystem services using expert modified benefit transfer, we conducted a more in-depth analysis using primary data for specific services. We first selected which ecosystem services to value through a dual process of literature review and expert opinions. From Table 1, we selected those services from mangroves that have been considered as the most important by the report on wetlands from the Millennium Ecosystem Assessment, as well as those that are more commonly cited and valued in the literature. From this list we

[3] An international dollar would buy in the cited country a comparable amount of goods and services a U.S. dollar would buy in the United States. This term is often used in conjunction with Purchasing Power Parity (PPP) data.

selected the following ecosystem services: 1) fisheries, 2) climate regulation, and 3) coastal protection. We recognize this is a first approximation of the most important services in the area. However, we consider these three services to be the most relevant for the scope of this study since they encompass a broad range of services, most commonly evident in the Gulf of Nicoya and that are important for the local populations.

In April 2018, we interviewed governmental officials from each of SINAC's Conservation Areas working around the Gulf, to validate our list of ecosystem services and to rank their importance (i.e. low, medium, high) in each area. The experts selected for this survey are currently working on mangrove projects in each Conservation Area. Although they do not represent the complete statistical population of experts in the field, they work in this ecosystem and have relevant local experience and first-hand information. Appendix 1 provides the list of experts interviewed.

Each of these ecosystem services were valued according to the most appropriate methods for each one (Millennium Ecosystem Assessment, 2005b; Lal, 2003; Mehvar et al., 2018; Salem & Mercer, 2012; Himes-Cornell et al., 2018; Brander et al., 2012) (Table 3).

Table 3. Selected methods for economic valuation of ecosystem services from the mangroves of the Gulf of Nicoya.

Ecosystem service	Method
Food (fisheries)	Production/ha and market price
Coastal protection	Modelling and benefit transfer
Climate regulation	Social cost of carbon, marginal abatement cost

Climate regulation – Carbon stocks

We estimated the economic value of the total organic carbon storage in the mangrove forests of the Gulf of Nicoya using the Marginal Abatement Cost of Carbon (MAC) as the value of carbon stock per hectare. MAC are the costs of eliminating an additional unit of carbon emissions, and "these costs are the benefits forgone when scarce resources are used to avoid the chances of negative impacts of emissions instead of being used in alternative activities" (Jerath, 2012,

p35), in other words, MAC represents the opportunity costs. Specifically, we used the estimate that Fisher et al., (2007) produced for the 4th Assessment Report of the IPCC, with a mean MAC of $125/tC (calculated for the year 2010). This value was then converted to 2015 international dollars.

We applied the following equation to estimate the value of the carbon storage service:

$$V_{cs} = TC * MAC * A_m \quad (1)$$

Where V_{cs} is the value of the carbon storage service, TC is the total carbon stored per hectare, MAC is the marginal abatement cost of one tonne of carbon and A_m is the area of mangrove in hectares.

We obtained the total carbon stored at the ecosystem level (i.e. sum of carbon in all epigeous components plus carbon in the soil) per hectare from Cifuentes et al (2014), who estimated that TC in mangroves in the Gulf of Nicoya ranges between 413 and 1334 MgC/ha at 3 meters of depth (Cifuentes-Jara et al., 2014).

The simplest way of calculating the V_{cs} is using a mean TC for the entire area of analysis, but this can produce an imprecise result because of local variations in mangrove characteristics due to forest structure and stature. Therefore, we took the values of each research plot that Cifuentes et al. (2014) assessed in different locations through the Gulf and grouped them statistically and geographically to have a more precise estimate.

Climate regulation – Carbon sequestration

The methodology to estimate the value of carbon sequestration is different from the one for carbon stocks (Ramirez et al., 2002). Here, we used the Social Cost of Carbon (also referred as the Marginal Damage Cost). The SCC is defined as the net present value of the incremental damage on the environment and society due to the increase in carbon dioxide emissions. In other words, the SCC is the damage avoided by reducing emissions by one tonne (Tol, 2011).

For policy purposes, SCC is equal to the Pigouvian tax (i.e. tax on market activities that generates negative externalities) that could be placed on carbon (Tol, 2011), because SCC reflects, in theory, what a society should be willing to pay now to avoid the future damage caused by the increase of carbon emissions (Jerath, 2012).

We valued the carbon sequestration service applying the following equation:

$$V_{cseq} = SR * SCC * 3.67 * A_m \quad (2)$$

Where V_{cseq} is the value of the carbon sequestration service, SR is the sequestration rate in tonnes of CO_{2eq} per hectare per year, 3.67 is the conversion factor to obtain CO_{2eq} from C, A_m is the area of mangrove in hectares and SCC is the Social Cost of Carbon as estimated in the meta-analysis that Tol (2011) conducted with 311 published estimates. In this study, the mean estimate for SCC is $177/tC, and $80/tC (calculated for the year 2010) if only peer review papers are considered. We chose the peer reviewed values since they have a higher quality. This value was then converted to 2015 international dollars.

We applied a sequestration rate of 6 CO_{2eq}/ha/year for mangroves as reported in Murray, et al. (2010) and Maldonado & Zarate-Barrera (2015). This sequestration rate is also very similar to the 6.96 CO_{2eq}/ha/year value from Chmura et al. (2003) as cited in Sifleet et al. (2011) and is well within the conservative values for annual tropical forest growth rates (Cifuentes 2008).

Fisheries

We first conducted a literature review to identify the most important commercial species, (i.e. those that are fished the most and that have the highest prices). Because this is a study on ecosystem services from mangroves, we then selected those species that depend in some way on this ecosystem (e.g. for nursery, protection, food, etc.). To estimate the value of fisheries in this context, we should estimate the marginal production, or in other words, how much an additional hectare of mangrove represents an additional amount of fish caught.

Costanza et al. (1989) provide a method to estimate the marginal production by deriving catch respect the area of mangroves. In this method, catch in a year *t* has to be estimated first through a regression model that determines how much of the catch is actually related to the mangroves. Nevertheless, this catch model requires the fishing effort as one of the independent variables of the regression model, and unfortunately this data is not available in Costa Rica for any fish species, and consequently, it is not possible to apply this method. Therefore, we had to assume that the marginal and average products of mangrove area are equal for all species harvested. This could result in an overestimation, because the marginal product is generally lower than average product. However, there is also a compensating underestimation because the market price does not fully capture the value of fishing to society.

Data on catch and price for every species was obtained from the National Institute of Fisheries and Aquaculture (INCOPESCA). We started by processing data for the year 2015 from the statistics department of INCOPESCA, which is published on their website[4]. Nevertheless, this data is aggregated into "commercial categories", which can contain the same species but of different weight in two or more categories, and therefore, we needed to disaggregate the dataset by species with the help of the research department of INCOPESCA. Once we had the catch of each species in kilograms that can be attributed to mangroves, we multiplied it by the average price of each one to derive the value of the catch attributable to mangroves.

Coastal Protection

The coastal protection service of mangroves was determined through a combination of economic and biophysical techniques that together constitute a *benefit transfer method modified by modelling*. This technique consisted broadly of three stages: 1) determine an economic value of the ecosystem service of coastal protection per hectare of mangrove through the same process of benefit transfer explained above, 2) geographically model the variables that play a role in the provision of this service in order to identify which areas of the Gulf are more vulnerable and where and with what intensity the mangroves provide the service, and 3) multiply the per hectare value obtained previously by the classified geographic areas according to its service provision.

[4] https://www.incopesca.go.cr/publicaciones/estadiscticas/historico/2015.html

We used the Coastal Vulnerability model of INVEST (Integrated Valuation of Ecosystem Services and Tradeoffs) at a resolution of 250 x 250 meters grid cells to determine the exposure of communities in the Gulf of Nicoya to erosion and inundation during storms through a qualitative estimate in terms of a vulnerability index.

After the model generated the different output maps, such as the exposure index, we used the habitat role map for each shoreline segment to classify the total area of mangrove forests of the Gulf of Nicoya into three categories (low, medium and high) depending on the level of protection that mangroves provide. For each category, we assigned a weight as follows: Low = 0.33, Medium = 0.66 and High 1. We finally multiplied the per hectare value of the coastal protection service (calculated using benefit transfer) by these weights and then by the area of mangroves of each category.

$$CPv = CPvh * W * A \quad (3)$$

Where:

CPv = Coastal protection value

CPvh = Coastal protection value per hectare

W= Weight of mangrove category

A= Area of mangrove category

Results

Benefit transfer results

We updated the mangrove cover map from Rivera (2018) by extracting salt ponds and shrimp farms that were not taken into account in that map (Figure 2). These activities in total accounted for 814 hectares, which results in a new total mangrove area in the Gulf of Nicoya of 19,924 hectares.

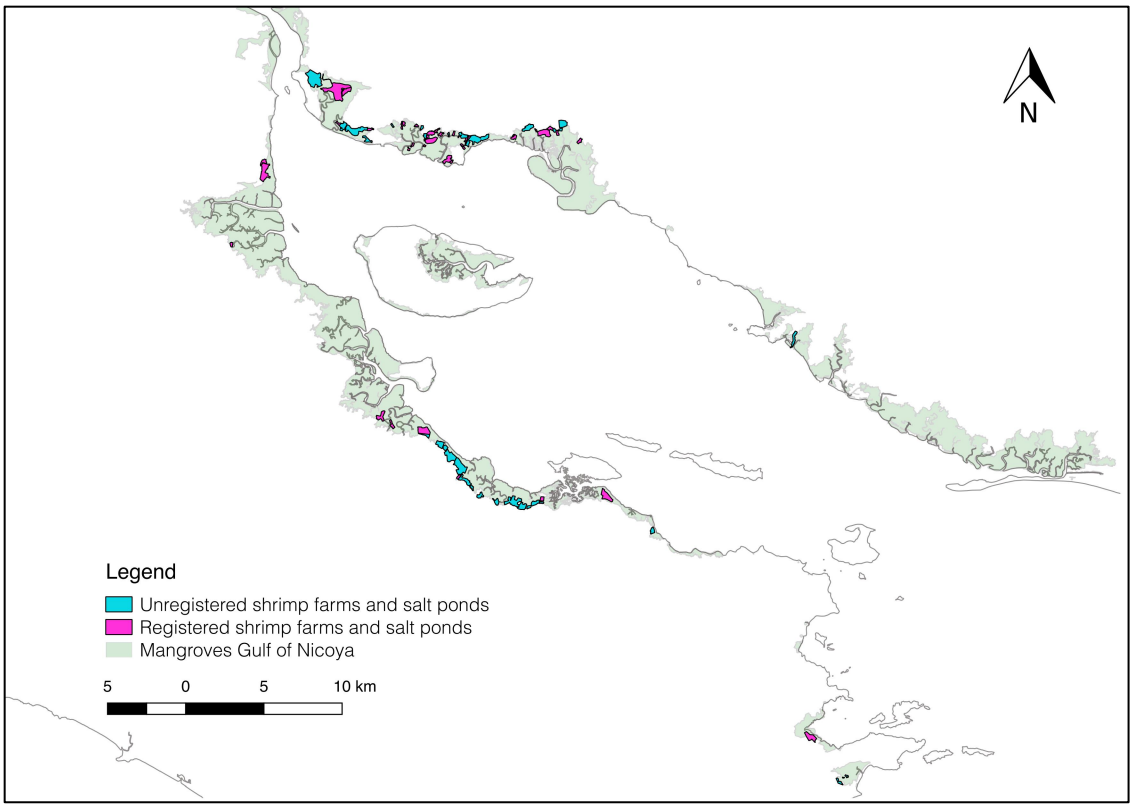

Figure 2. Shrimp farms and salt ponds that have a permit to operate by the government, as well as those that do not have it but are operating or operated in the past.

The first part of the application of the benefit transfer method, which was the estimation of a per hectare per year value from the ESVD of the ecosystem services provided by mangroves, shows that the ecosystem service with the highest mean value is timber and fuelwood, $17,652/ha/year, followed by biodiversity protection ($10,651/ha/year) and coastal protection ($7,638/ha/year). Other services with high economic value are food ($2,002/ha/year) and raw materials ($1,366/ha/year). Nevertheless, median values provide a different panorama, with coastal protection with the highest value ($2,997/ha/year), followed by timber and fuelwood ($315/ha/year), food ($293/ha/year) and climate regulation ($287/ha/year). We found that one hectare of mangrove can provide average economic benefits of $95,979 per year (median=$6,226 /ha/year) through the provision of these 11 ecosystem services valued. By multiplying these values by the mangrove cover in the Gulf, we estimated the economic value of

11 ecosystem services of these mangroves in $812 million per year (median=$88 million/year) (Table 4).

Table 4. Ecosystem services that were valued using the ESVD, the number of estimates that were used, and the minimum, maximum, mean and median values per hectare per year of each service that was calculated, as well as the results of the application of these values to the total mangrove extension of the Gulf of Nicoya (19,924 ha) using the benefit transfer method. All values are in 2015 international dollars.

Ecosystem Service	Value per hectare per year					Gulf of Nicoya		National Assessment	
	Number of estimates	Min Value	Max Value	Mean Value	Median value	Mean Value	Median value	Mean Value	Median value
Provisioning Services									
Food	18	0.06	22,804	2,002	293	39,896,691	5,840,970	72,587,083	10,626,922
Medical/Bioprospecting	3	10	734	258	31	5,144,858	613,949	9,360,432	1,117,003
Fibbers	1			6	6	112,718	112,718	205,076	205,076
Fodder	1			15	15	294,726	294,726	536,218	536,218
Sand, rock, gravel. Coral	2	0.06	104	52	52	1,037,136	1,037,136	1,886,941	1,886,941
Timber and fuel wood	9	52	22,443	17,652	315	351,713,024	6,267,881	639,898,252	11,403,632
Other raw material	7	1	5,328	1,366	233	27,220,649	4,652,300	49,524,597	8,464,283
Total Provisioning Services	**47**	**74**	**139,371**	**21,351**	**945**	**425,419,802**	**18,819,680**	**773,998,598**	**34,240,074**
Regulating Services									
Climate regulation	4	11	2,428	753	287	15,011,447	5,726,869	27,311,467	10,419,328
Coastal protection	8	180	27,638	7,638	2,997	152,187,141	59,708,937	276,885,638	108,633,010
Total Regulating Services	**15**	**777**	**33,187**	**9,856**	**3,970**	**167,198,587**	**65,435,806**	**304,197,105**	**119,052,337**
Cultural Services									
Recreation/tourism	3	52	944	354	65	7,047,295	1,287,048	12,821,680	2,341,624
Total Cultural Services	**3**	**52**	**944**	**354**	**65**	**7,047,295**	**1,287,048**	**12,821,680**	**2,341,624**
Support Services									
Biodiversity protection	5	15	36,313	10,651	116	212,214,578	2,315,253	386,098,120	4,212,315
Total Support Services	**19**	**53**	**381,885**	**64,418**	**1,247**	**212,214,578**	**2,315,253**	**386,098,120**	**4,212,315**
TOTAL	**84**	**955**	**555,387**	**95,979**	**6,226**	**811,880,262**	**87,857,786**	**1,477,115,503**	**159,846,351**

Using the same method, we estimated the total mean value of the ecosystem services provided by the total extent of mangroves in Costa Rica as $1.5 billion per year (median=$160 million/year). Considering the change in national cover of mangroves from 1980 to 2013 (FAO, 2007a), we estimate that Costa Rica lost an average $1.1 billion per year (median=$120 million per year) during that period because of the loss of ecosystem services from mangroves.

It is worth noting that these values calculated using benefit transfer, express potential values – the supply of ecosystem services – since they were not related with the local beneficiaries of each ecosystem service. We calculated the demand of some of these ecosystem services with the help of a panel of experts, allowing us to modify the value of ecosystem services of the Gulf of Nicoya and have more accurate results. Specifically, the panel of experts assessed the locations and extensions of the following ecosystem services: 1) medical/bioprospecting, 2) fibers, 3) fodder, 4) sand, rock, gravel and coral, 5) timber and fire wood, 6) other raw materials and 7) recreation. Of these seven services, we excluded medical/bioprospecting, fibbers, sand, rock, gravel and coral and other raw materials since the experts pointed out that none of these are being demanded/used in the Gulf.

According to the panel of experts, the remaining three ecosystem services are demanded in a small portion of the area of the Gulf of Nicoya, with firewood and timber accounting for 2,811 hectares (14% of the total area of mangroves in the Gulf), tourism 2,273 hectares (11%) and fodder 998 hectares (5%) (Figures 3, 4, 5).

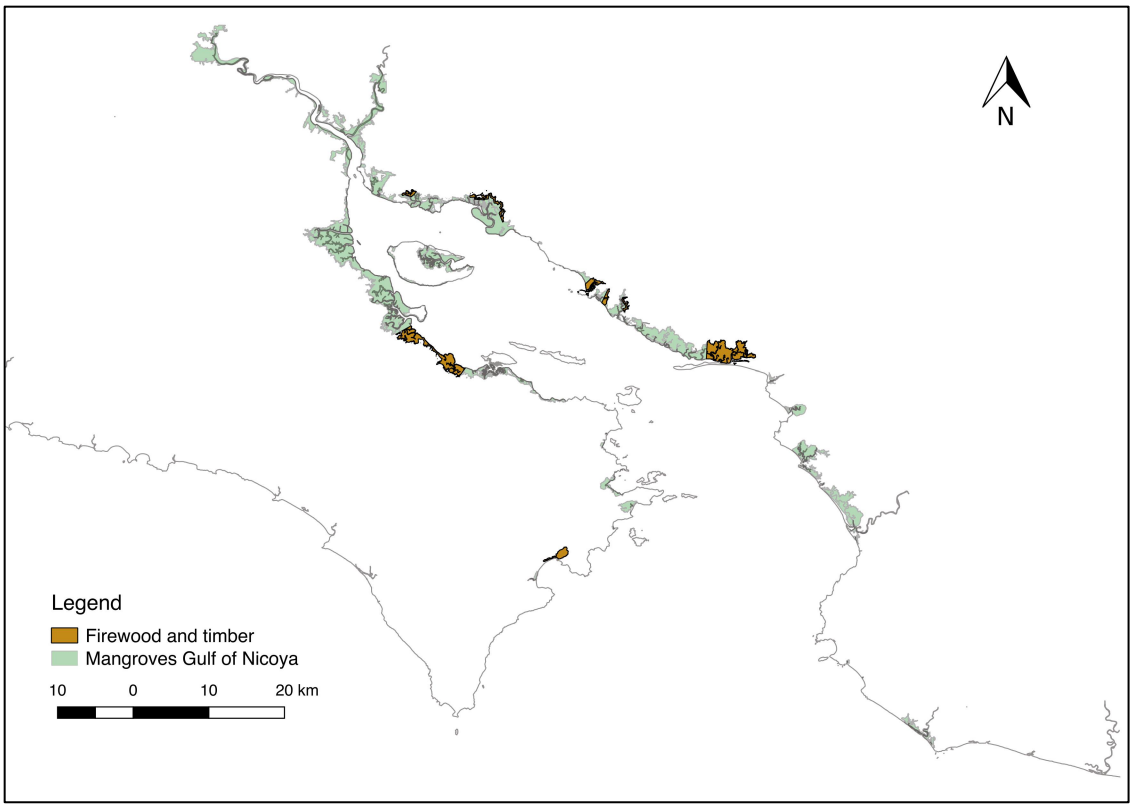

Figure 3. Locations where firewood and timber are extracted, in the districts of Puntarenas, Chomes, Colorado-Abangares, Lepanto and Paquera.

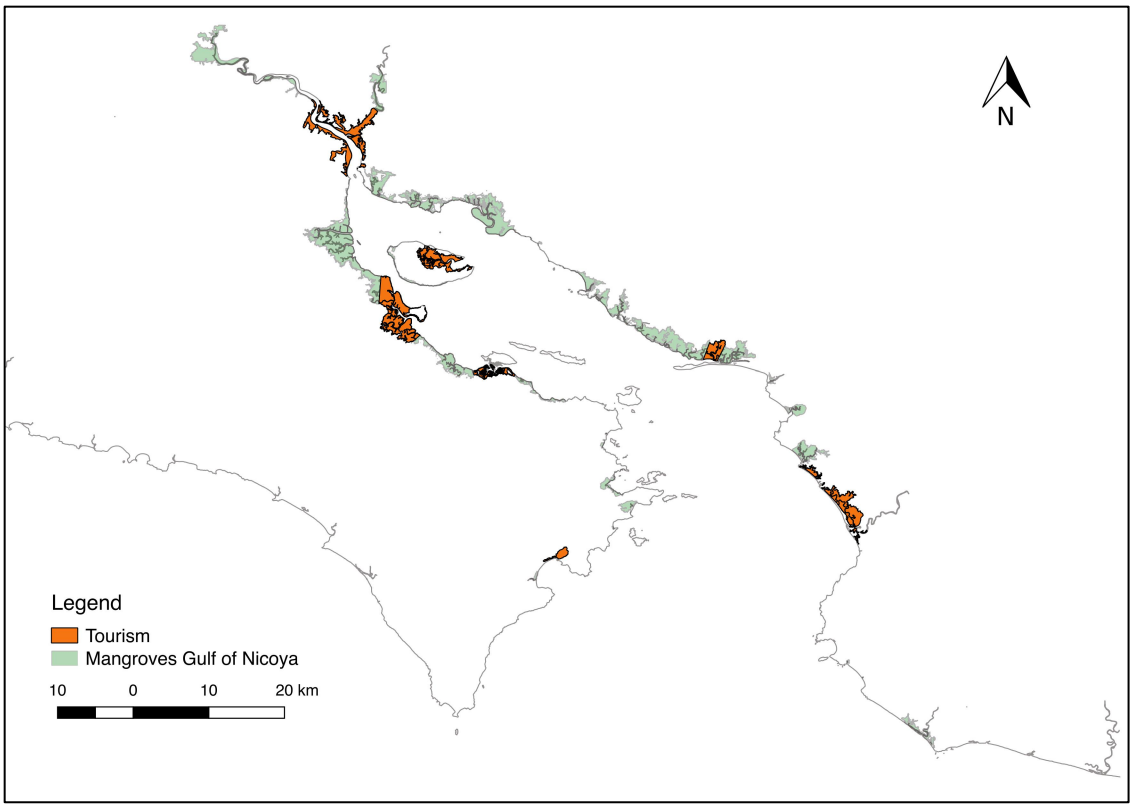

Figure 4. Locations where touristic activities related to mangroves are developed, in the districts of Puntarenas, Tarcoles, Porozal, Nicoya, Chira Island, San Pablo-Nandayure, and Lepanto.

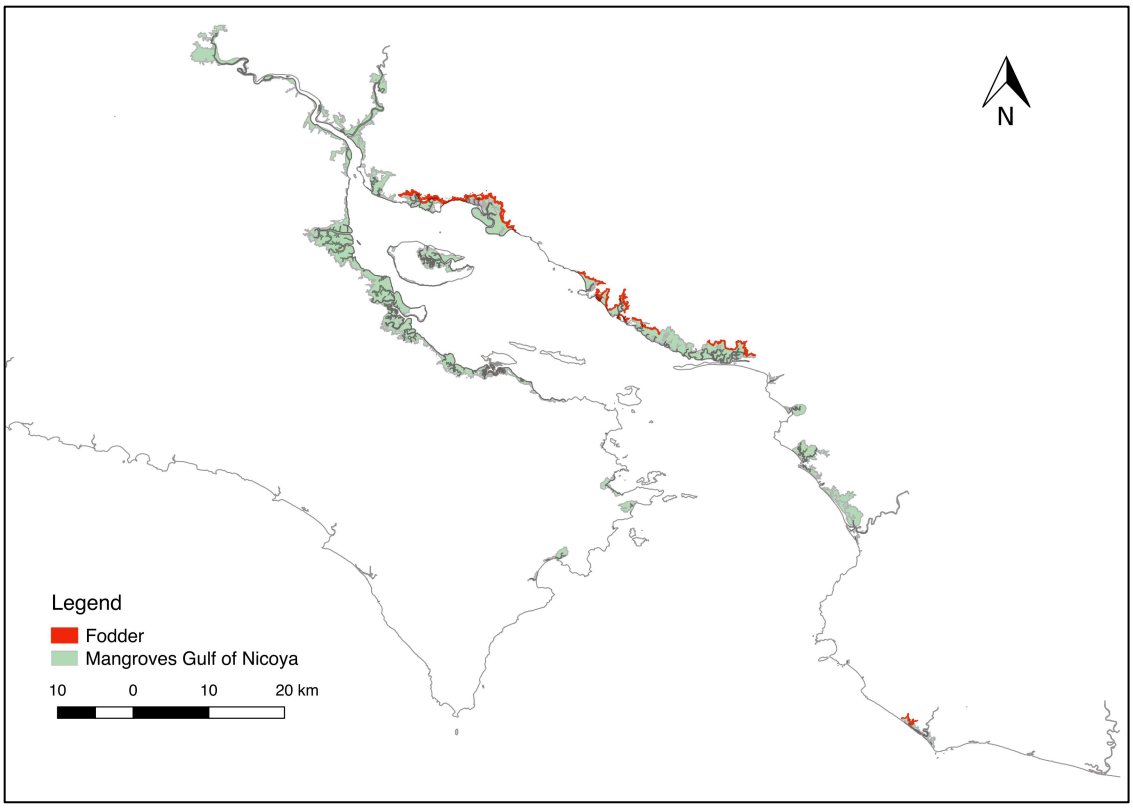

Figure 5. Locations where fodder is used for cattle, in the districts of Puntarenas, Chomes, Manzanillo and Colorado-Abangares.

Having determined which ecosystem services are provided in reality by mangrove forests in the Gulf of Nicoya, we produced a new set of value estimates (Table 5). According to our modified benefit transfer, the highest mean values of the mangroves of the Gulf of Nicoya comes from biodiversity protection ($212 millions/year), coastal protection ($152 millions/year) and timber and fuelwood ($50 millions/year). The highest median values are from coastal protection ($60 millions/year), food ($5.8 millions/year) and climate regulation ($5.7 millions/year), all three ecosystem services valued in this study as well using primary methods.

By applying the expert modified benefit transfer based on seven ecosystem services, we estimated that the mean total value of the mangrove forests of the Gulf of Nicoya is $470 million per year, and a median value of $75 million per year.

Table 5. Results from the expert modified benefit transfer. Zero indicates services that experts considered did not apply for the Gulf of Nicoya, and numbers in blue indicate services that were re-estimated according the areas defined by experts.

Ecosystem Service	Gulf of Nicoya	
	Mean Value	Median value
Provisioning Services		
Food	39,896,691	5,840,970
Medical/Bioprospecting	0	0
Fibers	0	0
Fodder	14,760	14,760
Sand, rock, gravel. Coral	0	0
Timber and fuelwood	49,618,917	884,259
Other raw material	0	0
Total Provisioning Services	**89,530,368**	**6,739,990**
Regulating Services		
Climate regulation	15,011,447	5,726,869
Coastal protection	152,187,141	59,708,937
Total Regulating Services	**167,198,587**	**65,435,806**
Cultural Services		
Recreation/tourism	804,021	146,838
Total Cultural Services	**804,021**	**146,838**
Support Services		
Biodiversity protection	212,214,578	2,315,253
Total Support Services	**212,214,578**	**2,315,253**
TOTAL	**469,747,554**	**74,637,886**

Primary studies results

The three ecosystem services we chose to value using this approach in the Gulf of Nicoya, were validated by the experts at SINAC. These services were also ranked by the experts from low to high depending on their importance in each Conservation Area (Table 6)

Table 6. Ecosystem services ranked by experts depending on their level of provision in each Conservation Area.

Ecosystem service	ACOPAC	ACAT	ACT
Food (fish)	3	3	3
Food (mollusks)	3	3	3
Coastal protection	1-3	2	2

Low=1, Medium=2, High=3

All interviewees agreed that food, including, fish, mollusks and shrimp, is the most important benefit that local communities received from mangroves. In the case of coastal protection, experts from ACOPAC differentiated the service geographically between Puntarenas and the rest of the Conservation Area due to differences in economic activities and the intensity of urbanization. Climate regulation was not included in this survey since it is not geographically dependent across the study area. Other services that are important in the Gulf of Nicoya are education/research in ACOPAC, which is focused on mollusks and the health of the mangrove; and salt production and shrimp aquaculture in ACAT and ACT.

Climate regulation - Carbon stocks

From the statistical and geographical grouping of carbon stocks that we conducted, we divided the total extension of mangrove in the Gulf of Nicoya in three zones (Figure 6). Zone 1, the upper part of the Gulf, has the lowest carbon stocks, 547 MgC/ha, and Zone 3 the highest, 1175 MgC/ha. This range of values of carbon stocks follow a latitudinal gradient in the Gulf, probably due to differences in micro-elevation and hydrodynamics, underlying geomorphology, and salinity from the north end of the area to the mouth of the Gulf at its southern end.

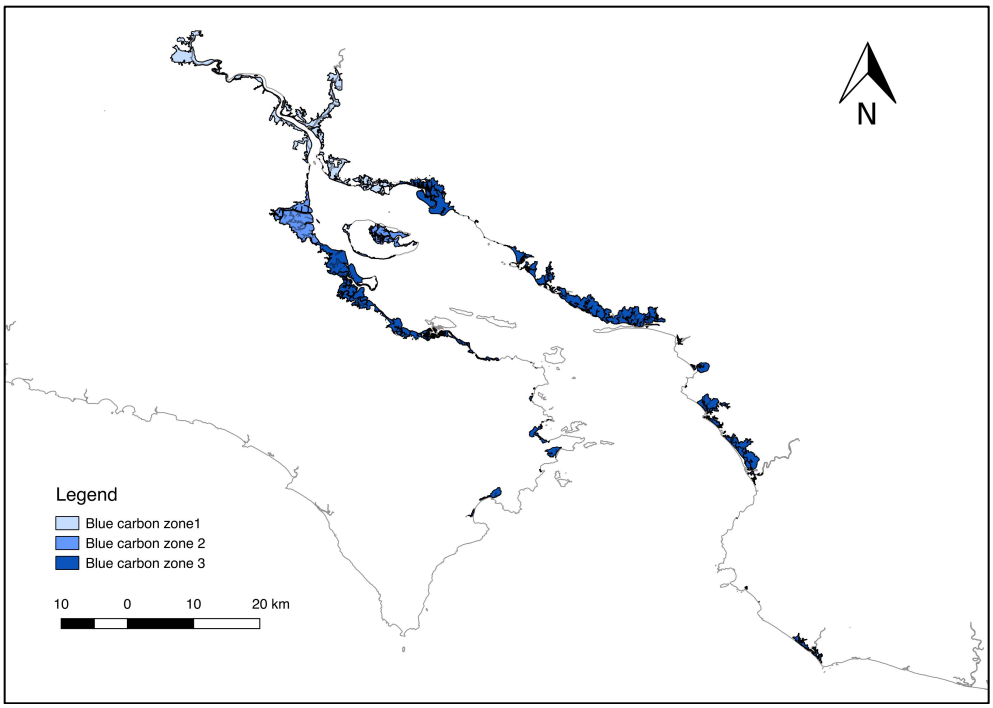

Figure 6. Distribution of carbon stocks according to its concentration across the Gulf of Nicoya. Darker areas show higher densities of carbon stored.

Zone 3 is also the largest in extension, and therefore, the zone with the highest total carbon stocks and the highest value, $1,9 billion/year. Zone 2 and 1 are valued in $377 millions and $359 millions respectively. The total economic value of the carbon stock services of the three zones, and therefore, the entire Gulf, is $2,6 billions (Table 7).

Table 7. Economic value of the service of carbon stocks disaggregated by blue carbon concentration zones and sites in the Gulf of Nicoya.

Zone	Site	C ecosystem (Mg/ha)	Mean C (Mg/ha)	Area (ha)	C per zone (Mg)	MAC (2015 $/MgC)	Total economic value (2015 $)
1	Buenaventura	413.09	546.83	4,830	2,641,141.66	135.82	358,719,860.91
	Bebedero	601.49					
	Níspero	625.90					
2	Isla Chira	839.96	847.02	3,280	2,778,183.36	135.82	377,332,863.28
	Jesús	854.07					
3	Thiel	1,010.65	1,175.17	11,814	13,883,447.57	135.82	1,885,649,849.22

Colorado	1,074.39
Paquera	1,160.58
Puntarenas	1,212.19
Lepanto	1,259.46
Jicaral	1,333.74

Because our study is focused on the economic valuation of ecosystem services, and services are flows and not stocks, we cannot add these results of carbon stocks to the rest of economic values of ecosystem services. Nevertheless, we consider important to estimate both the biophysical and economic values of carbon stocks in the Gulf of Nicoya for future policy decisions. Our estimates can also feed directly into new national natural capital estimates for Costa Rica, which to date do not include this type of information and are, thus, grossly underestimated.

Climate regulation - Carbon sequestration

Applying the Social Cost of Carbon of 87 $/MgC to a sequestration rate of 6 $MgCO_2eq/ha/year$, the total economic value of the carbon sequestration service is $38,151,655 (Table 8).

Table 8. Total economic value of the carbon sequestration service based on the Social Cost of Carbon and a mean sequestration rate obtained from the literature.

Sequestration rate ($MgCO_2eq/ha/year$)	SCC (2015 $/MgC)	SCC (2015 $/$MgC0_2$)	Area (ha)	Total economic value (2015 $)
6	87	319	19,924	38,151,655

Fisheries

In Costa Rica, between 75-80% of the total fish landings are made by the artisanal fleet, and approximately 95% of these landings come from the Pacific Ocean, and more specifically, the Gulf of Nicoya (Ocean Outcomes, 2018). To estimate the value of fisheries in the Golf of Nicoya, we selected the species with the highest commercial interest that are caught by the artisanal fleet. According to a sampling made by Araya & Vasques (2005), 40% of fish catches comes from

four species of the family Sciaenidae: *Cynoscion albus* (Queen corvina), *Cynoscion squamipinnis* (Scalyfin corvina), *Cynoscion phoxocephalus* (Sharpnose corvina) and *Cynoscion stolzmanni* (Yellowfin corvina). This study also found that white shrimp (*Litopenaeus sp*.) was one of the most important species from the commercial perspective (Araya & Vasques, 2005).

In another study conducted by Araya et al. (2007), they estimated that, between 2002 and 2005, these same species (except the Yellowfin corvina) represented 31% of fish catch in the Gulf of Nicoya. Furthermore, a more recent study from Marín (2015), in which the author sampled more than sixty fish species caught in the Gulf of Nicoya in 2014, found similar results than Araya and Vasques (2005) and Araya (2007). Marín argues that five species represented 76% of the total catch sampled (Queen corvina = 43%, Scalyfin corvina = 13%, White shrimp = 9%, Sharpnose corvina = 7%, and Snook = 4%).

From INCOPESCA's fish catch database for the Gulf of Nicoya (Table A3.1 from Appendix 3), we determined that the commercial categories of "first large", "first small", and "class"[5] represented together 30% of 2015 catches, spotted rose snapper 4%, white shrimp 4%, and bivalves 2%. These six commercial categories account for 40% of the total catch. According to Marín (2018), the categories of "first large", "first small" and "class", can be disaggregated by species as shown in A3.2-A3.4 from Appendix 3.

The species that were fished the most under these three commercial categories are the Queen corvina, the Scalyfin corvina and the Sharpnose corvina, which supports the findings of the studies previously mentioned.

The same INCOPESCA data base (Table A3.1 from Appendix 3) that contains the aggregated information for fish catch in the Gulf of Nicoya, also provides aggregated data for the extraction of bivalves, and therefore, this information had to also be disaggregated, which was done by the Statistics Department of that organization, as shown in Table A3.5 from Appendix 3.

After processing the initial data base and determining the species that conforms each commercial category, we confirmed that all fish species contained in "first large", "first small" and "class" utilise mangroves as habitat during their life cycle (Rönnbäck, 1999). This is also the

[5] First large = individuals of 2kg or more. First small = individuals of less than 2kg. Class = individuals between 800g and 1kg

case for spotted rose snapper and white shrimp (Goti, 1991; Rönnbäck, 1999), as well as for the species of bivalves assessed in this study (Morton, 2013) (Table A3.6 from Appendix 3). Once we had identified the species of highest commercial interest, and confirmed that these species depend on mangroves, we were able to select these categories/species to value in this study.

We obtained the monthly mean prices for all commercial categories from INCOPESCA (2018). The results of the annual value of first large, first small, class, spotted rose snapper and white shrimp are shown in Table 9. The category of first small has the highest annual value, ($1,475,451) and the highest catch (406,087kg). Although white shrimp have the lowest catch of these five categories, it is the second most valuable of them, since it has the highest monthly mean price per unit (kg) of all, which is three times or more the price of the other categories.

In the case of bivalves, we used the values on catch and mean price per species per area of extraction provided by Duran (2018) (Table 10). Here, clams are the type of bivalves that are extracted in the highest quantities (25,090kg), representing 70% of the total extraction of bivalves, but they have the lowest mean price ($1.4/kg) and therefore, they account for only 20% of the economic value of this category of organisms. On the other hand, piangua represents 15% of the annual extractions (5,556kg), but it accounts for 74% of the economic value due to its high mean price per unit ($22/kg).

Because bivalves stay in the same location for the majority of time, in contrast with fish, that move around, inside and outside the Gulf, it is possible to determine their exact location of extraction, and therefore, map the provision of this ecosystem service with high accuracy (Figures 7 and 8). From this analysis, we found that Jicaral is the location with the highest extraction and economic gains, followed by Colorado and Chomes.

Table 9. Total monthly catch (kg) and price (2015 USD) of the most important (in terms of catch and value) commercial categories in the Gulf of Nicoya.

Commercial category	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	TOTAL
First large (kg)	13,253	24,203	18,507	19,477	7,112	2,174	1,807	24,183	25,321	17,555	11,613	13,842	**179,047**
Price (USD)	6	6	6	6	5	6	5	4	3	4	5	6	5
Value/month (USD)	76,260	148,663	119,902	124,242	37,440	12,751	9,803	92,754	85,792	69,311	55,425	79,905	**912,247**
First small (kg)	46,844	45,461	56,887	35,964	9,564	3,825	3,062	36,013	55,640	38,722	40,682	33,423	**406,087**
Price (USD)	4	4	4	4	3	3	3	3	3	3	4	4	4
Value/month (USD)	174,907	169,323	226,917	146,781	33,147	12,931	10,652	123,083	165,789	126,043	147,613	138,264	**1,475,451**
Class (kg)	35,865	32,767	40,488	42,621	19,439	11,043	22,107	26,693	41,291	25,213	20,742	32,420	**350,689**
Price (USD)	2	2	2	2	2	2	2	2	2	2	2	2	2
Value/month (USD)	61,985	58,629	77,683	81,618	39,812	25,393	51,242	47,613	70,294	44,321	40,372	64,660	**663,623**
Spotted rose snapper (kg)	6,112	5,420	6,838	11,067	9,473	5,641	3,478	5,341	6,916	4,775	6,464	2,199	**73,724**
Price (USD)	5	4	5	4	4	3	4	4	4	4	4	5	4
Value/month (USD)	28,057	23,929	32,023	48,302	33,179	19,542	12,380	21,119	27,513	18,099	24,584	10,888	**299,616**
White shrimp (kg)	12,346	10,008	6,560	6,669	1,711	60	5	10,233	7,104	5,325	4,204	6,916	**71,141**
Price (USD)	17	17	18	19	18	14	28	13	13	13	13	13	16
Value/month (USD)	206,984	169,174	117,449	128,252	30,558	860	139	132,027	89,647	67,168	52,740	87,517	**1,082,514**

Source: INCOPESCA, 2018

Table 10. The commercial category "Bivalves" disaggregated by species, showing each species catch (kg) and value (2015 USD) in each location of the Gulf of Nicoya in 2015

Location	Bivalve	Catch (kg)	Mean price (USD/kg)
Berrugate	Clams	4	1
	Piangua	8	21
Chira	Piangua	21	21
Chomes	Clams	612	2
	Chora	516	2
	Mussels	1,340	2
	Piangua	733	21
Colorado	Clams	446	1
	Chora	1,300	1
	Mussels	220	2
	Piangua	1,523	27
Corozal	Clams	244	1
	Piangua	341	24
Isla Venado	Clams	1,132	1
	Chora	130	1
	Piangua	182	27
Islita	Clams	1,390	2
	Chora	50	2
	Mussels	225	3
	Piangua	1,125	20
Jicaral	Clams	20,420	1
	Chora	1,000	2
	Piangua	1,198	28
Las Ramas	Piangua	4	18
Moraga	Clams	50	2
	Mussels	150	2
	Piangua	88	20
Pajaritas	Clams	257	2
Palito	Chora	204	2
	Clams	112	1
	Piangua	126	20
Punta Morales	Clams	423	1
	Chora	96	2
	Mussels	200	2

Piangua	207	21

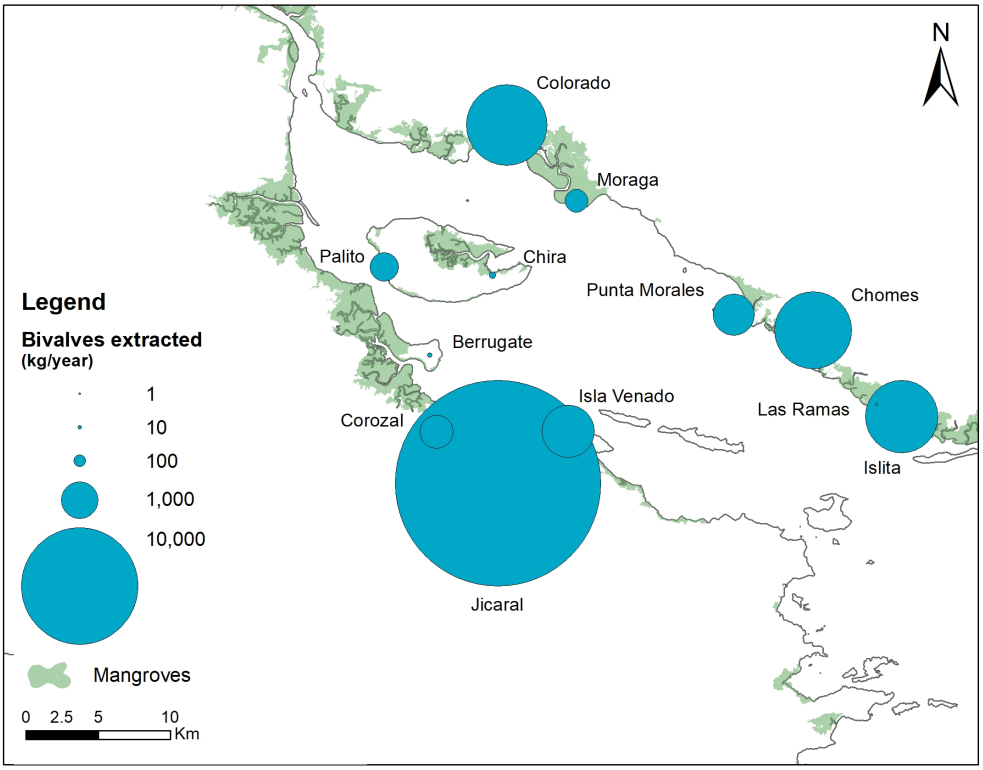

Figure 7. Proportion of total bivalves extracted in each location of the Gulf of Nicoya in 2015. Source: Own elaboration with data from INCOPESCA

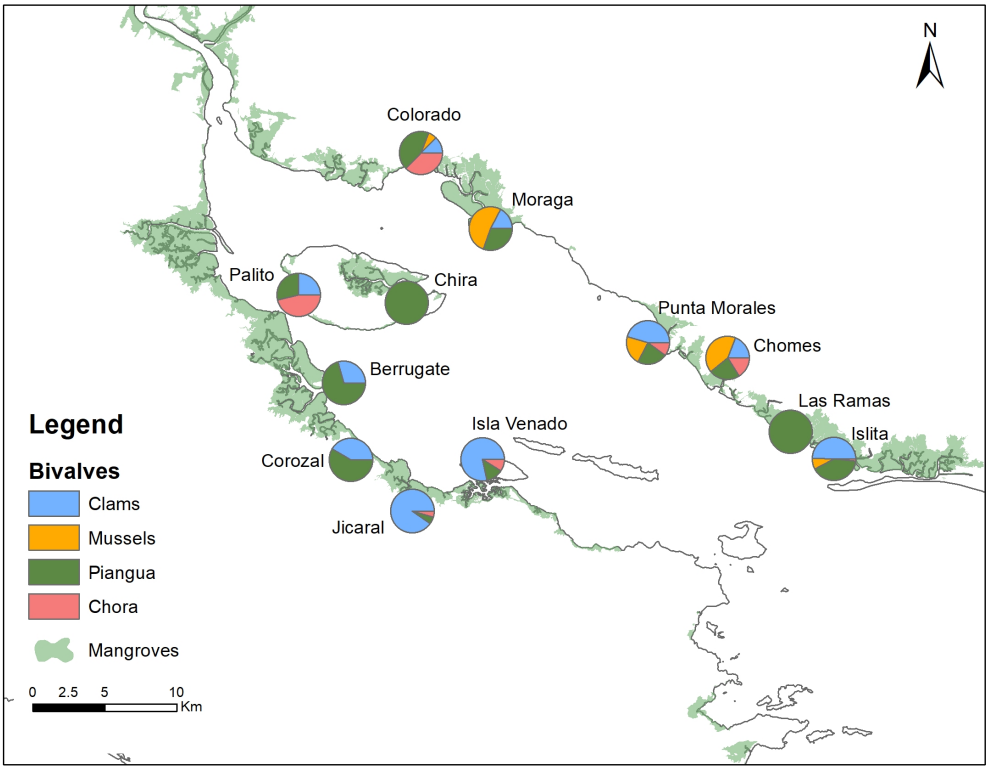

Figure 8. Species composition of the total bivalves extracted at each location in 2015.
Source: Own elaboration with data from INCOPESCA

Looking at the total catches and economic values by species in 2015 in the Gulf of Nicoya, the queen corvina is the species that is fished in the highest quantities of all species assessed (311,771kg), as well as having the highest economic value ($1,264,579). The Scalyfin corvina is the second most fished species of all (267,892kg) but has the third highest economic value ($709,818) since white shrimps are the second most valuable species ($1,082,514) (Table 11).

In total, the provisioning service of food (i.e. fisheries) in the Gulf of Nicoya has an economic value of $4,613,471. As established in the methods section, we assumed that the marginal and average products of mangrove area are equal for all species harvested (following Costanza et al. 1989), which results in a catch of 54kg per hectare of mangrove, and a total value of $222 per hectare.

Table 11. Summary of total catch (kg) and value (2015 USD) of each species fished in 2015 in the Gulf of Nicoya.

Species	**Total**		**Total per ha**	
	Catch (kg)	Value (USD)	Catch per ha (kg)	Value per ha (USD)
Scalyfin corvina	267,892	709,818	13	34
Sharpnose corvina	106,057	301,408	5	15
Queen corvina	311,771	1,264,759	15	61
Golden croaker	71,637	256,578	3	12
Striped corvina	2,758	7,892	0.1	0.4
Highfin king croaker	438	829	0.02	0.04
Pacific smalleye croaker	14,621	30,912	1	1
Panama kingcroaker	4,632	9,997	0.2	0.5
Yellowfin corvina	14,841	54,579	1	3
Armed snook	40,424	92,036	2	4
Union snook	16,729	32,211	1	2
Blackfin snook	16,374	47,180	1	2
Flathead Mullet	7,878	14,908	0.4	1
Atlantic Tripletail	8,791	16,636	0.4	1
Snook	46,650	203,389	2	10
Sea catfish	2,782	5,265	0.1	0.3
Barracuda	1,546	2,925	0.1	0.1
Spotted rose snapper	73,724	299,616	4	14
White shrimp	71,141	1,082,514	3	52
Clams	25,090	35,943	1	2
Piangua	5,556	133,956	0.3	6

Chora	3,296	5,246	0.2	0.3
Mussels	2,135	4,874	0.1	0.2
Total	**1,116,765**	**4,613,471**	**54**	**222**

Coastal protection

The Sea Level Rise map that we produced using data from the AVISO+ website (Figure 9) shows the northern zone of the Gulf to be the one that is experimenting the highest rise, with 2.93 mm/year in the districts of Chomes, Pitaya, and parts of Puntarenas, and 2.88 mm/year in the districts of Manzanillo, Colorado-Abangares, Mansion and Chira Island. In the middle zone of the Gulf, in the districts of El Roble, Espiritu Santo, San Juan Grande, Paquera and the majority of Tarcoles, sea level is rising at a rate of 2.7 mm/year. Finally, the southern zone of the Gulf has the lowest trends of sea level rise, in the district of Cobano is 2.46 mm/year and in Jaco, one of the most populated beaches of the country, 2.32 mm/year.

Using the INVEST Coastal Vulnerability model we produced a total of nine maps. The outcome maps of shore exposure, relief, natural habitats (based on the map of mangrove forests cover), wave exposure, surge potential and sea level rise (based on the map that we produced as explained before) (Figures A4.1- A4.6 from Appendix 4), were used in combination to estimate the coastal exposure index which shows the more vulnerable areas of the Gulf to erosion and inundation during storms (Figure 10). On one hand, the coastal exposure index maps show that the districts of Chomes, Pitahaya, Puntarenas, El Roble, Espíritu Santo, San Juan Grande, and Tárcoles as the most vulnerable. On the other hand, Quebrada Honda, Mansión, San Pablo Nandayure, and Lepanto, which are all located in the western side of the Gulf, have the lowest vulnerability index.

Our main outcome map, the habitat role (Figure 12), is the difference between the coastal exposure map (Figure 10) and the coastal exposure without habitats map (Figure 11), and it was used to classify the total area of mangroves of the Gulf of Nicoya depending on the intensity of the provision of the coastal protection service, showing the highest intensity in the districts of Chomes, Pitahaya, Puntarenas and the majority of Tárcoles, and the lowest intensity in Quebrada Honda and Mansión (Figure 13).

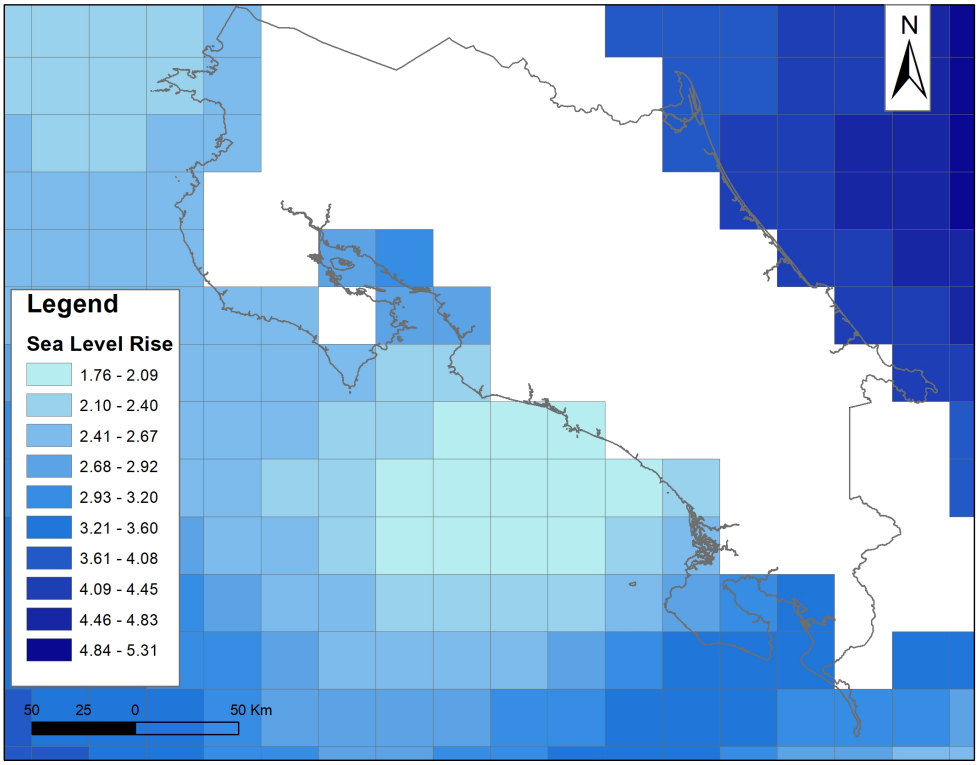

Figure 9. Mean sea level rise in Costa Rica in mm/year. Darker areas show higher sea level rise. The Gulf of Nicoya is experiencing a medium sea level rise compared to the other areas of the country.

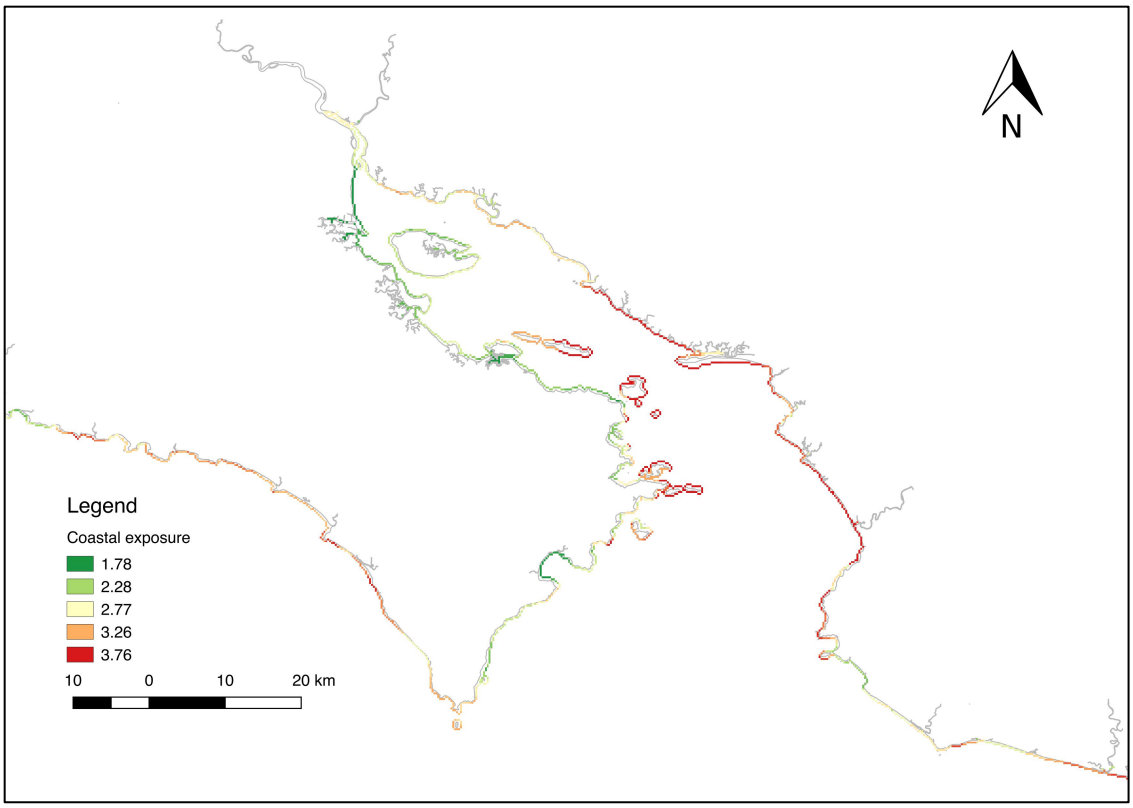

Figure 10. Coastal exposure index raster. Coastal areas in red means higher vulnerability, and in green lower vulnerability. This model output takes in consideration both ecological and social-economic factors.

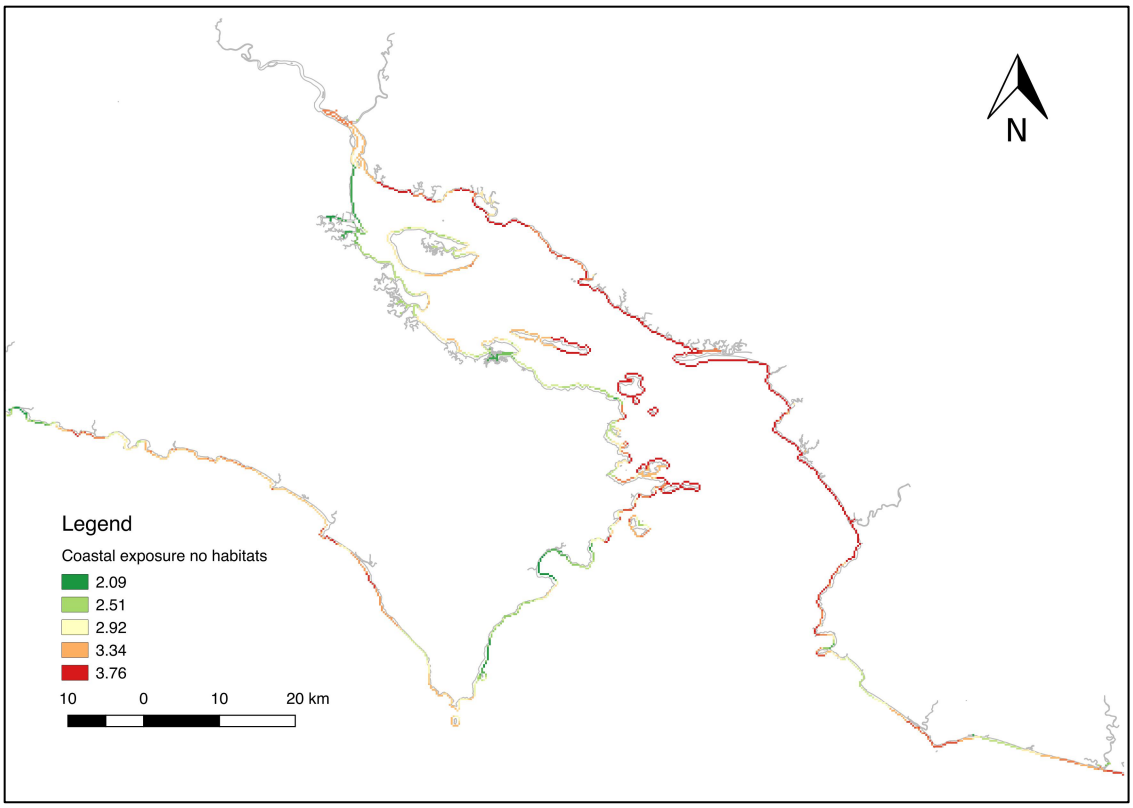

Figure 11. Coastal exposure without taking into consideration habitats, raster containing values computed from the same equation as the coastal exposure raster except the natural habitats layer has been replaced by the constant 5.

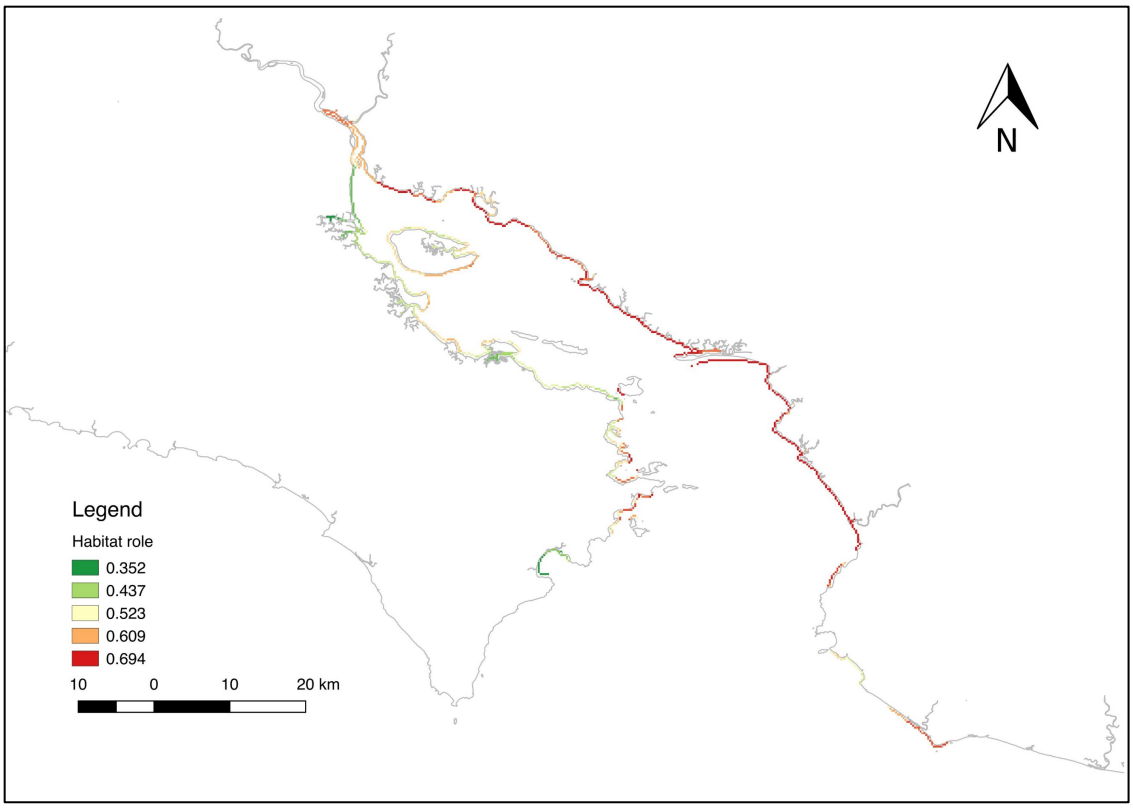

Figure 12. Habitat role, raster difference between the coastal exposure map (Figure 10) and the coastal exposure without habitats map (Figure 11).

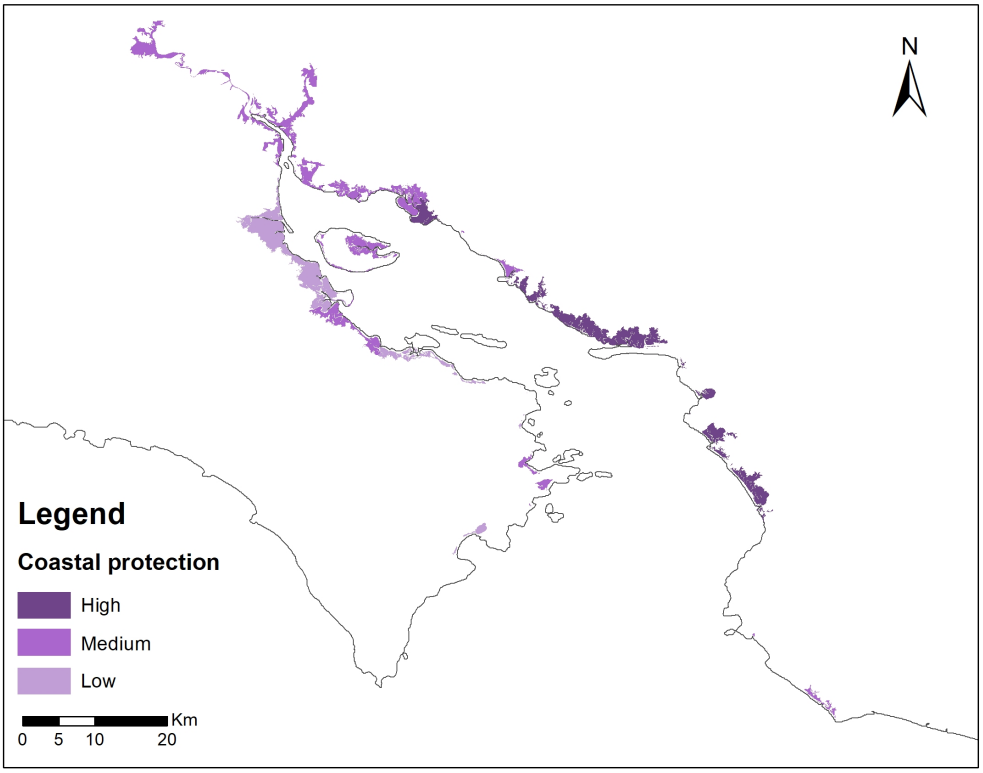

Figure 13. Total area of mangrove forest of the Gulf of Nicoya classified according to its level of coastal protection.

The area categorization of the Gulf of Nicoya using the habitat role map produced by the model, resulted in 5,157 hectares that receive a low protection from mangroves, 8,894 hectares with medium protection, and 5,874 with high protection. It is important to note again that the protection level that mangroves provide for each location is a function of all the variables mentioned in the Methods section that we used as inputs for the model.

We estimated, through the benefit transfer modified by modelling method, that the total mean value of the coastal protection service is $103 million per year, and a median total value of 40 million per year (Table 12). It is worth noting that this represents a snapshot of the current cover and health of the ecosystem, and further research could assess how biophysical variables, such as sea-level rise, can affect the coastal protection service from mangroves in the future, which we assume is a nonlinear function, and therefore, the value of the ecosystem service would depend on two interacting nonlinear functions. Furthermore, considering how this service and sea-level rise interact with each other, there may also be a tipping point somewhere along this curve where the protective role of mangroves is overwhelmed by the magnitude and/or rate of sea-level rise.

Table 12. Summary table of the mean and median total value of the coastal protection service of mangroves in the Gulf of Nicoya, classified by the levels of protection established through the modeling using INVEST.

Level of coastal protection	Mean value per ha (2015 $/year)	Median value per ha (2015 $/year)	Area under each level of protection (ha)	Weight	Mean total value (2015 $/year)	Median total value (2015 $/year)
Low	7,638.00	2,997.00	5,156.96	0.33	12,998,323.96	5,100,285.01
Medium	7,638.00	2,997.00	8,893.76	0.66	44,834,155.66	17,592,035.16
High	7,638.00	2,997.00	5,873.86	1.00	44,864,542.68	17,603,958.42
Total			**19,924.58**		**102,697,022.30**	**40,296,278.58**

Combining the values of the expert modified benefit transfer with the estimates from the primary studies, we calculated the mean total value of the ecosystem services assessed from mangrove forests in the Gulf of Nicoya in $408 million per year, and a median total value of $86 million (Table 13).

Table 13. Summary table comparing the results of the three different methods used to estimate the economic value of the ecosystem services of the mangroves of the Gulf of Nicoya. Numbers in black were estimated through benefit transfer (except subtotals and totals), numbers in blue were estimated using expert modified transfer and numbers in green were estimated using primary studies.

Ecosystem Service	Benefit transfer		Expert Modified transfer		Expert Modified transfer + Primary studies	
	Mean Value	Median value	Mean Value	Median value	Mean Value	Median value
Provisioning Services						
Food	39,896,691	5,840,970	39,896,691	5,840,970	4,613,471	4,613,471
Medical/Bioprospecting	5,144,858	613,949	0	0	0	0
Fibers	112,718	112,718	0	0	0	0
Fodder	294,726	294,726	14,760	14,760	14,760	14,760
Sand, rock, gravel. Coral	1,037,136	1,037,136	0	0	0	0
Timber and fuelwood	351,713,024	6,267,881	49,618,917	884,259	49,618,917	884,259
Other raw material	27,220,649	4,652,300	0	0	0	0
Total Provisioning Services	**425,419,802**	**18,819,680**	**89,530,368**	**6,739,990**	**54,247,148**	**5,512,490**
Regulating Services						
Climate regulation	15,011,447	5,726,869	15,011,447	5,726,869	38,151,655	38,151,655
Coastal protection	152,187,141	59,708,937	152,187,141	59,708,937	102,697,022	40,296,279
Total Regulating Services	**167,198,587**	**65,435,806**	**167,198,587**	**65,435,806**	**140,848,677**	**78,447,933**
Cultural Services						
Recreation/tourism	7,047,295	1,287,048	804,021	146,838	804,021	146,838
Total Cultural Services	**7,047,295**	**1,287,048**	**804,021**	**146,838**	**804,021**	**146,838**
Support Services						
Biodiversity protection	212,214,578	2,315,253	212,214,578	2,315,253	212,214,578	2,315,253
Total Support Services	**212,214,578**	**2,315,253**	**212,214,578**	**2,315,253**	**212,214,578**	**2,315,253**
TOTAL	**811,880,262**	**87,857,786**	**469,747,554**	**74,637,886**	**408,114,424**	**86,422,515**

Discussion

Comparative analysis between techniques of the hybrid approach

Because we aimed to value economically a wide range of ecosystem services provided by mangrove forests in the Gulf of Nicoya, we needed to apply a hybrid approach methodology that had never been used in Costa Rica, and probably elsewhere, combining traditional and novel methods. Applying this approach yielded the first estimations ever done of these ecosystem services in the Gulf of Nicoya, which represents a clear step forward to communicate and utilize in different financial mechanisms the value of this natural capital.

When the benefit transfer method was used, we presented our results both as mean values and median values because we found a significant variance between the estimates that where extracted from the primary studies, which was the case for all ecosystem services assessed. For example, in terms of per hectare per year values, we found that for fisheries the range of values goes from $1 (Turpie, 2000) to $22,804 (R. K. Turner et al., 2003), for timber from $52 (Turpie, 2000) to $22,443 (Gren & Söderqvist, 1994), for coastal protection from $180 (Emerton, 2005) to $27,638 (Barbier, 2007), for tourism from $65 (Tri et al., 2000) to $944 (Cooper et al., 2009), and for biodiversity protection from $15 (Gunawardena & Rowan, 2005) to $36,312 (Bann, 1999).

These results from primary studies vary due to several factors, including valuation method, location, population, study site area, GDP/capita of the country, etc. As more studies become available, it will become possible to estimate the relative influence of these factors on the final results and reduce the variance significantly. For example, DeGroot et al. (2012) produced a meta-regression based on 244 studies of the value of inland wetlands including 17 variables that explained 44% of the variance in the valuation estimates. This sort of analysis will have to wait for more studies of mangrove values. In the meantime, we simply state the range of estimates, their mean and median values in order to communicate the uncertainty in our current estimates.

It is also key to note that when we use the benefit transfer technique, we are estimating the potential value of ecosystem services for a region since we are assuming that the total area

of that region provides these services (i.e. assuming that there are beneficiaries throughout the region under valuation), and for this reason we adjusted our estimates with the help of a panel of experts. The results of the expert modified benefit transfer represent 58% of the initial mean value calculated by benefit transfer, and 85% of the median value, which shows the impact of removing or modifying the value of several ecosystem services.

Having estimated economic values using both benefit transfer methods proved to be helpful to have more accurate results, plus it was a tool for capacity building for the experts that participated in this exercise. Nevertheless, by conducting primary studies for three ecosystem services, we were able to compare these results with the ones obtained in the previous two valuation methods used, showing very similar results in the case of the median value of fisheries, $5.8 million dollars using benefit transfer and $4.6 million dollars with primary studies. This confirms that, if done properly, benefit transfer can be a good first approximation to the value of ecosystem services when time and budget are limited. Furthermore, even though we applied a 3-tier approach to try to have more precise estimates of the economic value of ecosystem services, our results should be used as a reference value (probably an undervaluation), not an exact value, since they have a certain error range that was not possible to calculate at this moment of the research.

In the case of coastal protection, our results using biophysical and demographic modelling varied in relation with benefit transfer in a similar way that our results varied between transfer techniques, since we disaggregated potential areas and actual areas receiving the benefits, resulting in a 67% of the mean and median value calculated using traditional benefit transfer (the proportional difference of mean and median value in relation with the transfer estimates are equal because the variables that we modified, area and its protection capacity, were the same in quantity in both cases).

The results from climate regulation are the most dissimilar between valuation methods, probably due to variables such as the carbon sequestration rate estimated or selected from the literature, and the economic value assigned to each ton of carbon. In neither case, carbon storage or carbon sequestration, we chose the market prices approach, as it was done in a previous study on mangroves from the Gulf of Nicoya conducted by Arguedas-Marín (2015), because these

"prices are generally lower in value since consumers participating in carbon markets are not in a position or are willing to pay the full price required to supply the benefits of carbon storage or sequestration" (Jerath et al., 2016, pp 165), while MAC and SCC are calculated using economic models that combines biophysical factors or climate change and socio-economic aspects of economic growth under different climate change scenarios.

In terms of the total economic values of the ecosystem services assessed, if we compare the combined results from the expert modified benefit transfer and the primary studies with the results from the traditional benefit transfer, the mean total value represents 50% of the original estimates from benefit transfer, and the median total value a 98%, which supports the argument that if benefit transfer is conducted carefully, it can yield good approximations of the actual value of ecosystem services.

Considering only the median values, the combined total value of climate regulation and coastal protection accounts for 91% of the total value of ecosystem services in the Gulf of Nicoya. Moreover, adding the third most valuable service, fisheries, which represents 5% of the total median value, means that using primary studies we were able to estimate the value of 96% of the total median value of ecosystem services in the Gulf, demonstrating again the importance of having selected these ecosystem services to value using primary research.

Lastly, considering again the median total value of ecosystem services from mangroves, it represents 0.16% of the GDP in Costa Rica in 2015, which is also the exact equivalent of the total national budget of the Ministry of Environment of Costa Rica in 2015 (Ministerio de Hacienda de CR, 2015). Furthermore, taking in consideration the recent estimation of the total national expenditure of Costa Rica in environmental protection, which is 0.19% of the GDP (CEPAL, 2018), our estimates of the total median value of ecosystem services from mangroves would be equal to 85% that expenditure.

Policy implications in Costa Rica

Our study supports policies, strategies and initiatives in Costa Rica on wetlands conservation and restoration. One example of these efforts is the work that the Central Bank of Costa Rica is conducting on environmental accounting. Specifically, our results can be

incorporated in the ongoing project on the System of Environmental-Economic Accounting 2012 – Experimental Ecosystem Accounting (SEEA Experimental Ecosystem Accounting), since we developed vital information to be implemented in this framework such as the measurement of the ecosystem (i.e. mangroves) and the biophysical and economic assessment of the services it provides.

Another example is the National Wetlands Policy 2017-2030 which Costa Rica launched in 2017, with the goal "to manage integrally the wetland ecosystems of Costa Rica to contribute to the national development by conserving their ecological integrity and sustainable use of the ecosystem services they provide for current and future generations". To accomplish this goal, the policy established the following two axes of action that are related to ecosystem services and their valuation: 1) "Wetland ecosystem conservation, and its goods and services"; and 2) "Development, ecosystem services provision, and climate adaptation".

Our study contributes mainly to axis 2 and, more specifically, to the following guidelines and activities of the National Wetlands Policy:

- Guideline 2.1 "Scientific and traditional knowledge of the supply of ecosystem services from wetlands". Under the guideline's activity of "Map and determine which productive activities are consistent with the sustainable uses of wetland ecosystems, by analysing the relationship between supply and demand of goods and services", we estimated the economic value and mapped ecosystem services such as fisheries and tourism, providing information on the dependence that these productive activities have on mangroves.
- Guideline 2.4 "Incentives that promote the adoption of good practices, in order to protect the ecological integrity of wetland ecosystems". One of the guideline's activities aims to "create a fund or a financial program for the conservation of wetland ecosystems in the Municipalities", which can use our economic estimates to conduct cost-benefit analysis for the establishment of this fund, as well as to develop financial mechanisms based on our estimates of the value of the ecosystem services from mangroves in the Gulf of Nicoya.

- Guideline 2.5 “Sustainable use, related to the mitigation and adaptation of wetland ecosystems and human populations”. In particular, the activity of “Develop and implement a blue carbon strategy” will be highly benefitted from our findings, especially because we determined that 91% of the economic value of mangroves in the Gulf of Nicoya comes from ecosystem services directly related to climate change mitigation and adaptation (i.e. climate regulation and coastal protection). Our results provide a clear understanding of both the biophysical and socio-economic variables that should be considered to develop a strategy to protect and enhance blue carbon ecosystems such as mangrove forests.

We do not aim to provide here an extensive list of laws, policies and initiatives that our study supports, but to illustrate how economic valuations such as the one conducted here can help to put in practice many governmental actions towards wetlands protection under an ecosystem approach, as well as to stimulate the creation of pioneering policies and programs such as “blue payments for ecosystem services”.

Conclusions

This study constitutes the first of its kind in comparing results of economic valuation of ecosystem services of mangroves using a hybrid “three-tier method”. Starting with traditional benefit transfer, we added expert modified benefit transfer and finally primary studies. Our study supports the use and accuracy of properly conducted benefit transfer. To the best of our knowledge, this is the first hybrid integration of benefit transfer with the INVEST modelling tool for economic valuation of mangroves.

We demonstrated that mangrove forests play a critical role in climate change mitigation and adaptation strategies, and at the same time provide many services that have a positive impact on the well-being of the local communities that depend on these ecosystems.

Furthermore, mangroves are crucial for communities that can be far from their location, such as the case of the provision of food which is consumed throughout Costa Rica.

Our estimates can be used as the bases for policies and strategies towards wetlands conservation and social well-being, and they can also be the basis for future research on social-ecological systems, in order to better understand both the dependence of society on mangrove forests in Costa Rica and in other parts of the world, as well as the human impacts on those ecosystems and their services.

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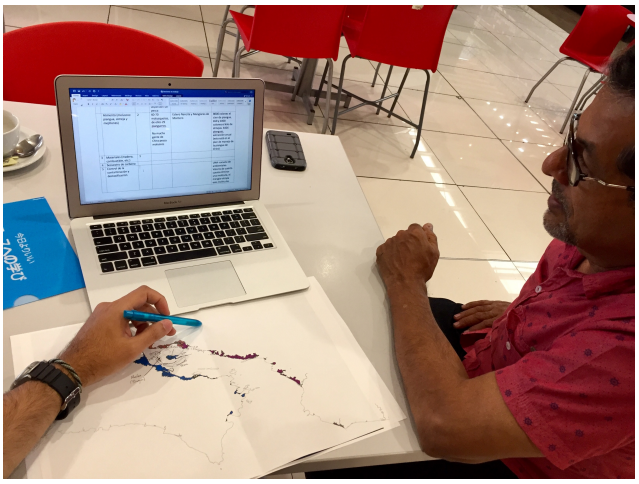
Appendix 1. Experts interviewed for the validation of the list of ecosystem services

Name	Expertise	Meeting place	Date of meeting
Francisco Pizarro	Marine biologist in charge of the management plan for Chira Island	Heredia, Costa Rica	April 11th, 2018
Jamileth Cubero	Mangrove projects - Central Pacific Conservation Area	Puntarenas, Costa Rica	April 16th, 2018
Lara Anderson	Mangrove projects –Tempisque Conservation Area	Guanacaste, Costa Rica	April 16th, 2018
Celso Alvarado	Mangrove projects – Arenal Tempisque Conservation Area	Guanacaste, Costa Rica	April 16th, 2018
Pilar Arguedas	Mangrove projects – INCOPESCA	Puntarenas, Costa Rica	April 20h, 2018




Appendix 2. Studies used for the benefit transfer

Ecosystem service	Method	Reference
	Direct market pricing	Cooper, E., L. Burke and N. Bood (2009) Coastal capital : Belize - The economic contribution of Belize's coral reefs and mangroves. WRI Working Paper. World Resources Institute, Washington, D.C., 53pp.
	Direct market pricing	Turpie, J.K. (2000) The use and value of natural resources of the Rufiji Floodplain and Delta, Tanzania. Rufiji Environmental Managemet Project, Technical report No. 17.
	Direct market pricing	Turner, R.K., J. Paavola, P. Cooper, S. Farber, V. Jessamy and S. Georgious (2003) Valuing nature: lessons learned and future research directions. Ecological Economics 46(3): 493-510.
	Direct market pricing	Samonte-Tan, G.P.B., A. T. White, M. A. Tercero, J. Diviva, E. Tabara and C. Caballes (2007) Economic Valuation of Coastal and Marine Resources: Bohol Marine Triangle, Philippines. Costal Management 35(2): 319-338.
	Direct market pricing	Samonte-Tan, G.P.B., A. T. White, M. A. Tercero, J. Diviva, E. Tabara and C. Caballes (2007) Economic Valuation of Coastal and Marine Resources: Bohol Marine Triangle, Philippines. Costal Management 35(2): 319-338.
	Direct market pricing	Gunawardena, M. and J.S. Rowan (2005) Economic valuation of a mangrove ecosystem threatened by shrimp aquaculture in Sri Lanka. Environmental Management 36(4): 535-550.
Food	Direct market pricing	Gunawardena, M. and J.S. Rowan (2005) Economic valuation of a mangrove ecosystem threatened by shrimp aquaculture in Sri Lanka. Environmental Management 36(4): 535-550.
	Direct market pricing	White, A.T., M. Ross and M. Flores (2000) Benefits and costs of coral reef and wetland management, Olango Island, Philippines. In: Cesar, H. (ed), "Collected essays on the economics of coral reefs". Kalmar, Sweden: CORDIO, Kalmar University: 215-227.
	Direct market pricing	Barbier, E.B. (2007) Valuing ecosystem services as productive inputs. Economic Policy 22(1): 177-229.
	Factor Income / Production Function	Do, T.N. and J. Bennett (2005) An economic valuation of wetlands in Vietnam's Mekong Delta: a case study of direct use values in Camau Province. Occasional Paper No. 8. Environment Management and Development Program, APSEG, ANU.
	Direct market pricing	Janssen, R. and J.E. Padilla (1999) Preservation or Conversion? Valuation and evaluation of a mangrove forest in the Philippines. Environmental and Resource Economics 14(3): 297-331.
	Benefit Transfer	Gren, I.M. and T. Soderqvist (1994) Economic valuation of wetlands: a survey. Beijer International Institute of Ecological Economics. Beijer Discussion Paper series No. 54, Stockholm, Sweden.
	Direct market pricing	Ruitenbeek, H.J. (1988) Social cost-benefit analysis of the Korup Project, Cameroon. WWF for Nature Publication, London, UK.
	Direct market pricing	Turpie, J., B. Smith, L. Emerton and J. Barnes (1999) Economic value of the Zambezi Basin Wetlands. Zambezi Basin Wetlands conservation and resource utilization project. IUCN Regional Office for Southern Africa.

	Direct market pricing	Turpie, J., B. Smith, L. Emerton and J. Barnes (1999) Economic value of the Zambezi Basin Wetlands. Zambezi Basin Wetlands conservation and resource utilization project. IUCN Regional Office for Southern Africa.
	Direct market pricing	Turpie, J., B. Smith, L. Emerton and J. Barnes (1999) Economic value of the Zambezi Basin Wetlands. Zambezi Basin Wetlands conservation and resource utilization project. IUCN Regional Office for Southern Africa.
	Direct market pricing	Bann, C. (1997) An economic analysis of alternative mangrove management strategies in Koh Kong Province, Cambodia. Economy and Environment Program for Southeast Asia (EEPSEA research report series), International Development Research Centre.
	Direct market pricing	Tri, N.H. (2002) Valuation of the mangrove ecosystem in Can Gio mangrove biosphere reserve, Vietnam. The Vietnam MAB National Committee, UNESCO / MAB.
	Benefit Transfer	Do, T.N. and J. Bennett (2005) An economic valuation of wetlands in Vietnam's Mekong Delta: a case study of direct use values in Camau Province. Occasional Paper No. 8. Environment Management and Development Program, APSEG, ANU.
Medical/Bioprospecting	Direct market pricing	Emerton, L., R. Seilava and H. Pearith (2002) Bokor, Kirirom, Kep and Ream National Parks, Cambodia: Case Studies of Economic and Development Linkages. Field Study Report. International Centre for Environmental Management, Brisbane and IUCN.
	Contingent Valuation	MANR (2002) Valoracion economica del humedal barrancones. Proyecto Regional de Conservación de los Ecosistemas Costeros del Golfo de Fonseca –PROGOLF.
Fibbers	Direct market pricing	Turpie, J., B. Smith, L. Emerton and J. Barnes (1999) Economic value of the Zambezi Basin Wetlands. Zambezi Basin Wetlands conservation and resource utilization project. IUCN Regional Office for Southern Africa.
Fodder	Direct market pricing	Khalil, S. (1999) Economic valuation of the mangrove ecosystem along the Karachi coastal areas. In: Hecht, J. (ed), "The Economic Value of the Environment: Cases from South Asia". Washington, D.C., IUCN - The World Conservation Union.
Sand, rock, gravel.	Direct market pricing	Turpie, J., B. Smith, L. Emerton and J. Barnes (1999) Economic value of the Zambezi Basin Wetlands. Zambezi Basin Wetlands conservation and resource utilization project. IUCN Regional Office for Southern Africa.
Coral	Direct market pricing	Bann, C. (1997) An economic analysis of alternative mangrove management strategies in Koh Kong Province, Cambodia. Economy and Environment Program for Southeast Asia (EEPSEA research report series), International Development Research Centre.
Timber	Direct market pricing	Turpie, J.K. (2000) The use and value of natural resources of the Rufiji Floodplain and Delta, Tanzania. Rufiji Environmental Managemet Project, Technical report No. 17.
	Direct market pricing	Gunawardena, M. and J.S. Rowan (2005) Economic valuation of a mangrove ecosystem threatened by shrimp aquaculture in Sri Lanka. Environmental Management 36(4): 535-550.

	Benefit Transfer	Do, T.N. and J. Bennett (2005) An economic valuation of wetlands in Vietnam's Mekong Delta: a case study of direct use values in Camau Province. Occasional Paper No. 8. Environment Management and Development Program, APSEG, ANU.
	Benefit Transfer	Do, T.N. and J. Bennett (2005) An economic valuation of wetlands in Vietnam's Mekong Delta: a case study of direct use values in Camau Province. Occasional Paper No. 8. Environment Management and Development Program, APSEG, ANU.
	Direct market pricing	Janssen, R. and J.E. Padilla (1999) Preservation or Conversion? Valuation and evaluation of a mangrove forest in the Philippines. Environmental and Resource Economics 14(3): 297-331.
	Benefit Transfer	Gren, I.M. and T. Soderqvist (1994) Economic valuation of wetlands: a survey. Beijer International Institute of Ecological Economics. Beijer Discussion Paper series No. 54, Stockholm, Sweden.
	Benefit Transfer	Gren, I.M. and T. Soderqvist (1994) Economic valuation of wetlands: a survey. Beijer International Institute of Ecological Economics. Beijer Discussion Paper series No. 54, Stockholm, Sweden.
	Avoided Cost	Sathirathai, S. (1998) Economic valuation of mangroves and the roles of local communities in the conservation of natural resources: case study of Surat Thani, South Thailand. Unpublished report, EEPSEA research report series, Singapore.
	Direct market pricing	Tri, N.H. (2002) Valuation of the mangrove ecosystem in Can Gio mangrove biosphere reserve, Vietnam. The Vietnam MAB National Committee, UNESCO / MAB.
	Direct market pricing	Turpie, J.K. (2000) The use and value of natural resources of the Rufiji Floodplain and Delta, Tanzania. Rufiji Environmental Managemet Project, Technical report No. 17.
	Direct market pricing	White, A.T., M. Ross and M. Flores (2000) Benefits and costs of coral reef and wetland management, Olango Island, Philippines. In: Cesar, H. (ed), "Collected essays on the economics of coral reefs". Kalmar, Sweden: CORDIO, Kalmar University: 215-227.
Fuelwood and charcoal	Direct market pricing	Bann, C. (1997) An economic analysis of alternative mangrove management strategies in Koh Kong Province, Cambodia. Economy and Environment Program for Southeast Asia (EEPSEA research report series), International Development Research Centre.
	Benefit Transfer	Gren, I.M. and T. Soderqvist (1994) Economic valuation of wetlands: a survey. Beijer International Institute of Ecological Economics. Beijer Discussion Paper series No. 54, Stockholm, Sweden.
	Direct market pricing	Khalil, S. (1999) Economic valuation of the mangrove ecosystem along the Karachi coastal areas. In: Hecht, J. (ed), "The Economic Value of the Environment: Cases from South Asia". Washington, D.C., IUCN - The World Conservation Union.
	Direct market pricing	Do, T.N. and J. Bennett (2005) An economic valuation of wetlands in Vietnam's Mekong Delta: a case study of direct use values in Camau Province. Occasional Paper No. 8. Environment Management and Development Program, APSEG, ANU.

	Direct market pricing	Turner, R.K., J. Paavola, P. Cooper, S. Farber, V. Jessamy and S. Georgious (2003) Valuing nature: lessons learned and future research directions. Ecological Economics 46(3): 493-510.
	Benefit Transfer	Do, T.N. and J. Bennett (2005) An economic valuation of wetlands in Vietnam's Mekong Delta: a case study of direct use values in Camau Province. Occasional Paper No. 8. Environment Management and Development Program, APSEG, ANU.
	Direct market pricing	Christensen, B. (1982) Management and utilisation of mangroves in Asia and the Pacific. FAO, Rome. Environment Paper No. 3. Food and Agriculture Organization of the United Nations, Rome, Italy.
Other raw materials	Direct market pricing	Turpie, J., B. Smith, L. Emerton and J. Barnes (1999) Economic value of the Zambezi Basin Wetlands. Zambezi Basin Wetlands conservation and resource utilization project. IUCN Regional Office for Southern Africa.
	Factor Income / Production Function	Ahmad, N. (1984) Some aspects of economic resources of Sundarban mangrove forest of Bangladesh.
	Direct market pricing	Nickerson, D.J. (1999) Trade-offs of mangrove area development in the Philippines. Ecological Economics 28 (2): 279-298.
	Direct market pricing	Tri, N.H. (2002) Valuation of the mangrove ecosystem in Can Gio mangrove biosphere reserve, Vietnam. The Vietnam MAB National Committee, UNESCO / MAB.
Regulating services		
	Avoided Cost	Emerton, L (ed) (2005) Values and rewards: counting and capturing ecosystem water services for sustainable development. IUCN Water, Nature and Economics Technical Paper No. 1, IUCN — The World Conservation Union, Ecosystems and Livelihoods Group Asia.
Climate regulation	Benefit Transfer	Turpie, J.K. (2000) The use and value of natural resources of the Rufiji Floodplain and Delta, Tanzania. Rufiji Environmental Managemet Project, Technical report No. 17.
	Replacement Cost	Sathirathai, S. (1998) Economic valuation of mangroves and the roles of local communities in the conservation of natural resources: case study of Surat Thani, South Thailand. Unpublished report, EEPSEA research report series, Singapore.
	Benefit Transfer	Cesar, H. and C.K. Chong (2004) Economic valuation and socioeconomics of coral feefs: methodological issues and three case studies. Wildfish Center Contribution No. 1721.
Erosion prevention	Avoided Cost	Emerton, L (ed) (2005) Values and rewards: counting and capturing ecosystem water services for sustainable development. IUCN Water, Nature and Economics Technical Paper No. 1, IUCN — The World Conservation Union, Ecosystems and Livelihoods Group Asia.
	Direct market pricing	Ruitenbeek, H.J. (1994) (1994) Modelling economy-ecology linkages in mangroves: Economic evidence for promoting conservation in Bintuni Bay, Indonesia. Ecological Economics 10(3): 233-247

	Replacement Cost	Samonte-Tan, G.P.B., A. T. White, M. A. Tercero, J. Diviva, E. Tabara and C. Caballes (2007) Economic Valuation of Coastal and Marine Resources: Bohol Marine Triangle, Philippines. Costal Management 35(2): 319-338.
	Avoided Cost	Badola, R.and S.A. Hussain (2005) Valuing ecosystem functions: an empirical study on the storm protection function of Bhitarkanika mangrove ecosystem, India. Environmental conservation 32(1): 85-92.
	Avoided Cost	Emerton, L (ed) (2005) Values and rewards: counting and capturing ecosystem water services for sustainable development. IUCN Water, Nature and Economics Technical Paper No. 1, IUCN — The World Conservation Union, Ecosystems and Livelihoods Group Asia.
	Avoided Cost	Cooper, E., L. Burke and N. Bood (2009) Coastal capital : Belize - The economic contribution of Belize's coral reefs and mangroves. WRI Working Paper. World Resources Institute, Washington, D.C., 53pp.
Protection against extreme events	Replacement Cost	Gunawardena, M. and J.S. Rowan (2005) Economic valuation of a mangrove ecosystem threatened by shrimp aquaculture in Sri Lanka. Environmental Management 36(4): 535-550.
	Replacement Cost	Barbier, E.B. (2007) Valuing ecosystem services as productive inputs. Economic Policy 22(1): 177-229.
	Contingent Valuation	Bann, C. (1999) A contingent valuation of the mangroves of Benut, Johor State, Malaysia. Report to DANCED, Copenhagen, Denmark.
	Replacement Cost	Sathirathai, S. (1998) Economic valuation of mangroves and the roles of local communities in the conservation of natural resources: case study of Surat Thani, South Thailand. Unpublished report, EEPSEA research report series, Singapore.
	Replacement Cost	Barbier, E.B., I. Strand and S. Sathirathai (2002) Do open access Conditions affect the valuation of an externality? Estimating the welfare effects of Mangrove-Fishery Linkages in Thailand. Environmental and Resource Economics 21(4): 343-367.
Cultural services		
	Direct market pricing	Cooper, E., L. Burke and N. Bood (2009) Coastal capital : Belize - The economic contribution of Belize's coral reefs and mangroves. WRI Working Paper. World Resources Institute, Washington, D.C., 53pp.
Recreation/tourism	Contingent Valuation	Ammour, T., N. Windervoxhel and G. Sencion (2000) Economic valuation of mangrove ecosystems and sub-tropical forests in Central America. In: Dore M. and R. Guevara (ed), "Sustainable Forest management and Global Climate Change". Edward Elgar Publishing, UK.
	Travel Cost	Tri, N.H. (2002) Valuation of the mangrove ecosystem in Can Gio mangrove biosphere reserve, Vietnam. The Vietnam MAB National Committee, UNESCO / MAB.
Supporting services		
Biodiversity protection	Benefit Transfer	Samonte-Tan, G.P.B., A. T. White, M. A. Tercero, J. Diviva, E. Tabara and C. Caballes (2007) Economic Valuation of Coastal and Marine Resources: Bohol Marine Triangle, Philippines. Costal Management 35(2): 319-338.

	Contingent Valuation	Gunawardena, M. and J.S. Rowan (2005) Economic valuation of a mangrove ecosystem threatened by shrimp aquaculture in Sri Lanka. Environmental Management 36(4): 535-550.
	Contingent Valuation	Bann, C. (1999) A contingent valuation of the mangroves of Benut, Johor State, Malaysia. Report to DANCED, Copenhagen, Denmark.
	Contingent Valuation	Bann, C. (1999) A contingent valuation of the mangroves of Benut, Johor State, Malaysia. Report to DANCED, Copenhagen, Denmark.
	Mitigation and Restoration Cost	Tri, N.H. (2002) Valuation of the mangrove ecosystem in Can Gio mangrove biosphere reserve, Vietnam. The Vietnam MAB National Committee, UNESCO / MAB.
	Direct market pricing	Turpie, J.K. (2000) The use and value of natural resources of the Rufiji Floodplain and Delta, Tanzania. Rufiji Environmental Managemet Project, Technical report No. 17.
	Factor Income / Production Function	Samonte-Tan, G.P.B., A. T. White, M. A. Tercero, J. Diviva, E. Tabara and C. Caballes (2007) Economic Valuation of Coastal and Marine Resources: Bohol Marine Triangle, Philippines. Costal Management 35(2): 319-338.
	Contingent Valuation	Bann, C. (1999) A contingent valuation of the mangroves of Benut, Johor State, Malaysia. Report to DANCED, Copenhagen, Denmark.
	Direct market pricing	Janssen, R. and J.E. Padilla (1999) Preservation or Conversion? Valuation and evaluation of a mangrove forest in the Philippines. Environmental and Resource Economics 14(3): 297-331.
	Direct market pricing	Christensen, B. (1982) Management and utilisation of mangroves in Asia and the Pacific. FAO, Rome. Environment Paper No. 3. Food and Agriculture Organization of the United Nations, Rome, Italy.
Nursery	Direct market pricing	Christensen, B. (1982) Management and utilisation of mangroves in Asia and the Pacific. FAO, Rome. Environment Paper No. 3. Food and Agriculture Organization of the United Nations, Rome, Italy.
	Direct market pricing	Christensen, B. (1982) Management and utilisation of mangroves in Asia and the Pacific. FAO, Rome. Environment Paper No. 3. Food and Agriculture Organization of the United Nations, Rome, Italy.
	Factor Income / Production Function	Sathirathai, S. (1998) Economic valuation of mangroves and the roles of local communities in the conservation of natural resources: case study of Surat Thani, South Thailand. Unpublished report, EEPSEA research report series, Singapore.
	Direct market pricing	Burbridge, P.R. and Koesoebiono (1984) Management of mangrove exploitation in Indonesia. In: Soepadmo, E., A.N. Rao and D.J. Macintosh (ed), "Proceedings Asian Symposium on Mangrove Environment: Research and Management". Kuala Lumpur, 25-29 Aug. 1980. University of Malaya and UNESCO.
	Direct market pricing	Lal, P.N. (1990) Conservation or conversion of mangroves in Fiji. East-West Centre Occasional Papers 11
	Factor Income / Production Function	Levine, S. and M. Mindedal (1998) Economics of multiple-use natural resources: the mangroves of Vietnam. MSc Thesis, University of Copenhagen

Direct market pricing	Morton, R.M. (1990) Community structure, density, and standing crop of fishes in a subtropical Australian mangrove area. Marine Biology 105: 385-394.
Direct market pricing	Nickerson, D.J. (1999) Trade-offs of mangrove area development in the Philippines. Ecological Economics 28 (2): 279-298.
Factor Income / Production Function	Barbier, E.B., I. Strand and S. Sathirathai (2002) Do open access Conditions affect the valuation of an externality? Estimating the welfare effects of Mangrove-Fishery Linkages in Thailand. Environmental and Resource Economics 21(4): 343-367.

Appendix 3. Statistics on artisanal fisheries from the Gulf of Nicoya

Table A3.1. Total catch (kg) in the Gulf of Nicoya by artisanal fisheries for 2015. Data is aggregated by commercial category.

Commercial category	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Total
First large	13,253	24,203	18,507	19,477	7,112	2,174	1,807	24,183	25,321	17,555	11,613	13,842	179,047
First small	46,844	45,461	56,887	35,964	9,564	3,825	3,062	36,013	55,640	38,722	40,682	33,423	406,087
Class	35,865	32,767	40,488	42,621	19,439	11,043	22,107	26,693	41,291	25,213	20,742	32,420	350,689
Junk	42,572	36,262	44,691	43,398	20,534	14,092	13,534	27,686	36,740	29,480	31,024	26,350	366,363
Golden croaker tail	8,535	6,756	12,425	10,629	6,731	4,031	3,565	10,243	9,886	11,855	10,748	6,708	102,112
Coral Hawk fish	608	1,528	1,064	387	1,667	2,798	4,158	3,442	3,058	2,180	4,034	520	25,444
Snapper	6,136	4,022	7,175	4,787	1,978	2,709	2,419	1,571	4,645	2,237	1,547	3,813	43,039
Spotted rose snapper	6,112	5,420	6,838	11,067	9,473	5,641	3,478	5,341	6,916	4,775	6,464	2,199	73,724
Pacific red snapper	19	230	58	355	192	979	302	136	71	600	0	714	3,656
Mahi Mahi	12,495	1,319	44	402	1,294	101	39	10	1,794	7,148	855	0	25,501
Marlin	0	0	0	0	0	0	0	0	0	0	0	0	0
White marlin	0	0	0	0	0	0	0	0	0	0	0	0	0
Striped Marlin	0	0	0	0	0	0	0	0	0	0	0	0	0
Sailfish	84	0	76	95	0	144	118	0	107	840	59	0	1,523
Sword fish	47	0	0	0	0	1,134	0	0	0	0	0	0	1,181
Wahoo	0	0	0	0	0	0	0	0	0	0	0	0	0
Sardine	64	0	775	1,025	0	0	150	0	0	0	0	0	2,014
Tuna	14,540	26,452	24,157	30,394	12,282	5,969	23,982	18,215	18,300	16,981	9,890	16,038	217,200
Ballywoo	0	0	0	0	0	0	0	0	0	0	0	0	0
Cazon	287	935	1,064	960	747	575	941	758	490	486	344	232	7,819
Posta	0	0	45	0	282	0	67	154	118	25	0	0	691

Maco	0	0	0	0	0	0	0	0	0	0	0	0	0
Treacher	0	0	0	0	0	0	0	0	118	122	0	0	240
White shrimp	12,346	10,008	6,560	6,669	1,711	60	5	10,233	7,104	5,325	4,204	6,916	71,141
Brown shrimp	0	0	0	0	0	0	0	0	0	0	0	0	0
Pink shrimp	0	0	0	0	0	0	0	0	0	0	0	0	0
Kolibri shrimp	0	0	0	0	0	0	0	0	0	0	0	0	0
Northern nylon shrimp	0	0	0	0	0	0	0	0	0	0	0	0	0
Royal shrimp	0	0	0	0	0	0	0	0	0	0	0	0	0
Atlantic seabob	55	22	14	9	2	0	0	0	13	0	1	1	117
Prawn	0	0	0	0	0	0	0	0	0	0	0	0	0
Pacific prawn	128	23	80	40	4	0	8	10	30	34	0	56	413
Caribbean prawn	0	0	0	0	0	0	0	0	0	0	0	0	0
Squid	0	0	8	0	0	0	0	0	359	334	0	5	706
Octopus	0	0	0	0	0	0	0	0	0	0	0	0	0
Bivalves	728	2,488	2,327	1,581	1,107	0	0	6,096	5,394	6,335	7,148	2,899	36,103
Cambute	0	0	0	0	0	0	0	0	0	0	0	0	0
Shark fin	0	0	0	0	0	0	0	0	0	0	0	0	0
Filet	86	100	578	89	0	0	0	19	104	184	104	124	1,388
Buche	2	4	12	5	23	0	0	0	0	0	0	0	46
Crab	1,590	2,310	1,418	20	8	0	0	0	43	3,102	2,598	58	11,147
Turtles	0	0	0	0	0	0	0	0	0	0	0	0	0

Source: INCOPESCA, 2018

Table A3.2. The commercial category "Class" disaggregated by species, showing each species catch (kg) and value (2015 USD) in the Gulf of Nicoya in 2015

Species	% Sp	Catch (kg)	Value (USD)
Scalyfin corvina	43	151,363	286,431
Sharpnose corvina	14	48,210	91,230
Queen corvina	14	48,141	91,098
Armed snook	9	31,498	59,605
Union snook	5	16,411	31,054
Pacific smalleye croaker	4	12,757	24,141
Atlantic Tripletail	3	8,791	16,636
Flathead Mullet	2	7,878	14,908
Blackfin snook	2	7,073	13,385
Panama kingcroaker	1	3,924	7,425
Yellowfin corvina	1	3,451	6,530
Snook	1	3,075	5,818
Sea catfish	1	2,782	5,265
Golden croaker	1	2,128	4,027
Barracuda	0.4	1,546	2,925
Striped corvina	0.3	1,224	2,316
Highfin king croaker	0.1	438	829
Total	100	**350,689**	**663,623**

Source: Marín, 2018

Table A3.3. The commercial category "First small" disaggregated by species, showing each species catch (kg) and value (2015 USD) in the Gulf of Nicoya in 2015

Species	% Sp	Catch (kg)	Value (USD)
Scalyfin corvina	29	116,528	423,387
Queen corvina	29	115,991	421,434
Golden croaker	17	69,510	252,552
Sharpnose corvina	14	57,847	210,178
Snook	4	16,726	60,773
Blackfin snook	2	9,301	33,795
Armed snook	2	8,926	32,431
Yellowfin corvina	2	6,832	24,824
Pacific smalleye croaker	0.5	1,864	6,771
Striped corvina	0.4	1,535	5,576

Panama kingcroaker	0.2	708	2,572
Union snook	0.1	318	1,157
Total	100	**406,087**	**1,475,451**

Source: Marín, 2018

Table A3.4. The commercial category "First large" disaggregated by species, showing each species catch (kg) and value (2015 USD) in the Gulf of Nicoya in 2015

Species	% Sp	Catch (kg)	Value (USD)
Queen corvina	82	147,640	752,226
Snook	15	26,849	136,797
Yellowfin corvina	3	4,558	23,224
Total	100	**179,047**	**912,247**

Source: Marín, 2018

Table A3.5. The commercial category "Bivalves" disaggregated by species, showing each species catch (kg) and value (2015 USD) in the Gulf of Nicoya in 2015

Bivalve	Capture (kg)
Clams	25,090
Piangua	5,556
Chora	3,296
Mussels	2,135
Total	**36,077**

Source: Duran, 2018

Table A3.6. List of species that were selected to be valued in this study, indicating its scientific name and family, as well as the references that indicate that they use mangroves during their life cycle.

Common name (English)	Scientific name	Family	Reference
Scalyfin corvina, Weakfish	*Cynoscion squamipinnis*	Sciaenidae	Rönnbäck, 1999
Cachema weakfish, Sharpnose corvina	*Cynoscion phoxocephalus*	Sciaenidae	Rönnbäck, 1999
Queen corvina, Whitefin weakfish	*Cynoscion albus*	Sciaenidae	Rönnbäck, 1999
Golden croaker, Tallfin croaker	*Micropogonias altipinnis*	Sciaenidae	Rönnbäck, 1999
Striped corvina, Striped weakfish	*Cynoscion reticulatus*	Sciaenidae	Rönnbäck, 1999
Berrugato real, Highfin king croaker	*Menticirrhus nasus*	Sciaenidae	Rönnbäck, 1999
Pacific smalleye croaker	*Nebris occidentalis*	Sciaenidae	Rönnbäck, 1999
Panama kingcroaker	*Menticirrhus panamensis*	Sciaenidae	Rönnbäck, 1999
Stolzmann's weakfish, Yellowfin corvina	*Cynoscion stolzmanni*	Sciaenidae	Rönnbäck, 1999
Armed snook	*Centropomus armatus*	Centropomidae	Rönnbäck, 1999
Union snook	*Centropomus unionensis*	Centropomidae	Rönnbäck, 1999
Blackfin snook, Pacific blackfin	*Centropomus medius*	Centropomidae	Rönnbäck, 1999
Flathead Mullet, Black True Mullet	*Mugil cephalus*	Mugilidae	Rönnbäck, 1999
Atlantic Tripletail	*Lobotes surinamensis*	Lobotidae	Carpenter 2001
Black Robalo, Black Snook	*Centropomus nigrescens*	Centropomidae	Rönnbäck, 1999
White snook	*Centropomus viridis*	Centropomidae	Rönnbäck, 1999
Sea catfish	*Notarius troscheli*	Ariidae	Rönnbäck, 1999
Mexican Barracuda, Barracuda	*Sphyraena ensis*	Sphyraenidae	Rönnbäck, 1999
Spotted rose snapper	*Lutjanus guttatus*	Lutjanidae	Rönnbäck, 1999
White shrimp	*Litopenaeus*	Penaeidae	Rönnbäck, 1999; Goti 1991
Piangua	*Anadara multicostata*	Arcidae	Morton 2013
Piangua	*Anadara similis*	Arcidae	Morton 2013
Piangua	*Anadara tuberculosa*	Arcidae	Morton 2013
Rockmussel	*Modiolus capax*	Arcidae	Morton 2013
Chucheca	*Grandiarca grandis*	Arcidae	Morton 2013
Chora mussel	*Mytella guyanensis*	Mytilidae	Morton 2013
Clam	*Polymesoda inflata*	Corbiculidae	Morton 2013
Green clam	*Polymesoda radiata*	Corbiculidae	Morton 2013
Sandy clam	*Donax californicus*	Donacidae	Morton 2013
White clam	*Leukoma asperrima*	Veneridae	Morton 2013
White clam	*Protothaca grata*	Veneridae	Morton 2013

Appendix 4. Output maps generated by the INVEST Coastal Vulnerability Model.

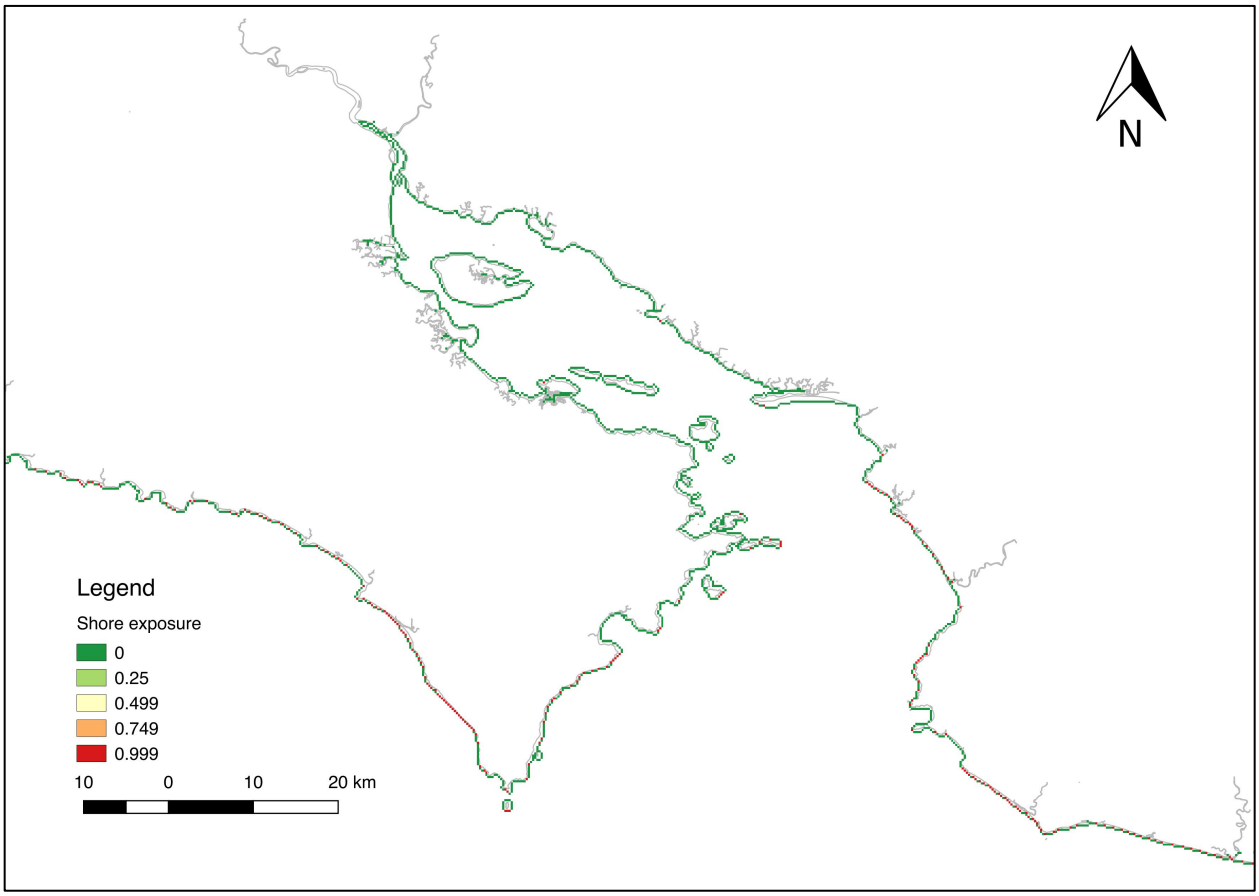


Figure A4.1. Shore exposure, a raster where the cells corresponding to the shoreline segments are either 0 if sheltered or 1 if exposed.

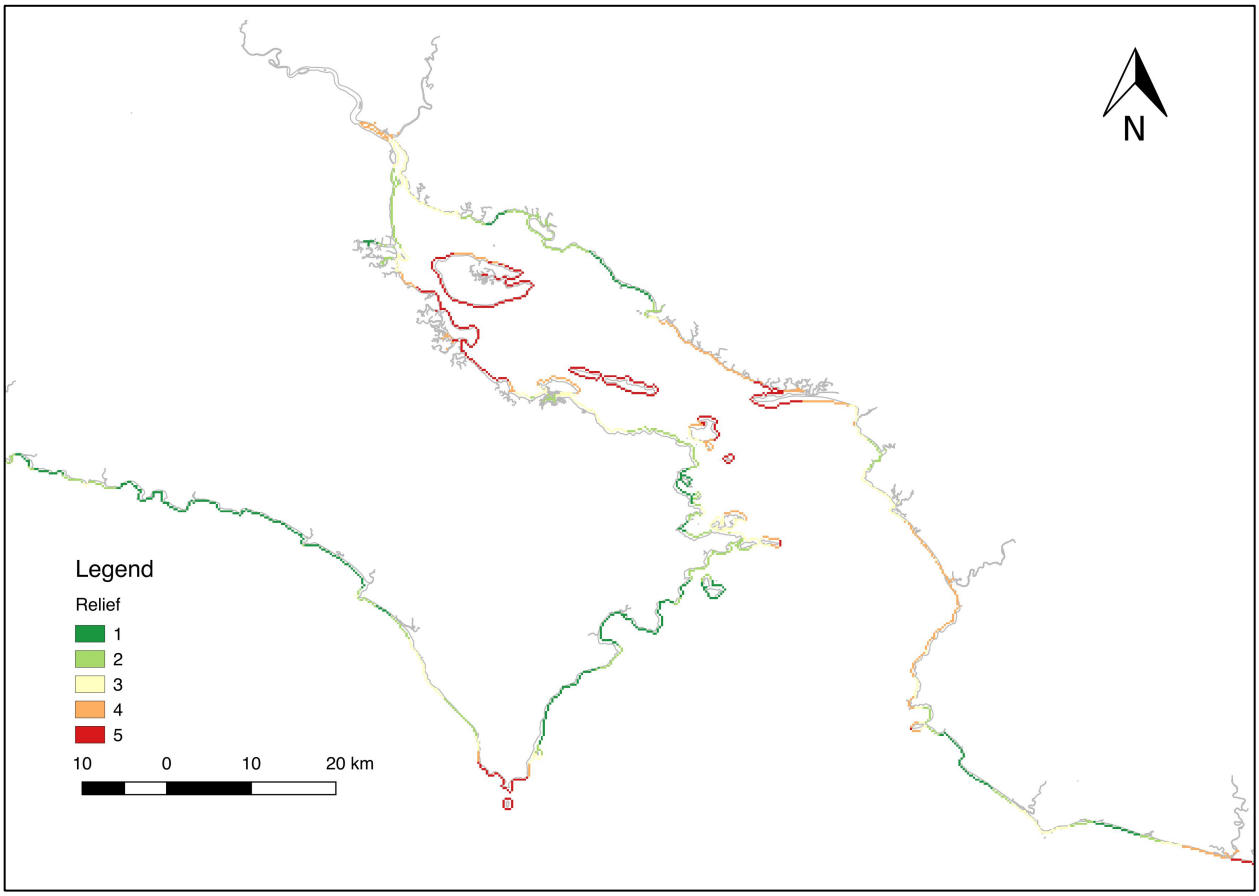


Figure A4.2. Relief, a raster where shore segments are valued from 1 to 5 depending on the average elevation around that cell. Lower values indicate lower elevations.

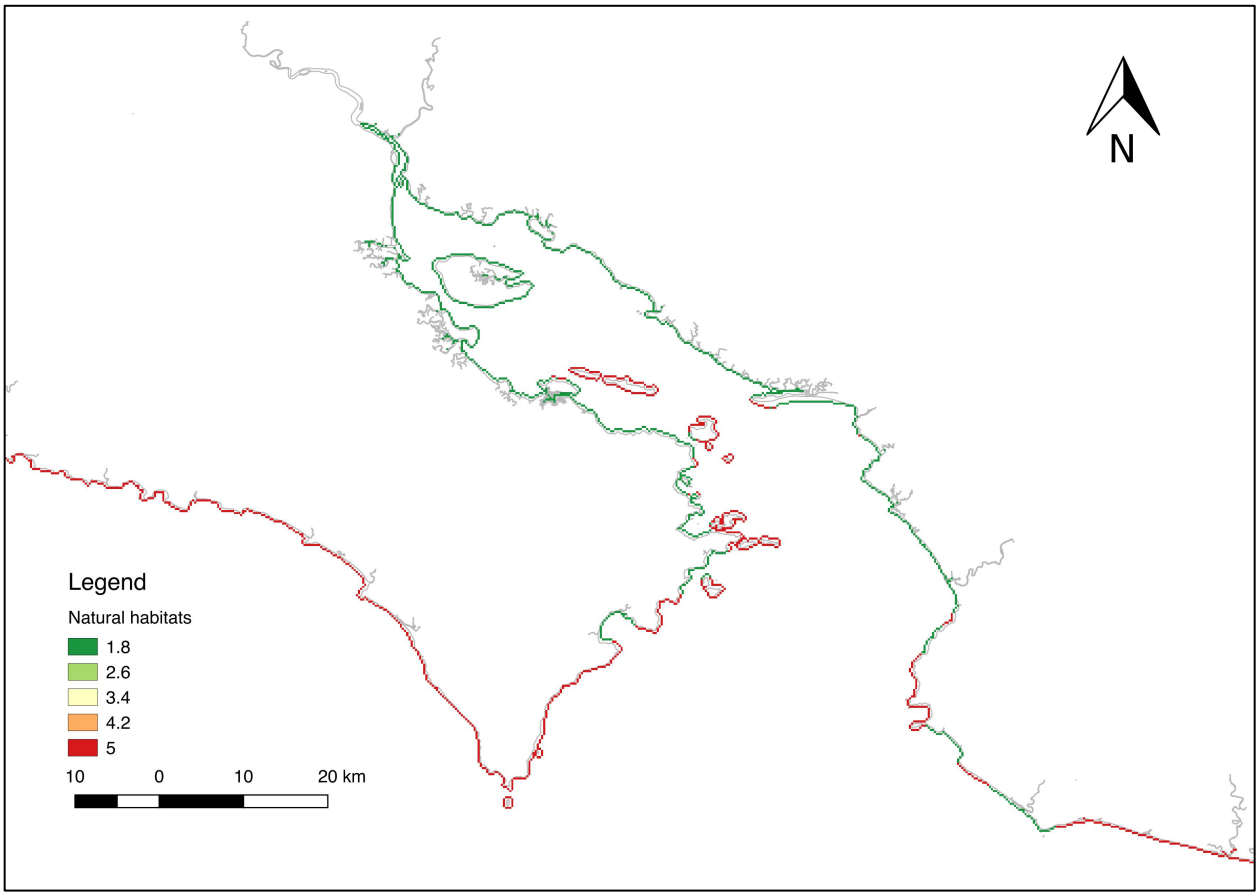


Figure A4.3. Natural habitats, a raster where shore segments are valued according to the natural habitats that are present there, which in this case are all mangroves.

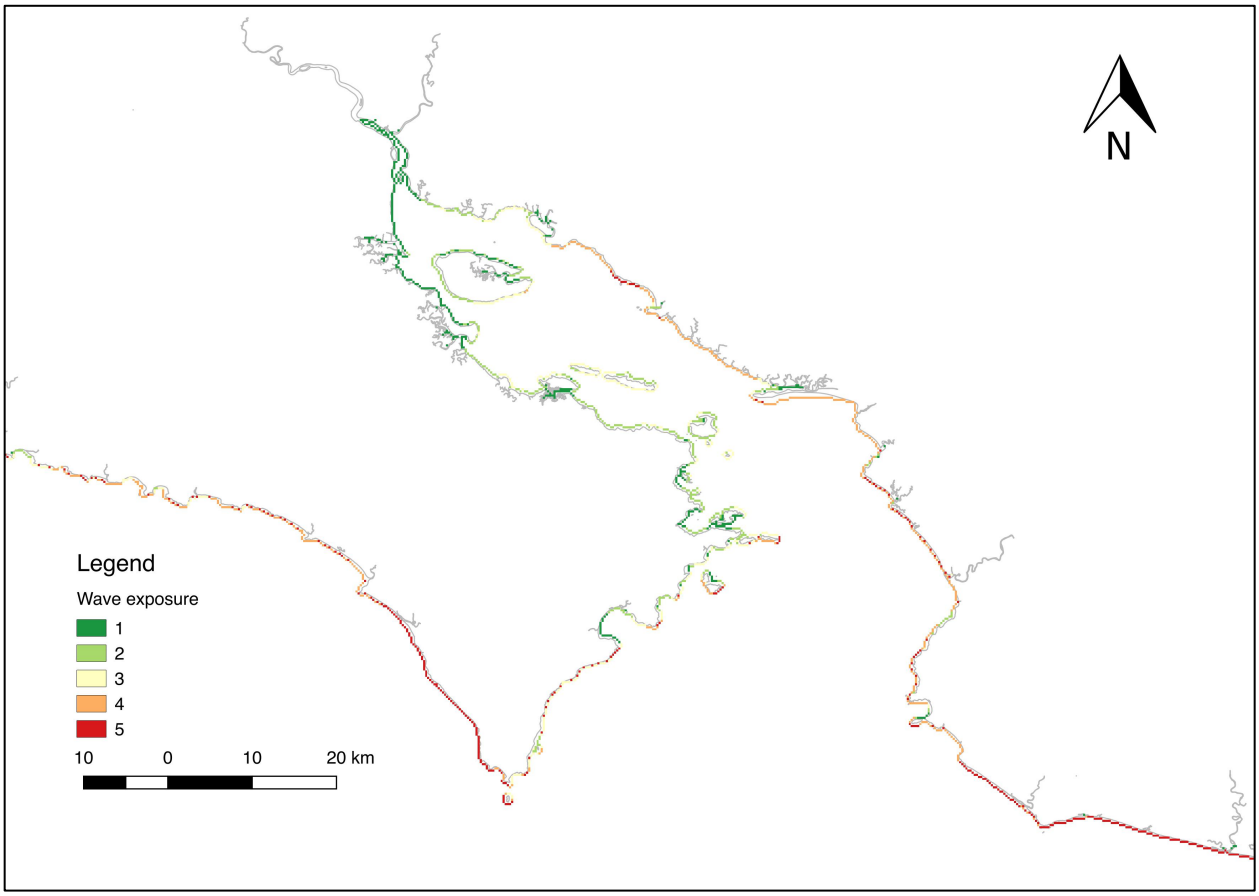


Figure A4.4. Wave exposure, a raster where shore segments are ranked in a similar way to wind exposure, but according to their exposure to wave.

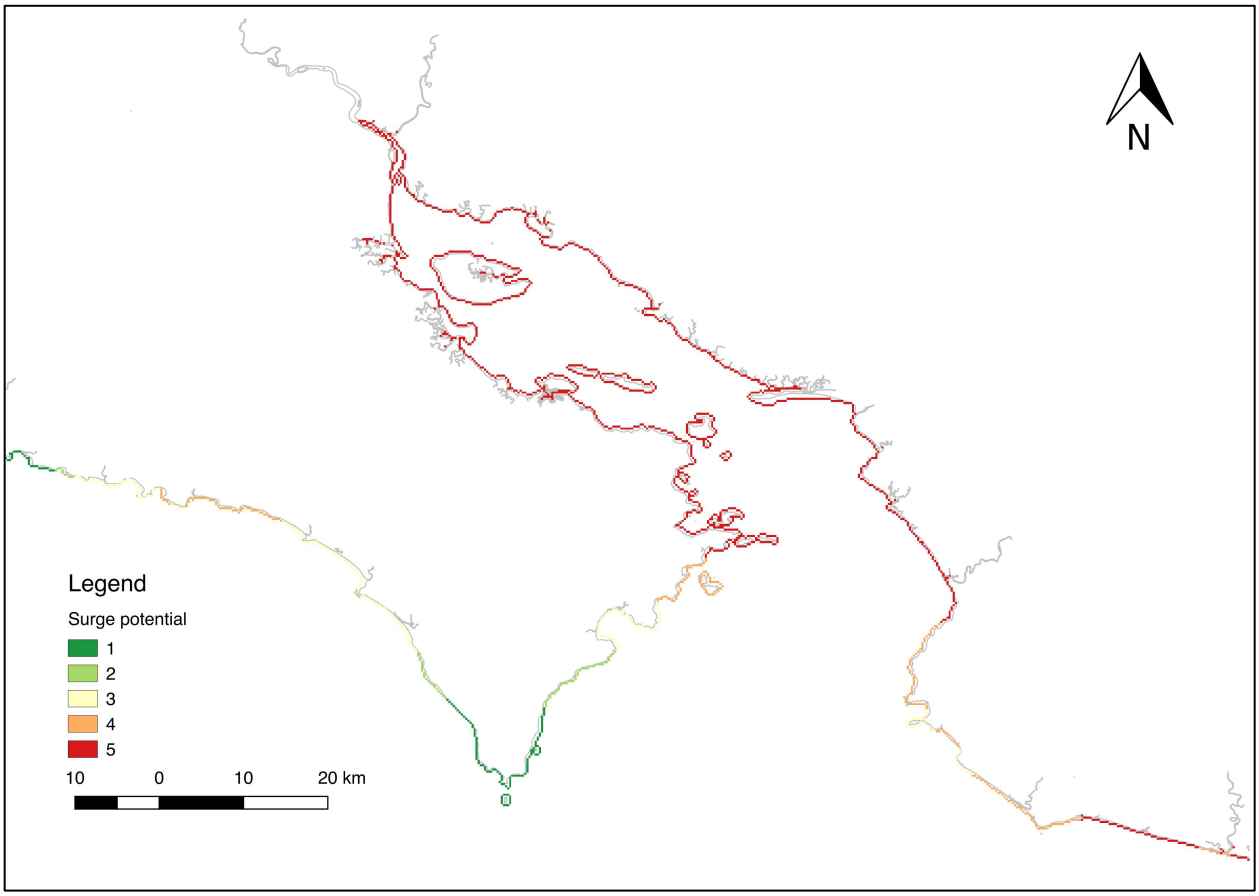


Figure A4.5. Surge potential. a raster where segments are ranked according to their exposure to potential surge. First, the exposed segments are assigned a rank in equal proportion between 1 and 5, depending on their distance to the edge of the continental shelf. Then, these values are propagated along the sheltered coast. Isolated coastline segments (such as islands) are assigned the rank of the closest (already ranked) segment.

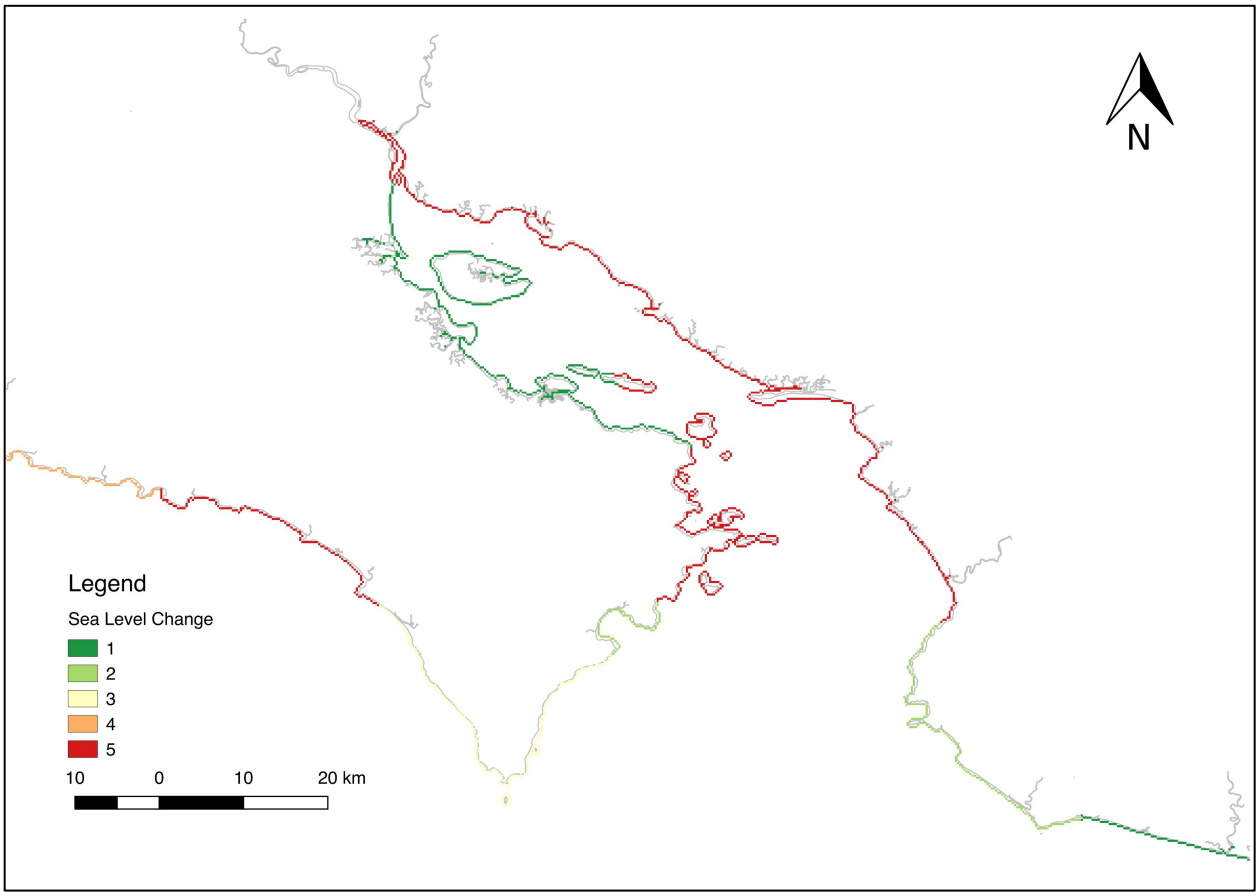


Figure A4.6. Sea level change, a raster with segments ranked in equal proportion between 1 and 5 based on the sea level rise value from the input shapefile.

A research and policy agenda for establishing a payment for ecosystem services scheme for the conservation and restoration of marine and coastal ecosystems in Costa Rica

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Abstract

Costa Rica has ten times more marine area than terrestrial area, with a variety of marine and coastal ecosystems that provide valuable goods and services to society at many scales. Nevertheless, these ecosystems have been significantly degraded by unsustainable economic activities mainly because of the public good nature of marine and coastal ecosystem services. They are undervalued or not valued at all under current markets, and lack the appropriate institutions and mechanisms to internalize their value in order to better manage them. As a solution to this research and policy gap, I propose here the creation of the Blue Fund, a new PES scheme focused on marine and coastal ecosystems with the goal of conserving, enhancing and restoring them. To build this new scheme, I propose a six-step process: 1) ecosystem assessment, 2) ecosystem services selection and valuation, 3) threats identification and prioritization, 4) creation of funding sources and investment, 5) implementation of conservation and restoration projects, and 6) evaluation and adaptation. In each of these steps, examples of how they can be developed are provided with a focus on mangroves and coral reefs, although the method can be applied to any ecosystem. The Blue Fund proposed here can also be a sub-fund of a broader institution and financial mechanism that I have named the Natural Capital Fund, recognizing the linkages between ecosystems and the need to increase the productivity of these types of funds by unifying loose initiatives into one that can be better managed.

Key words: natural capital, ecosystem services, payment for ecosystem services, coral reefs, mangroves

Introduction

Costa Rica is more than a green country. It is a blue one, with more than ten times more marine area (589,000 km^2) than continental area (51,900 km^2), and a shoreline of 1,254 km in the Pacific and 212 km in the Caribbean. It has a variety of coastal and marine ecosystems such as sea grasses, coral reefs, mangrove forests, rocky shores, cliffs, estuaries, and different types of beaches (e.g. sandy, muddy, rocky) (Morales, 2013). Furthermore, Costa Rica has now more marine area protected than forest area, having established six Marine Protected Areas that together account for 15,836 km^2, which represents 53% of the total area protected under different mechanisms (e.g. National Parks accounts for 6,294 km^2, 21%) (Corrales, 2017).

Nevertheless, degradation of marine and coastal ecosystems has occurred mainly because many of their services are public goods. Markets fail to incorporate partially or entirely their economic value and institutions that can internalize their value are missing. A straightforward solution to this problem can be the creation of financial incentives capable of incorporating environmental externalities (positive externalities in the case of services provided by marine and coastal ecosystems), in which beneficiaries of these services pay ecosystem stewards to conserve, enhance or restore ecosystems in order to maintain the flow of services (Schomers & Matzdorf, 2013).

The creation of financial incentives for ecosystem services, generally called Payment for Ecosystem Services (PES), can be conceptualized from Coasean or Pigouvian perspectives. Perhaps the most widely used definition of PES, which is founded on the Coase theorem, is the one provided by Wunder (2005) based on the following 5 criteria: "1. a voluntary transaction where 2. a well-defined ecosystem service (or a land-use likely to secure that service) 3. is being bought by a (minimum one) ecosystem service buyer 4. from a (minimum one) ecosystem service provider 5. if and only if the ecosystem service provider secures ecosystem service provision (conditionality)". This definition has been criticized over the years for being restricted to market mechanisms, and in practice pure Coasean PES schemes (i.e. one established and funded through private voluntary negotiations) are limited to a small minority of all PES schemes described in the literature (Tacconi, 2012; Schomers & Matzdorf, 2013), one of the most clear examples is the

case of water bottling companies paying directly upstream farmers to implement good farming practices to maintain the quality of the water used by these companies.

Contrary to the Coasean approach, the Pigouvian one is characterized by the intervention of the government, acting as a third party between sellers and buyers, and applying financial mechanisms such as taxes for negative externalities and subsidies for positive externalities like marine and coastal ecosystem services (marine and costal ecosystem services). Pigouvian PES are widespread around the world, and they focus on the provision of public goods (Schomers & Matzdorf, 2013), which in the case of marine and costal ecosystem services, they are the great majority and therefore, this approach is perhaps the most appropriate.

To better take into consideration ecosystem services that are public goods in PES schemes, which cannot be managed entirely under a Coasean approach and therefore, they require the intervention of other actors, Muradian et al. (2010) propose the defining PES "as a transfer of resources between social actors, which aims to create incentives to align individual and/or collective land use decisions with the social interest in the management of natural resources", meaning that transfers can happen under a market or something similar, or through other financial mechanisms like incentives (not restricted to economic ones) or public subsidies. This definition has been considered more in line with ecological economics (Farley & Costanza, 2010). Tacconi (2012) propose a similar PES definition as an alternative to the marketed-centered one from Wunder (2005), in which "a PES scheme is a transparent system for the additional provision of environmental services through conditional payments to voluntary providers". Other alternative definitions to Wunder (2005) are proposed by, Porras et al. (2008), and Wunder (2015) who updated his 2005 definition in order to be broader in scope.

For the development of a Payment for Marine and Coastal Ecosystem Services, a hybrid concept of PES between the definition of Muradian et al. (2010) and the one from Tacconi (2012) might be the most useful, since the former is broad enough to let a more flexible and modern approach in the design of the financial mechanism, and the latter stating the importance of additionality and conditionality.

Developing a Payment for Marine and Coastal Ecosystem Services

In a recent study from Salzman et al. (2018) on the global state of PES schemes, the authors identified 555 active PES programs around the world, with combined annual payments over $36 billion. Furthermore, they determined that of those 555 PES schemes, 387 (70%) are watershed PES schemes, 120 (21%) biodiversity and habitat programs, and 48 (9%) forest and land-use carbon programs. None of the PES schemes identified in the study are designed for marine or coastal ecosystems, making evident a clear research and policy gap for the creation and implementation of financial incentives for these ecosystems.

To address this, I propose in this paper the creation of a new PES program for marine and coastal ecosystems, which I call the Blue Fund. The general goal of the Blue Fund is to change the behavior of any actor that has or can have a negative impact on marine and coastal ecosystems, and reward those actors that have a positive impact on these ecosystems, in order to protect, enhance or restore its ecosystem services. This financial mechanism is intended to be developed in combination or in addition to a stronger marine legislation, not instead of it. The first challenge, and opportunity, of establishing this PES scheme is that it is targeted to multiple ecosystems, which are public property, instead of only one as it is in the current PES scheme for privately owned forests. Although this Payment for Marine and Coastal Ecosystem Services can be implemented in several ecosystems, in this paper I chose coral reefs and mangroves to illustrate the general principles and steps to build such a scheme, two of the most important ecosystems in terms of the value of the benefits they provide to society (Mehvar et al., 2018).

To construct this fund, first, some key elements of any PES scheme should be described and specified under this context. Engel et al. (2008) mentions three main elements of any type of PES scheme: 1) the seller(s), 2) the buyer(s) and 3) the financial mechanism(s). According to the authors, the sellers of ecosystem services are the actors who are in a position of safeguarding the delivery of the ecosystem services, which in the vast majority of PES schemes are private landowners as in the case of the current PSA program of Costa Rica. Nevertheless, under the Blue Fund, there would be only one seller, the government, since it is the only owner of all marine and coastal ecosystems as stated in the environmental legislation of Costa Rica, creating, this way, a monopoly of marine and coastal ecosystem services.

Regarding the buyers, depending on the type of PES program, they can be the actual users of the ecosystem service as in the case of Coasean PES (i.e. user-financed), or the buyers can be others (e.g. the government or an NGO) that play a role of intermediary between the seller and the buyers, as in the case of Pigouvian PES (government-financed). In the current PES scheme of Costa Rica, the government acts as the intermediary through the National Fund for Forest Financing (FONAFIFO, by its initials in Spanish), and therefore, is the only buyer of the program, creating a monopsony. In the proposed Blue Fund, quite the opposite occurs, instead of only one buyer, there are many across different social and productive sectors in Costa Rica as well as overseas (Table 1).

Table 1. Difference in buyers and sellers in the current and proposed PES scheme.

	Forests PSA	**Blue Fund**
Buyer	Government	Many sectors
Seller	Private land owners	Government

The third key element of a PES program according to Engel et al. (2008) is the financial mechanism *per se*. Here, the type of activity must be defined, in most cases being a payment to providers for specific land uses that generate the desired ecosystem services. Furthermore, several questions must be answered, such as which activities are going to be funded, what the origin of the funds is going to be, how much to pay to those actors that have a positive impact on these ecosystems and how much to charge to those ones that have a negative impact, among others that are going to be addressed later in this section.

Six-steps method for a new PES scheme

Having described broadly the three core elements of a PES scheme and how they could operate under the Blue Fund, this section presents a general framework to design a Payment for Marine and Coastal Ecosystem Services, with a focus on mangroves and coral reefs, but the same

approach can be applied to any other marine and coastal ecosystem such as estuaries, open ocean, sea grasses, and beaches, among others.

In addition to these core elements, Farley & Costanza (2010) describe ten broader principles concerning the use of PES schemes, which are framed under "The Heredia Declaration on Payment for Ecosystem Services, “a consensus statement signed by international and local experts outlining the mechanisms for successfully implementing PES at the global, regional and local level”, product of a workshop organized in Heredia, Costa Rica, in March 2007. The ten principles are: 1) measurement, 2) bundling, 3) scale-matching, 4) property rights, 5) distribution issues, 6) sustainable funding, 7) adaptive management, 8) education and politics, 9) participation and 10) policy coherence.

These principles will be considered in the design of the proposed PES scheme here, but it is worth reflecting at this point on the tenth principle. Any PES scheme needs to form part of a coherent set of policies to address ecosystem use and management in order to be effective. Although Costa Rica has a strong environmental legislation, it lacks of a broader policy (beyond the current Forest Law where its PES scheme was created) to address in detail natural capital valuation and management, expressing the “official” definition of natural capital and ecosystem services the country wants to apply, valuation methods, and the general framework for institutions and financial mechanisms design, among many others. Therefore, ideally, before creating a new PES scheme, Costa Rica should create a “Natural Capital Management Law”.

Moreover, PES schemes can be better designed and successfully implemented if specific enabling conditions are present (which are somehow related to the ten principles cited before), such as having a strong ecosystem science, an effective way of communicating this science to the stakeholders involved in the PES program in order to promote buy-in of the project or policy, linking the ecological and social values and benefits of ecosystem services to human populations, strong existing institutions (e.g. legal frameworks, regulatory underpinnings, policy support networks, government support and public attitudes and political will, among others) to help facilitate the development of rules and management structures of the scheme, the fit of the existing governance structure with the PES structure and scale, and controllable transaction costs, to cite the most common (Huber-Stearns et al., 2017).

Regarding the actual design of the Blue Fund, the framework from Daily et al. (2009) is a good starting point to envision the logical sequence of steps to create this scheme, "it connects the science of quantifying services with valuation and policy work to devise payment schemes and management actions" (Daily et al., 2009, p. 22). The authors argue that in order to put in practice the concept of ecosystem services in decision making, one needs to understand the condition and functions of the ecosystems, which determine the ecosystem service they provide, value (not restricted to economic terms) those ecosystem services, create the institutions that will regulate the ecosystem and its services through different incentives designed to change people's behavior when it is harmful to the environment or to enhance an already positive behavior, and finally, take the decisions to put in practice the incentives.

Forest Trends and the Katoomba Group produced in 2010 a report that broadly follows the framework of Daily et al. (2009), it is a guide to start a payment for ecosystem services scheme focused on marine and coastal ecosystems (Forest Trends & The Katoomba Group, 2010). The authors of the report used the definition of Wunder (2005), which, as has been already stated, is limited in application in schemes dealing with public goods. Nevertheless, the four-step method they present describes specific elements that are key in the design of these type of schemes and that sometimes are overlooked. The four steps are: 1) identify marine ecosystem service prospects and potential buyers, 2) assess institutional and technical capacity, 3) structure agreements and 4) implement PES agreements. For the purposes of the general process that I present here to establish the Blue Fund, step 1 is the most relevant, the rest are more focused on the administrative/management and legal factors, which are beyond the scope of this paper.

A third example of conceptual and design method for payments for ecosystem services is the one presented by Lau (2013), which is focused on marine and coastal ecosystems, especially on blue carbon ecosystems, but with the goal of addressing other services in addition to carbon sequestration. The author establishes a five-step framework for this scheme: 1) identify ecosystem services and habitat of interest, and the biological and physical factors that contributes to the provision of the services, 2) identify the range of stakeholders who might be involved in the scheme, such as voluntary providers and potential buyers, 3) conduct monitoring activities to measure performance in comparison with a baseline using appropriate indicators, 4)

management activities to address specific threats (e.g. prevent deforestation, reduce habitat conversion, reduce pollution from land and habitat restoration), 5) assess the legal context for PES contracts (e.g. terms of the contract including the form of payment and conditions for compliance).

The Blue Fund has the general objective to invest public and private funds in marine and coastal ecosystem services conservation and restoration to enhance the well-being of people who depend on them. The research and policy agenda to establish this fund, consist of six steps (Figure 1), which are related somehow to the three frameworks presented before (Table 2), but they provide a more detailed overview of the science behind each step as well as the policy needs and implications. Each step contains a description and its general elements, as well as how these elements are applied in the case of a PES scheme focused on mangroves and coral reefs. It is worth noting that the information provided for these two ecosystems is an illustration of the information needed in each step, and does not pretend to be a complete assessment.

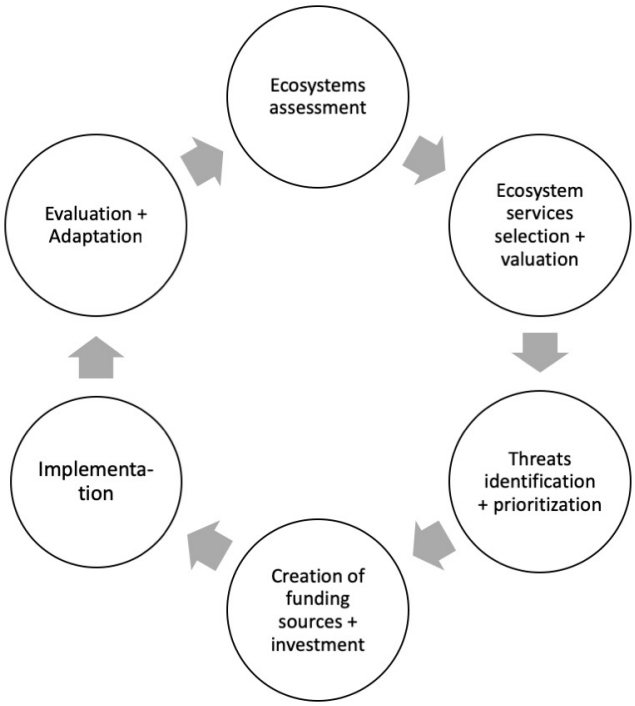

Figure 1. Steps to establish a new PES scheme for marine and coastal ecosystem services, the blue fund. This approach is not linear, it is adaptive.

Table 2. Comparison between approaches to better visualize the general elements of a new marine and coastal PES scheme.

Daily et al. (2009)	Forest Trends and the Katoomba Group (2010)	Lau (2013)	This paper
1. Understand the condition and functions of the ecosystems	1. Identify marine ecosystem service prospects and potential buyers	1. Identify ecosystem services and habitat of interest	1. Ecosystems assessment
2. Assess ecosystem services	2. Assess institutional and technical capacity	2. Identify voluntary providers and potential buyers	2. Ecosystem services assessment
3. Value ecosystem services	3. Structure agreements	3. Monitoring activities	3. Threats identification and prioritization
4. Create the institutions that will regulate the ecosystem and its services	4. Implement PES agreements	4. Management activities	4. Institutional design and financial mechanism
		5. Legal context assessment	5. Creation of funding sources and investment 6. Monitoring and Adaptation

Step 1: Ecosystems assessment

The process proposed here starts with the selection of the ecosystems that are going to be subject of the PES scheme, which for the purposes of illustration, are mangroves and coral reefs. A clear understanding of the ecosystems selected is needed, taking into consideration properties such as location, extension and health[6] of the ecosystem. The objective of this step is

[6] Very often there is not data on the health of the ecosystem under assessment, or at least for its entire extension, and conducting research on this can be time consuming and costly, therefore this is a

to establish a base line on the ecosystem cover and status, which is key to monitor the progress of the activities that are going to be put in place to conserve, restore and enhance the ecosystems. Furthermore, the baseline will be used to comply with the additionality and conditionality properties of a PES scheme, financing activities only if they are having a positive performance.

In the case of mangroves, in 2013 the total extension in Costa Rica was estimated in 36,250 ha (Programa REDD/CCAD-GIZ - SINAC, 2015) (Figure 2), representing 57% of its original cover as estimated in 1980 (FAO, 2007), which shows a 1.3% annual loss during that period of time. Costa Rica has extensive mangrove areas along its Pacific coastline. Here, mangrove forests are among the best developed, most diverse, and largest in Central America. They represent 99% of the total mangrove cover for the country, and consist of the following nuclear species: *Avicennia bicolor*, *Avicennia germinans*, *Conocarpus erectus*, *Laguncularia racemosa*, *Pelliciera rhizophorae*, *Rhizophora mangle*, *Rhizophora racemosa* and *Rhizophora harrisonii* (Kappelle, 2016).

In the north-west part of the Pacific coast mangroves tend to be smaller (up to 12 m tall in Potrero Grande, for example) possibly due to the prolonged dry season from December to April. A little further south, in more estuarine conditions, such as the mangroves of Tamarindo, *Rhizophora mangle* trees can reach 25-30 m tall. The south-east of the Pacific coast has the most extensive single area of mangroves in the deltaic system around the Térraba and Sierpe rivers (Spalding, 2010). The Atlantic coast has very different biophysical conditions, with a shoreline dominated by sandy shores and a small tidal range. The most important mangrove site here lies to the south around the Laguna de Gandoca (Spalding, 2010).

desirable information but not vital in case it is missing. The information that is strictly required in Step 1 is the location and area of the ecosystem.

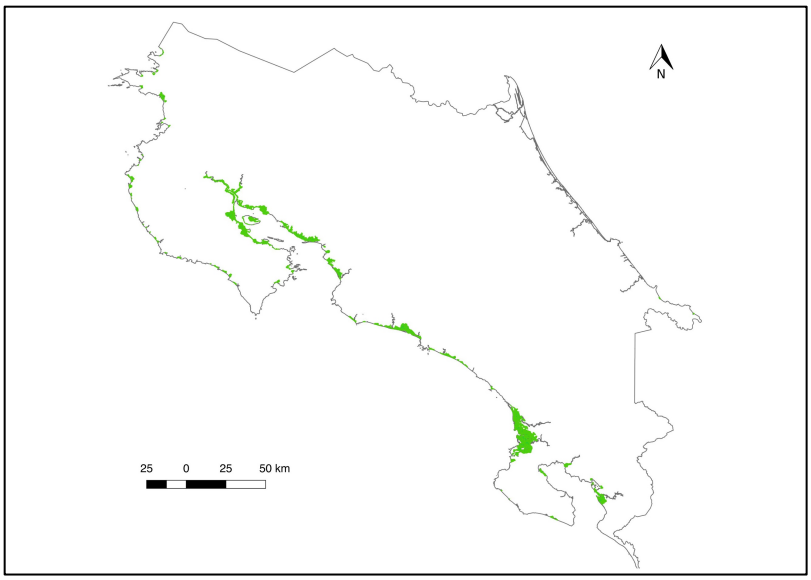

Figure 2. Mangroves of Costa Rica in 2013.
Source: Programa REDD/CCAD-GIZ - SINAC, 2015

Regarding coral reefs, the total cover area in the country is 970km^2 (Spalding et al., 2001, UNEP-WCMC, WorldFish Centre, WRI, TNC, 2010) (Figure 3). In the Pacific coast they can be divided into four geographic groups: 1) Papagayo-Nicoya, 2) Pacífico Central, 3) Osa-Golfo Dulce and 4) offshore islands including Isla del Caño (Cortes, 2016c). In Papagayo-Nicoya, the best studied area is Bahía Culebra, where the main reef-builders are *Pavona clavus* and *Pocillopora* spp, and other species includes *Pocillopora meandrina*, *Leptoseris papyracea* and *Fungia curvata*. In the Pacífico Central group, the greatest extension of coral reefs can be found in the Whale Marine National Park, where *Porites lobate* and *Pavona clavus* are the main reef building species. Coral reefs in Osa-Golfo Dulce can be found mainly in San Jocesito in the outer section of the Osa Peninsula, as well as in the shallow coasts of Golfo Dulce. Finally, perhaps the most extensive coral reefs in the country are in Isla del Caño, 15km offshore of Osa Peninsula, which are composed of twenty species of octocorals, two black corals, seventeen reef-building corals, and four ahermatypic coral species, with *Porites lobata* as the predominant species (Cortes, 2016c).

Cocos Island, the country's largest island and a world biodiversity hotspot, has extensive coral reefs, eighteen species of zooxanthellate corals and fifteen of azooxanthellates, the highest number of coral species found anywhere in Costa Rica (Cortes, 2016a). The main species of coral reefs in the Island are *Porites lobata*, *Pavona clavus*, *Pavona varians*, *Leptoseris scabra* and

Gardineroseris planulata (Cortes, 2016a). In the Caribbean coast, coral reefs are distributed in three sectors: 1) fringing reefs between Moín and Limón, 2) Cahuita National Park which has the largest fringing reef of the coast, and 3) fringing and patch reefs, carbonate banks, and an algal ridge between Puerto Viejo and Punta Mona (Cortes, 2016b).

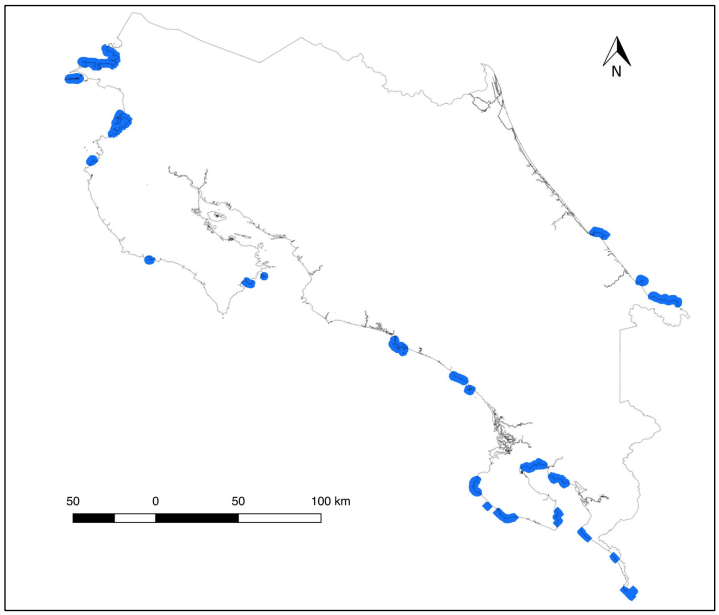

Figure 3. Coral reefs of Costa Rica in 2001.
Source: (UNEP-WCMC, WorldFish Centre, WRI, TNC, 2010)

Step 2: Ecosystem services selection and valuation

At the core of a PES scheme is the ecosystem services that are going to be "sold". As stated before, in the case of marine and coastal ecosystems, in Costa Rica, the government is the only seller or provider of ecosystem services since it is the sole owner of these ecosystems. Regarding the buyers or beneficiaries, they vary for each ecosystem service, and can be from a wide range of social sectors, as it is going to be explained later in this step.

Mangroves and coral reef provide many ecosystem services such as food, coastal protection, biological regulation, recreation and tourism, habitat and nursery for fishes of

commercial interest, among many others. A report from UNEP on marine and coastal ecosystems and human well-being describe these services based on the findings of the Millennium Ecosystem Assessment (UNEP, 2006). Beaudoin & Pendleton (2012) describes the most important ecosystem services from marine and coastal ecosystems, and Mehvar et al. (2018) provides the most recent review on identification and valuation of these ecosystem services (Table 3).

Table 3. Ecosystem services provided my mangroves and coral reefs, using the Millennium Ecosystem Assessment classification.

Provisioning services	Regulating services	Cultural services	Supporting services
Food	Biological regulation	Cultural and amenity	Biochemical
Fiber, timber, fuel*	Atmospheric and climate regulation	Recreational	Biodiversity (habitat, nursery)
Medicines, other resources	Human disease control	Aesthetics	Nutrient cycling and fertility
	Waste processing	Education and research	
	Flood/storm protection		
	Erosion control		

* Only for mangroves
Source: UNEP, 2006; Mehvar et al., 2018

Management policies for these ecosystem services vary depending on their characteristics, mainly what "type of good" is each ecosystem service, which is determined by the combination of its characteristics of rivalry and excludability. On one hand, rivalry is an innate property of the good which cannot be altered by policies or institutions, and it means in its purest form that the use of that good or service by an individual prevents its use by another individual, for example eating a banana (Daly & Farley, 2004). On the other hand, a good that is non-rival can be used by one individual without affecting significantly the use of the same good by others (Daly & Farley, 2004), as in the case of some ecosystem services such as water regulation. Nevertheless, ecosystem services such as recreation and tourism, are non-rival services but their quality can be affected by the number of its beneficiaries, and therefore, are called "congestible",

for example when a diver is enjoying a coral reef and is accompanied by another diver, but if instead of only one extra diver there would be twenty in the same coral reef, the experience of the first diver, (and the rest of the divers) would probably be negatively affected. Finally, a good can be anti-rival when it is enhanced by its use from multiple people (e.g. information) (Kemkes, Farley, & Koliba, 2010).

The second characteristic of a good, excludability, refers to those goods that its use can be prevented through policies, institutions and technology, and therefore, is not an innate characteristic as in the case of rivalry, but most rival goods can be made excludable through institutions (Kemkes et al., 2010) like the majority of provision services such as food, medicines and wood, among others, which are rival but they need to have property rights and laws to prevent its unauthorized use (Kemkes et al., 2010). On the contrary, goods and services can be inherently non-excludable when it is impossible to create property rights or it is too costly to restrict its use, as it would be the case to try limiting someone from the benefits of climate regulation (Daly & Farley, 2004).

Using the type of good categories described in Kemkes et al. (2010), the ecosystem services from mangroves and coral reefs listed before can be grouped as shown in table 4, these categories can be used to determine if a certain payment scheme would be an appropriate mechanism for its provision (Engel et al., 2008).

Table 4. Ecosystem services from mangroves and coral reefs categorized by type of good, defined by rivalry and excludability.

	Non-excludable		**Excludable**	
Non-rival	Biological regulation Freshwater storage and retention Hydrological balance Atmospheric and climate regulation Human disease control Waste processing Flood/storm protection	*Pure public good*	None	*Inefficient market good*

	Erosion control Biochemical Biodiversity (habitat, nursery) Nutrient cycling and fertility			
Rival	Food	*Common pool resource*	Fiber, timber, fuel Medicines, other resources	*Market good*
Congestible	Recreational Aesthetics	*Congestible public good*	None	*Toll or club good*
Anti-rival	Education and research Cultural and amenity	*Public good*	None	*Inefficient market good*

Almost all ecosystem services from mangroves and coral reefs are public goods or common pool resources, and only a few services (provisioning services such as food, timber and medicines) are market goods, highlighting the challenge of sustainably manage them. Because these ecosystems are completely owned and protected by the government, in theory, there should not be any market good since its extraction and commercialization would be illegal, although a few exceptions exist as in the case of extraction of bivalves in mangrove forests (Hernández-Blanco et al., 2018). Hence, all provisioning services here have been excludable through the national legislation, requiring only monitoring activities to enforce the respective laws. On the other hand, cultural, regulating and supporting services will require specific mechanisms to use them in a sustainable way, which will be analyzed in the fourth step of the methodology presented here.

Because it is beyond the scope of this paper to assess all marine and coastal ecosystems and all its services, I selected three services for mangroves and three for coral reefs to illustrate how a marine PES scheme could work, but this is a starting point; this methodology can be applied to more ecosystems and services. As described before, the government is the only seller of these six services (and any other marine and coastal ecosystem service). Each of the ecosystem services selected have different beneficiaries, who would be the buyers in a PES scheme, such as tourists, governments from other countries, and the private sector, among others (Table 5). It is worth noting that ecosystems do not produce services in insolation, therefore some of these specific services from these two ecosystems can have common beneficiaries, as in the case of coastal

protection, since a coral reef could represent a first wave energy barrier before it reaches the mangroves.

Table 5. List of ecosystem services that are going to be part of the PES scheme proposed here, as well as example of potential buyers/beneficiaries for each one. A code has been given to each service in order to facilitate the visualization of relations between services, beneficiaries, and later in this paper the relation with threats and solutions.

Ecosystem	Ecosystem Service	Examples of buyers/beneficiaries
Coral reefs	Tourism/Recreation (CRES1)	Tourists
	Coastal protection (CRES2)	Private and public sector
	Biodiversity (habitat, nursery) (CRES3)	Society as a whole, fisheries
Mangroves	Blue carbon (MES1)	Governments, business
	Coastal protection (MES2)	Private and public sector
	Food (fisheries, fish and mollusks) (MES3)	Fisheries

After selecting which ecosystem services are going to be part of the program, an economic valuation should be conducted for several reasons. First, the value of ecosystem services allow policy makers and potential buyers from the PES scheme to visualize, through a cost-benefit analysis, the return of the investment in activities to protect, restore or enhance ecosystems, which can be in a proportion as high as $100 dollars for each dollar invested (Balmford et al., 2002). Second, economic valuation studies can help generate demand for a service, but the value obtained should not be confused with the actual price of the ecosystem service (Forest Trends & The Katoomba Group, 2010). Third, the PES scheme can be funded through different sources, as it is going to be explained in the fourth step of this method, and one funding source can be the amount charged to polluters for their negative impact on mangroves and coral reefs, and establishing a value per unit of area (e.g. hectare, km^2) can be a useful reference to determine how much to charge them.

Mehvar et al. (2018) lists the most appropriate valuation methods for each marine and coastal ecosystem service, as well as some examples of economic estimates obtained in different parts of the world. For example, in the case of coral reefs, tourism and recreation in a marine national park in Seychelles was estimated in $88,000 for the whole area (Mathieu et al., 2003),

protection to coastal erosion in $160-172000 per km^2 of reef per year in Sri Lanka (Berg, Öhman et al., 1998), and habitat support for fisheries in the Caribbean Sea in $95-140 million (Burke & Jon Maidens, 2004). In Costa Rica, in 1996, the coral reef of Cahuita National Park was estimated in $1.4 million (Blair et al., 1996). Having estimates of the value of the services of coral reefs of Costa Rica using primary studies still remain as a research gap.

The estimates from De Groot et al. (2012) for coral reefs can be used as a first approximation for the economic value of the ecosystem services from coral reefs that are the target of the proposed PES scheme here (Table 6).

Table 6. Economic value of the ecosystem services of coral reefs. Estimates are in Int.$/ha/year, 2007 price levels

Ecosystem Service	Per hectare per year value
Recreation	96,302
Coastal protection	16,991
Biodiversity (habitat)	16,210

Source: De Groot et al. (2012)

Regarding mangroves, the most updated study for Costa Rica was conducted by Hernández-Blanco et al. (2018), in which the authors used a hybrid methodology to estimate the value of eleven ecosystem services for the total extension of mangrove forests in Costa Rica, and more sophisticated methods (e.g. modelling using INVEST) to value three ecosystem services (i.e. climate regulation, coastal protection and fisheries) in the Gulf of Nicoya (Table 7). The value of these three ecosystem services, which are the target of the PES scheme proposed here, can be incorporated in the program with a high degree of certainty, since they were valued using primary studies, instead of more rough technics such as benefit transfer which can produce values with even a 1500% error (Navrud & Ready, 2007).

Table 7. Economic value of three ecosystem services from mangroves in Costa Rica. Estimates are in 2015 USD.

Ecosystem Service	Per hectare per year value
Climate regulation	1,914.86
Coastal protection	5,154.44
Food	231.55

Source: Adapted from Hernandez-Blanco et al. (2018)

Step 3: Threats identification and prioritization

The extension and health of the ecosystem, which are determined in step 1, are product of the existence or absence of threats that have a detrimental impact on the ecosystem. It is crucial to identify threats in order to invest in the solutions that are most needed, which increases the efficacy and efficiency of the scheme (Schomers & Matzdorf, 2013). Furthermore, the identification and quantification of threats will be part of the baseline from which the success of the solutions implemented is going to be compared, incorporating this way in the scheme the core principle of additionality (Engel et al., 2008).

In Costa Rica, besides the myriad of ecosystem services that marine and coastal ecosystems provide, these ecosystems face numerous threats from habitat loss, invasive species, pollution, increasing human population, overharvesting and climate change. The 5th Report of Costa Rica to the Convention of Biological Diversity identified in 2014 in a general way the following drivers of degradation of these ecosystems: 1) Pollution from solid waste in areas such as Nicoya and the river mouth of the Tárcoles River, chemical pollution from oil spills in Puntarenas, and liquid pollution from watersheds that carry pesticides, organic compounds, detergents, heavy metals, pharmaceutical waste, among others; 2) sedimentation in regions such as Golfo Dulce; 3) over-exploitation of resources; 4) lack of planification and adequate regulation on economic activities such as coastal development and fisheries; 5) marine tourism; and 6) climate change and climate variability (specially the ENSO) (SINAC, 2014). Table 8 present the

main threats in Costa Rica to coral reefs and mangroves, which constitutes the list of some of the potential problems that the PES scheme could address.

Table 8. Main threats to coral reefs and mangroves in Costa Rica.

Ecosystem	Threat	Reference
Coral reefs	Warming impact during El Niño Southern Oscillation (ENSO), such as the bleaching events that have occurred in the North Pacific.	(Cortes, 2016b)
	Climate change impacts (e.g. warmer sea water and lower pH levels).	(IPCC, 2007)
	Phytoplankton blooms (e.g. in Guanacaste).	(Cortés & Jiménez, 2003)
	Terrigenous sedimentation from deforestation in the mountains and agricultural activities (in Pacific in the Whale National Marine Park and Golfo Dulce, as well as in the Caribbean in Cahuita).	
	Invasive species (e.g. sea urchin in the Pacific).	(Fernández, 2007)
	Algae blooms – red tides	(Vargas -Montero, 2004)
	Sewage waters from the town of Limon.	(Nielsen-Muñoz & Quesada-Alpízar, 2006)
	Tourism activities (e.g. hotel development, boats and yachts, divers, coral extraction by tourists).	(Dubinsky & Stambler, 1996)
	Plastic waste	(Lamb et al., 2018)
Mangroves	Changes in sedimentation regimes (e.g. from agricultural activities).	(Cortes, 2016b); (Morales, 2013);
	Shrimp farming.	(Nielsen-Muñoz & Quesada-Alpízar, 2006)
	Pollution from organic waste, pesticides, hydrocarbons, and solid waste (e.g. plastic).	
	Advance of the agricultural frontier.	
	Land use change.	
	Illegal logging.	
	Fishing using illegal fishing technics.	
	Climate change impacts, such as warmer soils, sea level rise, more frequent and severe storms.	

The identification and quantification of threats can be a complex, costly and time-consuming process, and therefore, it can be based, at least in the beginning, on information that has been gathered for a specific site or for areas with similar biophysical and socio-economic properties, as done in Table 8. Other options include more novel tools, such as modelling. For example, Arkema et al (2015) determined the marine and coastal ecosystems that are more at

risk in Belize from anthropogenic pressures by using INVEST, an analysis that helped determine the best options of coastal planning through different scenarios.

From Table 8, both coral reefs and mangroves are threatened by unsustainable agricultural activities (T1), climate change and climate variability (T2) and plastic waste (T3), and therefore, they can be prioritized as the main drivers of change and hence those that require the most urgent solutions. In the case of climate change, although this threat is treated here in some level equally to the other two threats, it is different in its nature, scale and magnitude, and consequently it should be assessed in future studies from a more global and long-term perspective, being probably the greatest threats of all to coral reefs and mangroves. The followings steps will address these specific threats.

Step 4: Creation of funding sources and investment

To incorporate the positive externalities (i.e. ecosystem services) from ecosystem conservation, enhancement and restoration, as well as the negative externalities (i.e. threats and negative impacts) from economic activities, into a new PES scheme in order to make them visible in the current economic system, a financial mechanism needs to be designed, containing the funding sources and investments of the activities that will address the environmental impacts, as well as the institutional arrangement that will manage this mechanism.

Having a clear picture of who the polluters of ecosystems and the buyers or beneficiaries of their services are, it is indispensable to create the most appropriate financial incentives that will produce the change in behaviour towards a better ecosystem management. As described in the last step, both coral reefs and mangrove are negatively affected by three main polluters. From the buyers perspective, taking in consideration the six ecosystem services selected for this analysis, coral reefs provide benefits to tourists (i.e. CRES1), business (i.e. CRES2), and fisheries (i.e. CRES 3), and mangrove forests provide benefits to business (i.e. MES1-2), fisheries (i.e. MES3) and governments (i.e. MES1) (Figure 4). These ecosystem-beneficiary relations are based on the

current situation in Costa Rica, which can change in the future under different policies and management scenarios.

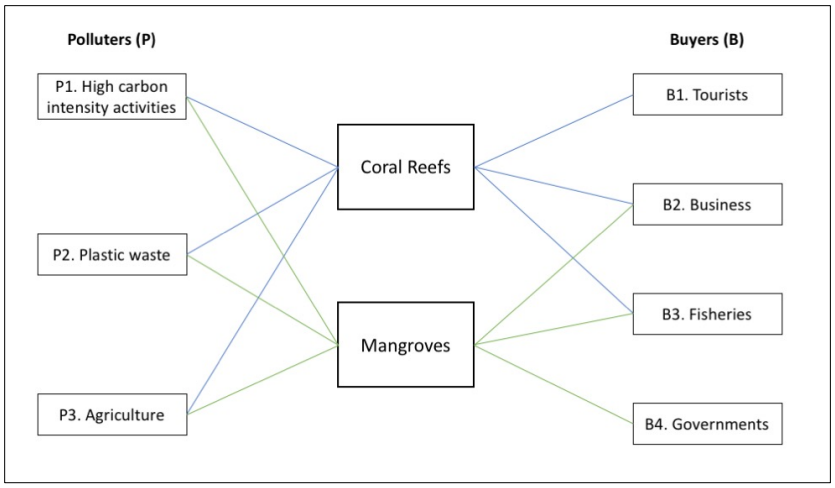

Figure 4. Polluters (P) and Buyers/Beneficiaries (B) of coral reefs and mangroves. Blue lines mean a relation with coral reefs, green lines with mangroves. This is an oversimplification to illustrate different impacts and benefits for each ecosystem.

The polluters and buyers will be the source of funding the government (since it is the only seller) will receive in order to be able to fund activities that will address the threats mentioned before. The government should design the institutional framework to manage these financial resources as well as to monitor the success of the investments. In this case I propose to create the Blue Fund to conduct these activities. Nevertheless, creating this single institution is not the only option, others could include expanding the current forest fund (i.e. FONAFIFO) into a broader fund such as a "Natural Capital Fund", including public forests (FONAFIFO only includes private forests) and a wide array of other ecosystems such as the different types of wetlands. One of the main arguments for having an integrated Natural Capital Fund is that ecosystems do not provide services in insolation from other ecosystems, as well as ecosystems are affected by activities in other ecosystems such as the case of coral degradation from pollution caused by sedimentation from unsustainable agricultural and forestry practices upstream. Another approach would be to create a common asset trust, "a legal relationship between trustees, who

manage a pool of wealth, and beneficiaries, for whom the wealth is managed" (Farley et al., 2015), for marine and coastal ecosystems.

In this paper, I chose to frame the PES scheme under the Blue Fund, which will be focused on coral reefs and mangroves. This fund would receive economic resources from activities that are degrading these ecosystems (i.e. polluters) as well as from activities that depend on their ecosystem services (Figure 5). This is not a comprehensive list of funding resources, it is rather a guide to create the most appropriate ones depending on the social and environmental context.

Funding from polluters would be a mandatory payment under the "polluters pay principle". Of course, these types of mandatory measures can be complex to implement, especially because all the regulations that need to be created and approved by the government. Nevertheless, they have shown to be a reliable source of funding, as in the case of the 3.5% tax on fossil fuels in Costa Rica to fund FONAFIFO. To address climate change (T2), a similar tax can be implemented, such as a carbon tax which has proven to be an effective measure to mitigate the carbon footprint of a country (Li & Lin, 2011). Further research is needed to determine the possible effects of such tax in different sectors in Costa Rica, specially the effect on poverty (Gonzalez, 2012).

Another funding source could be a tax on plastic since plastic waste (T3) is a significant threat to coral reefs and mangrove as described in Step 3. A recent study conducted by Pacheco et al (2018) determined that a tax on imports of plastic products of all types and plastic supplies in Costa Rica can generate $25 millions of income for the Blue Fund for 2019, $26 million for 2020 and $27 million for 2021. Moreover, the authors also estimated the annual decrease of the amount collected from this tax due to a decrease in the consumption of plastic from consumers and industry (which is the goal of the incentive), and calculated that by 2028, tax collection would be reduced in 50%. Possible decreases in income from taxes therefore need to be taken in consideration in order to make the financial structure of the fund more resilient; consequently, other funding sources needs to exist.

From the beneficiaries and buyers side, the Blue Fund could receive voluntary payments from these actors from the use of the ecosystem services listed in Step 2. In the case of tourism (CRES1) from coral reefs, setting a diving fee based on the Willingness to Pay (WTP) of divers for

conservation activities can provide an important source of funding, as it has been proven in many parts of the world. For example, Trujillo et al (2016) conducted a study on the willingness to pay for the conservation of the coral reefs in the Corals of Rosario and San Bernardo National Natural Park, in Colombia. The authors estimated that divers are willing to pay on average $90 per person, which is higher than the fee charged to enter the park.

In the Mu Ko Similan Marine National Park in Thailand, it was estimated that divers are willing to pay on average between $27-63 per person per year, which in this case, it is also significantly higher than the current diving fee of $5 per day, and therefore, the national park could in theory increase at least fivefold this fee without affecting the number of tourists willing to visit it; an extra funding that can be used for several administrative, research, educational, conservation and restoration activities (Asafu-Adjaye & Tapsuwan, 2008). In a similar study about the WTP for diving fees in the Bonaire National Marine Park, estimated that 94% of people interviewed were willing to pay at least $20 annually to access the marine park, more than 75% of divers said they would pay a minimum of $30 and 50% were willing to pay $50 or more. The current diving fee is $10, and therefore, there is a high potential to increase this fee to at least $20 without having an impact on visitation rates, which would obviously represent a double of the funding that the park can receive (Thur, 2010).

In all three examples, fees have been estimated arbitrarily, without taking into consideration the WTP of divers, and have been set much lower than the buyer is willing to pay. A reason for setting low diving fees, especially in developing countries, is that managers of the marine area are often concerned that the collection of fees will increase administrative expenses (Wielgus et al., 2010). Conducting similar studies in Costa Rica to estimate diver's WTP is therefore key to setting appropriate diving fees and maximize the financial resources than can be generated for the proposed Blue Fund. Payments can be differentiated for national and international tourists (two-tier pricing system), as it is already the case in the majority of protected areas in the country, and which has proven to be valid due to a higher WTP of international divers (Asafu-Adjaye & Tapsuwan, 2008; Trujillo et al., 2016; Wielgus et al., 2010; Emang et al., 2016).

A second example of a possible funding source from a voluntary approach is the creation of carbon credits from mangroves, which are among the most carbon-rich forests in the tropics, containing on average 1,023Mg of carbon per hectare (Donato et al., 2011). Carbon credits, and in this case blue carbon credits, can be generated by carbon markets under a compliance scheme (mandatory under national or international agreements) or by voluntary projects such as the case when governments or business choose to invest in carbon off-sets to reach a climate change mitigation goal (Locatelli et al., 2014). A voluntary market approach to sell blue carbon credits would probably be the best option, as a financial resource for the Blue Fund, because this market already exists, instead of waiting for regulated schemes to take effect (Ullman et al., 2013), although credits in the voluntary market often worth less than in the compliance market due to lower demand, quality standards and lack of transferability to the compliance market (Wylie et al., 2016; Ullman et al., 2013).

There are already some successful examples of mangroves conservation through the selling of blue carbon credits. The Mikoko Pamoja restoration and reforestation project in Gazi Bay, Kenya, which includes 117ha of nationally-owned mangroves, is a PES scheme with the community of Gazi Bay, in collaboration with Plant Vivo who manages the credits and research on carbon storage. The revenues collected from the sale of the credits, which has been $12,500 annually (each credit is sold at $6.5-10 for 2013-2014), are invested in paying one full time staff member, mangrove planting and conservation and community development projects (e.g. school construction, purchase of books and installation of water pumps, among others). Another example is the case of the India Sundarbans Mangrove Restoration project, which has been implemented as a Verified Carbon Standard project with the goal of planting 6,000ha over three years that will store 700,000 t of carbon over 20 years. The revenues from selling carbon credits are invested in paying the community for the work ($2.5 per day for four hours of planting) and for the technical survey and scientific monitoring required for carbon offset certification (Wylie et al., 2016).

More recently, The Nature Conservancy and XL Catlin announced a project to develop “Blue Carbon Resilience Credits”, which will incorporate both mitigation and adaptation to climate change benefits from coastal wetland ecosystems, with the goal that insurance

companies will be able to offset their carbon footprint while understanding at the same time the contribution these ecosystems provide for coastal protection (The Nature Conservancy, 2018).

Costa Rica has already advanced significantly in the process to produce blue carbon credits. The carbon storage capacity of the main mangroves of the country has been calculated, with estimates of 413-1,335 MgC/ha in the mangroves of the Gulf of Nicoya (Cifuentes-Jara et al., 2014) , and 391-438 MgC/ha at the Terraba-Sierpe National Wetland (BIOMARCC-SINAC-GIZ, 2012). Moreover, the economic value of both the carbon storage and sequestration services has been estimated, $131,585/ha and $1,915/ha respectively (Hernández-Blanco et al., 2018), allowing to conduct cost-benefit analysis that are key in decision making.

Another important step Costa Rica has taken to develop the general framework for blue carbon projects is the formulation of the Blue Carbon Strategy (currently waiting for the Ministry of Environment to validate it) (Hernández-Blanco, 2014; Hernández-Blanco, 2017). Furthermore, Costa Rica has begun the process of certifying with Gold Standard blue carbon credits in 587 hectares of the Térraba-Sierpe National Wetland through mangrove rehabilitation in a 30 years period, which would generate at least $2.5 million to the Ministry of Environment for conservation and restoration activities (Ruiz, 2018), demonstrating the high potential that blue carbon credits have in supporting the Blue Fund.

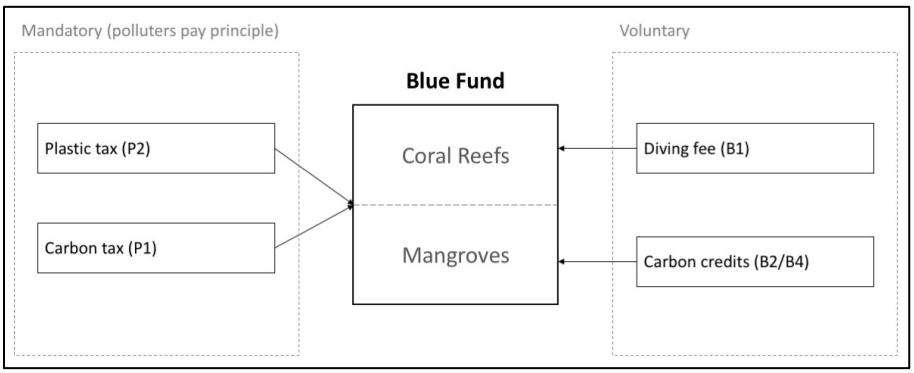

Figure 5. Mandatory and voluntary funding sources for the Blue Fund, each depending on the type of relation (negative impact or benefit) society has on coral reefs and mangroves.

These two funding sources are proposed here, taking into consideration the physical and institutional characteristics of ecosystem services, in which fees are recommended for

congestible club or toll services (i.e. diving), and voluntary markets for public good services (i.e. climate regulation) (Farley & Costanza, 2010). Again, this is not a comprehensive list of funding sources that can be created to maintain the conservation and restoration activities of the Blue Fund, but rather a condensed list and the analysis needed to develop possible financial resources. Other incentives should be established in response to other ecosystem services provided by coral reefs and mangroves, such as fisheries, coastal protection and habitat for biodiversity, among others not considered in this discussion.

Step 5: Implementation of conservation and restoration projects

The different projects that are going to be funded through the funds collected in the previous step, need to address the main threats identified in step 3, with the general goal of protecting, restoring or enhancing the ecosystem services provided by the targeted ecosystems. These projects can be developed by different social actors (e.g. NGOs, local communities, business), which will be called here "the implementers" and who will receive the payments to conduct management activities to protect these natural resources.

As stated by Engel et al. (2008), financial incentives (i.e. payments) ideally should be made directly on the bases of the ecosystem service provided, but these "output-based" payments are often not possible because quantifying the provision of ecosystem services as well as to make them evident for land users can be difficult. The authors argue that because of this, the majority of PES schemes make payments based on the area of land protected or enhanced through a particular activity, instead of paying for a unit of ecosystem service (e.g. ton of carbon sequestered). An "input-based" approach for paying for ecosystem services can be made on a per hectare basis, as in the case of mangrove reforestation or coral reef restoration, or based on other indicators such as working hours on cleaning a beach, number of invasive species individuals fished, and number of trees planted, among many others.

Payments to implementers must be equal or greater than the cost of conservation shouldered my participating communities (Mohammed, 2012). Tacconi (2012) also argues that

payments need to be at least equal to the benefits forgone by the ecosystem services providers[7] and equal or less than the value of the ecosystem service to the buyers. At the end, the payment amount will depend on factors such as economic value, financial value, transaction price, and relative costs of alternatives (Beaudoin & Pendleton, 2012). The economic value of each ecosystem service estimated in Step 2 is not used to set the price or the payment, but it is used to inform buyers the benefits they will receive from healthy ecosystems.

Payments can be made in-cash or in-kind. Monetary payments consist of direct payments to people to ensure the provision of the targeted ecosystem service. Examples of this includes the case of the Philippine's No-Fire Bonus scheme, where local level governments and communities are rewarded for fire control in state forests in their area with the purpose of maintaining watershed services and wildlife habitats (Tacconi, 2012); in South Africa unemployed individuals are hired to clear invasive plant species and to restore natural fire regimes in mountain catchments and riparian zones (Schomers & Matzdorf, 2013); the case of direct contracts for bird nest protection (Muradian et al., 2010); and the scheme in Tanzania that pays communities to find the nests of endangered sea turtles and then reporting them to project monitors (Mohammed, 2012).

In-kind payments can include a myriad of non-cash rewards, as it is implemented in the Rewarding Upland Poor for Environmental Services (RUPES) program, which include scholarships for local students, technical assistance to local farmers and investment in infrastructure such as roads and electricity and water systems (Schomers & Matzdorf, 2013).

Because of the nature of the PES scheme proposed here, in which the state is the only seller of ecosystem services, payments are made in two moments of the framework, firstly from private buyers to the government, through mechanisms such as the diving fee and blue carbon credits mentioned in the previous step, and secondly from the government to the implementers of the activities that are going to be described in this step. In the first case, ecosystem services are sold under a "stacking" or "layering" approach, in which separate payments are made for

[7] In the case of the proposed PES scheme here, the cost of opportunity will be related to the activities that implementers will stop doing (e.g. fishing, illegal logging, unsustainable agricultural practices) in order conduct conservation projects.

each ecosystem service. In the second case, services are sold under a "bundling" approach, in which payments are made for multiple ecosystem services grouped together into a single package of conservation outcomes (Lau, 2013).

The following is a sample of the many activities that can be developed through payments to different sectors of society to address the most significant threats to the targeted ecosystems of this PES scheme in order to protect and restore their ecosystem services (Table 9).

Table 9. Example of solutions (S) that can be funded to address the main threats (T) to mangroves and coral reefs.

Threat	Solution (activities to be funded)	Implementer	Payment
Agricultural activities (T1)	Sustainable agricultural practices as part of the current PES scheme (S1)	Farmers	In-cash or in-kind to decrease the environmental impact (e.g. agrochemicals, sedimentation) of agricultural practices.
Climate change and ENSO (T2)	Mangrove and coral reef restoration (S2)	Academia, local communities, private sector.	In-cash to pay people in charge of the project, and/or in-kind for any resources needed to conduct the activity.

Agricultural activities (T1) – Expansion of the current PES (S1). The payment for ecosystem services scheme of Costa Rica has been extensively described (Pagiola, 2008; Porras et al., 2013) due to its pioneering role, more than two decades ago, in developing new financial mechanisms for forest conservation. The program currently funds 16 activities with the goals of protecting and recovering forest cover (FONAFIFO, 2018), including reforestation activities, forest conservation, agroforestry systems and natural regeneration. Nevertheless, the program does not include any activities to support sustainable agricultural practices (SAP), which has proven to be very effective in reducing environmental impacts on downstream communities and ecosystems, such as the well-known case of the Catskill watershed in New York, where the city worked with farmers to help them manage their land in a way that they can meet their financial goals and, at the same time, provide clean water downstream (Tercek & Adams, 2013). Incorporating a new category of SAP under the current scheme could significantly help in

reducing the impact on mangroves and coral reefs from sedimentation and agrochemical pollutants.

Furthermore, payment for SAP plus all the current activities that FONAFIFO fund should be targeted to areas where they can help reduce the impact from upstream activities to coastal ecosystems. This interaction, between forest conservation and coastal conservation is not considered in the current program, perhaps because the program clearly targets forests, but also because the myopic vision that ecosystems function in insolation. Looking at the spatial distribution of all the activities that FONAFIFO funds, this disconnection between ecosystems conservation is evident in the current PES scheme (Figure 6), which can be solved relatively easy by promoting the conservation and restoration of forests due to the fact that its location can potentially enhance the health of coastal ecosystems.

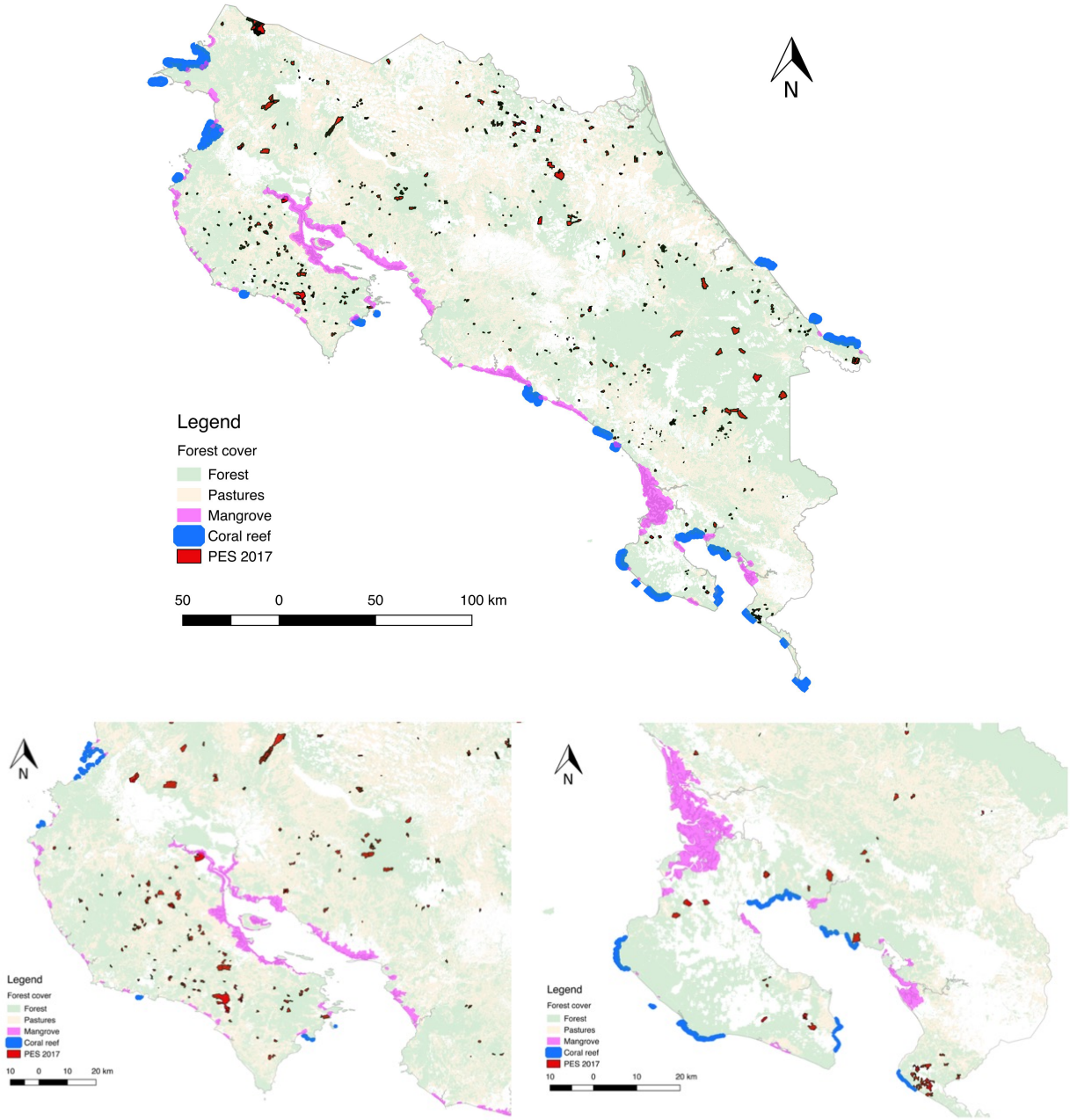

Figure 6. Location of all the PES projects of FONAFIFO in 2017 (in red), and the location of coral reefs (in blue) and mangroves (in purple) in Costa Rica, showing the current gap between projects than could link forest and coastal conservation.
Source: FONAFIFO, 2017; UNEP-WCMC, WorldFish Centre, WRI, TNC, 2010; Programa REDD/CCAD-GIZ - SINAC, 2015

Climate change and ENSO (T2) – Restoration projects (S2). Ecosystem degradation or loss can be caused by a multiple array of natural and anthropogenic impacts. Climate change is projected to be one of the main threats to biodiversity and the benefits it provides to society, increasing the negative impact of all the other threats (IPCC, 2014), and therefore, innovative measures to increase the resilience of ecosystems are needed. Restoration of degraded or lost ecosystems can be one of the main responses to climate change and climate variability (Harris et al., 2006), with the dual goal of enhancing and/or restoring the health of the ecosystem, and consequently, as a climate change adaptation and/or mitigation strategy for society.

Restoration of mangroves has been undertaken around the world, and it is estimated that globally, approximately 8,120 km^2 or 6% of former mangrove area is considered restorable (Worthington & Spalding, 2018). Considering factors that influence mangrove restoration such as recent sea level rise, projected future sea level rise, recent change in sediments, time since loss, average size of loss patches and the proximity of loss areas to remaining mangroves (Ocean Wealth, 2018a), Costa Rica has a mean restoration potential score of 67%, with a total restorable area of 1,306ha (Ocean Wealth, 2018b).

Increasing carbon sinks, in this case, increasing mangrove cover, needs to be accompanied by a set of policies that secure the long-term health and existence of these new forests as much as possible, since carbon must remain stored for more than 10,000 years in order to be considered a useful reduction of the atmospheric carbon. Hence, the mitigation value of restoring mangroves lies not in their present net uptake of CO_2, but in the longevity of their accumulated carbon stocks (Mackey et al., 2013).

There are already a few community-led restoration projects in Costa Rica. For example, in Chira Island, 6 women from the Palito community and 21 from the town of Montero started in 2013, a mangrove reforestation project with the support of Conservation International who helped providing the materials for construction of the nurseries and the advice from biologists, but no payment for the job. Apart from the reforestation project, women from Palito are dedicated to clams farming and the women from Montero to the extraction of molluscs, both activities with low profitability. By 2014, they had already sown 7,575 seedlings (Soto, 2014).

Another example is the mangrove restoration project in the community of Pitahaya in Puntarenas, which started in 2015, led by the Association of Artisanal Fishermen from San Luis, Puntarenas. In this project, ten people work and get paid $300 for a full-time job for 2 months through the National Employment Program, and other resources (e.g. plastic bags for the nurseries) are provided by the municipality of Puntarenas or by the University of Costa Rica. In this case, unlike the project at Chira Island, people in charge of reforestation activities are paid, which is key for the sustainability of the project, since it has been demonstrated that when communities does not receive this type of support, restoration projects get spread over a longer time horizon (Ranjan, 2019). To date, this project has planted 62,000 mangrove seedlings (Asociación de marinos artesanales de San Luis Puntarenas, 2018).

Regarding coral reefs, they have been lost or degraded by several anthropogenic and natural impacts, and climate change is projected to be one of the main causes of coral reef loss globally (IPCC, 2014). Apart from reducing greenhouse gases emissions worldwide, perhaps one of the most important measures to protect these ecosystems is to increase or restore its resilience to these threats. Due to this, several restoration techniques have been applied, including removing loose debris from the reef, rebuilding three-dimensional structures onto leveled-scarified reef surfaces, and transplanting corals back on the cleared reef surfaces, among others (Jaap, 2000).

One of the most promising restoration approaches is coral gardening, a technic inspired by silviculture, consisting of a two-step restoration process: 1) a mid-water nursery phase, where coral-nubbins are farmed, and then 2) nursery-grown coral colonies are transplanted to degraded reefs sites (Bongiorni et al., 2011). This technic has been tested worldwide, at least 86 coral species and over 100,000 colonies successfully farmed in different archetype nurseries (Rinkevich, 2014). Additional to the restoration benefits this method provides, it also generates important indirect benefits such as its contribution to the rapid creation of fish and invertebrate habitats, creation of genetic and genotypic repositories, and enhancement of physical connectivity of depleted adult populations, and perhaps more importantly, economic services in the form of employment and improved tourism opportunities (Lirman & Schopmeyer, 2016).

There are already a few projects of coral gardening in Costa Rica. In 2016, the University of Costa Rica started a project in Golfo Dulce, in the south of the country, a site that was selected because corals here have been recovering due to better conservation practices in the surrounding coastal areas, which has reduced sedimentation, and the responsible fishing that has been implemented there. The type of nursery in this project consists of a tree-shaped structure built with PVC pipe and fiberglass, in each of the branches, coral fragments that measure between 0.5 to 1.5 cm^2 hang. The project currently focuses on three genera of corals, *Pocillopora*, *Porites*, and *Pavona*, which has shown high survival rates, 100%, 57% and 98% respectively. One key aspect of this project was the development of a strategy to involve fishermen and other members of coastal communities (Blanco, 2018).

It is worth noting that reef restoration, as well as mangrove restoration, should be conducted in combination with other conservation strategies, such as watershed management, afforestation for erosion management, establishing of Marine Protected Areas, and sustainable fishing practices, among others (Lirman & Schopmeyer, 2016; UN Environment et al., 2018).

These two examples of ecosystem conservation, enhancement and restoration, illustrate the myriad of activities and projects that can be funded through the Blue Fund. Other activities that can be incorporated under this new PES scheme are cleaning projects to address the threat of plastic pollution (T3) like the one proposed by Pacheco et al (2018) to employ fishermen to do this, natural capital insurance such as the Coastal Zone Management Trust development by TNC in partnership with the government and hotel owners to protect coral reefs through continuous maintenance and the buying of reef insurance (TNC, 2017), research and development activities, blue bonds, among many others.

Step 6: Evaluation and Adaptation

The last step is the monitoring and evaluation of the activities undertaken to assess if they were successful, as well as to make any necessary changes to improve them. The main goal of this step is to assess the degree of additionality the project produced, which is a key element of

any PES scheme but that it is often ignored. Lack of additionality means that the resources invested did not produce additional improvements compared with a business as usual scenario or by not doing anything (Tacconi, 2012).

The other key element of PES schemes is conditionality, which needs to be evaluated in order to maintain or suspend payments made to implementers of conservation activities. According to Engel et al. (2008), for payments to be conditional, it must be possible to verify the existence of the ecosystem service and to establish a baseline against which additional units "provided" can be measured. Furthermore, the positive impact of the activities implemented (i.e. additionality) need to be measured using standardized indicators, which of course will vary depending on the activity being evaluated.

In the case of sustainable agricultural activities (S1), examples of indicators are crop production per drop of water withdrawn, food production per unit of GHG emissions, share of agriculture land enrolled in agricultural preserve, conversion of natural to agricultural land, share of cropland under conservation, fertilizer applied per unit of arable land, share of cropland under integrated pest management, and pesticide use per unit of cropland, among others (Reytar et al., 2014). For restoration projects, some examples of indicators are the number of mangrove seedlings planted, number of coral nubbins farmed, number of nursery-grown coral colonies transplanted, area restored, and the increase of biodiversity associated with these ecosystems.

One aspect that should be taken into consideration in the creation and implementation of this Fund, is that the future is impossible to predict, especially in the Anthropocene, where the Blue Fund is embedded in a complex social-ecological system subject to non-linear changes, which can impact some or all of the components of the funding sources and implementation activities. Therefore, this scheme needs to be adaptive to plausible future scenarios, many of the ideas proposed in this paper are based on a snapshot of the reality, but further research can take into account this complexity across space and time to make the program more resilient.

Conclusion

Following the six steps described in this paper, Costa Rica can be again a pioneer in establishing conservation strategies for marine and coastal ecosystems, which have received historically less attention than tropical forests. Each step explains its goal, and describe specific examples on how to apply it in mangrove forests and coral reefs, providing, as well, examples of current initiatives around the world that prove that it is possible to do it under this new PES scheme.

The Blue fund incorporates many key elements of any PES scheme that are absent partially or entirely in the current program for forests, such as the principle of additionality, a direct relation between the buyers and the seller, a direct relation between threats, funding sources and solutions, and the economic value of ecosystem services. These elements can also be incorporated in the current scheme to improve its financial sustainability and to enhance the efficacy and efficiency of the programs it funds.

Finally, it is worth stressing again that although the Blue Fund is proposed here as a new fund, ideally, it could represent a sub-fund of a broader institution and financial mechanism that I have named the Natural Capital Fund, recognizing the linkages between ecosystems and the need to increase the productivity of these type of funds by unifying loose initiatives into one that can be better managed.

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Conclusion

The concepts of natural capital and ecosystem services have been discussed in the literature for more than three decades, with an exponential growth in the early 2000s. They have proven to be an effective approach to make the intrinsically dependence that society has on healthy ecosystems evident, an ecological economics vision of the economic system embedded in the ecosystem. This conceptual framework has strengthened the argument, beyond moral reasons, of the urgent need to invest in natural capital conservation and restoration, as a development strategy that must complement the predominantly global myopic view of creating well-being focused on investments that are targeted in the great majority to the other types of capital (i.e. human, social and built).

From a global governance perspective, the dependence of society on nature is also clear by analyzing the Sustainable Development Agenda and its Sustainable Development Goals, adopted by all countries in 2015, in which Goals 15 (Life on land), 14 (Life below water), 6 (Clean water and sanitation) and 13 (Climate action) constitutes the basis to achieve the rest of the 13 goals, following the same ecological economics paradigm described before.

In order to advance in the implementation of this natural capital framework at a policy and research scale in Costa Rica, this research was able to help close the knowledge gap on the economic value of ecosystem services in the country, first by estimating its present and future value under four plausible development scenarios, then by exploring the value of terrestrial and coastal ecosystems at national and local scale. I was able to conduct the first study ever done on Ramsar Sites at a national scale, the most comprehensive study on the value of mangroves at a local scale, and the first estimate of the economic value of mangroves at the national scale. Moreover, I explored for the first time in the literature, a methodological comparison between valuation methods for mangroves, making evident the difference in the quality of results by applying aggregate methods (e.g. benefit transfer) and conducting primary studies.

It is worth noting that these studies were conducted in collaboration with scientists from many fields, such as economics, geography, ecology, engineering and forestry, among others, making clear that research on ecosystem services require a transdisciplinary approach to be able to incorporate and analyze the wide arrange of biophysical and economic variables that must be

taken into account to properly estimate the value of natural capital. This transdisciplinary work promotes scientists to expand their field of research, and more importantly, to leave their research silos to produce knowledge considering a more complex and systemic vision of how society and the environment interact.

Moreover, a collateral benefit of this research was the capacity building it represented for all the social actors that participated in one way or another in the valuation studies cited before, including local communities, non-governmental organizations, academia and the government, providing them the opportunity to learn about the application of economic valuation methods and its relevance for the environmental management they do. Many ideas emerged from the dialogues with these stakeholders, which hopefully can be developed in the near future, such as the estimation of a per hectare value of ecosystems in Costa Rica as a tool for environmental impact assessments, the need to conduct similar valuation studies in marine and coastal ecosystems such as coral reefs, and to propose a new PES scheme with a broader scope (including public forests and other ecosystems), among many others.

Regarding this last idea, this thesis made a substantial contribution to the creation of a new payment for ecosystem services scheme for marine and coastal ecosystems, with a focus on mangroves and coral reefs, the first of its kind developed for Costa Rica. This scheme can also represent a complement or contribution to the current PES scheme, because it provides key elements that had been ignored before, such as the principle of additionality, lower transaction costs, and a direct relation between threats, funding sources and solutions, and the economic value of ecosystem services. This specific piece of research addressed the issues related to managing natural capital, especially when they are public property, as in the case of marine and coastal ecosystems, which requires a different governance system, as well as specific instruments for its conservation and sustainable use.

The results of this research have numerous policy implications. The economic valuation of ecosystem services should be conducted to inform decision makers about the best management options to secure, as much as possible, the sustainability of social-ecological systems. In this regard, this thesis was able to produce new knowledge on economic and environmental information to support policies and strategies such as the Wetlands Policy, the

Climate Change Strategy, the Blue Carbon Strategy, the green accounts from the Central Bank under the System of Environmental-Economic Accounting of the UN, and the current development of the country's position on blue economy, among many others. Furthermore, this research, and specifically chapter 2 on the future of ecosystem services in Latin America and the Caribbean, offered a vision of natural capital under different development paths that the region can follow based on certain policy schemes, providing data for a dynamic policy making that takes into account the challenges imposed by the Anthropocene, instead of taking decisions based on the current assumption that the future is stable.

The overarching research question of this thesis was fully addressed, the five papers closed the gap on the value of ecosystem services in Costa Rica, assessing a wide arrange of terrestrial, marine and coastal ecosystems, as well as offering financial and management schemes to protect them and use them in a sustainable way. Future research should focus on getting a deeper understanding on the specific ecological features that sustain ecosystem services in Costa Rica, estimating biophysical and economic values of services based on primary studies in order to have a higher level of certainty, and assess the country's policy framework to identify enabling or restrictive conditions to implement successful financial mechanisms such as the Blue Fund.

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