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Environment, Rural Development and Risk Management Division

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Innovation in economic analysis and evaluation approaches for coastal protection and restoration investments in the Caribbean

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January 2020

ABSTRACT

Coastal zones are among the most economically productive areas of the world. However, they are also among the most vulnerable regions to disasters triggered by natural hazards. Recent recognition of the role of healthy coastal and marine ecosystems for reducing vulnerability in coastal communities has led to the design of coastal management strategies that incorporate direct investments in these ecosystems. However, there is a lack of knowledge and understanding of the economic benefits of coastal and marine ecosystems for society, which has led to the degradation of these ecosystems and hindered the prospects of sustainable investments in coastal resilience projects, including green infrastructure.

In this paper, we analyze the economic importance and ongoing threats of the main marine and coastal ecosystems of the Wider Caribbean region, and identify the underlying economic causes of their deterioration. The need to improve coastal resilience in the Wider Caribbean has led to innovative approaches for the protection of coastal zones and their population from erosion and flood risk, prioritizing the role of marine and coastal ecosystems for coastal protection and vulnerability reduction in coastal communities.

Based on this review, we develop an analytical framework for economic analyses and impact evaluations of coastal restoration and protection programs, with the objective of allowing practitioners to properly identify the cost-effectiveness of nature-based solutions for coastal resilience.

Keywords: Economic analysis, economic valuation, impact evaluation, coastal ecosystems, integrated coastal zone management, Wider Caribbean.

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INTRODUCTION

All countries in the Caribbean depend on coastal and marine ecosystem services such as tourism, fisheries and shoreline protection. However, coastal and marine ecosystems are rapidly degrading, in part because of lack of knowledge about the economic and social relevance of maintaining such ecosystems. Economic valuation can contribute to better-informed decision making about coastal use and development, as it allows not only for the identification of the ecosystem services provided, but also for the assessment of their benefits. Valuation can be a tool for informing holistic decision making around development, planning, conservation, and the provision of public goods and services (Waite et al., 2015). Cost-benefit analysis (CBA) and impact evaluation (IE) are also economic tools that can help to better understand the benefits provided by ecosystem services when natural infrastructure projects are proposed to enhance coastal resilience.

In this framework, the main goal of this document is to support and disseminate emerging knowledge and methods for projects related to climate-resilient integrated coastal zone

management, ICZM, economic assessments and impact evaluations in the Wider Caribbean Region, as well as mainstream the practice of carrying out cost-benefit assessments of projects as well as design impact evaluation strategies during the initial stages of interventions across Wider Caribbean countries.

The document starts with the conceptualization of the Wider Caribbean, and a description of the main marine and coastal ecosystems present in this region, including their definition, geographical distribution, provision of ecosystem services and their social and economic relevance, as well as the threats and trends they are subject to in terms of their conservation.

After that, there is a discussion on the underlying economic causes of the deterioration of coastal ecosystem services, emphasizing the concept of market failures (public goods, common-pool resources, externalities and asymmetric information) and their relationship with deterioration and depletion of ecosystem services. We also introduce the concept of Total Economic Value (TEV). In order to complement this set of ideas, we also include a discussion on the concepts of Integrated coastal zone management (ICZM) and natural infrastructure.

We present the main methods for the economic analysis of coastal natural-infrastructure projects, including cost-benefit analyses (CBA), with particular emphasis on the relevance of applying economic valuation techniques as a tool to assess the value of the benefits that ecosystem services provide. We propose a step-by-step scheme for performing economic valuation, including a review of the main models used for the quantitative estimation of changes in ecosystem services as well as the main methods for economic valuation. We examine a number of studies on economic valuation of coastal protection and other ecosystem services, focusing on the Wider Caribbean, is also included. Finally, there is a description of impact evaluation (IE) as an ex-post tool for the economic analysis of coastal protection and restoration projects. Besides explaining the main concepts around IE, we review the different approaches that can be applied to assess natural infrastructure projects.

The document ends with the challenges, recommendations, and innovations in the economic analysis of coastal restoration and protection projects in the Wider Caribbean.

I. STATE OF COASTAL AND MARINE ECOSYSTEMS IN THE WIDER CARIBBEAN

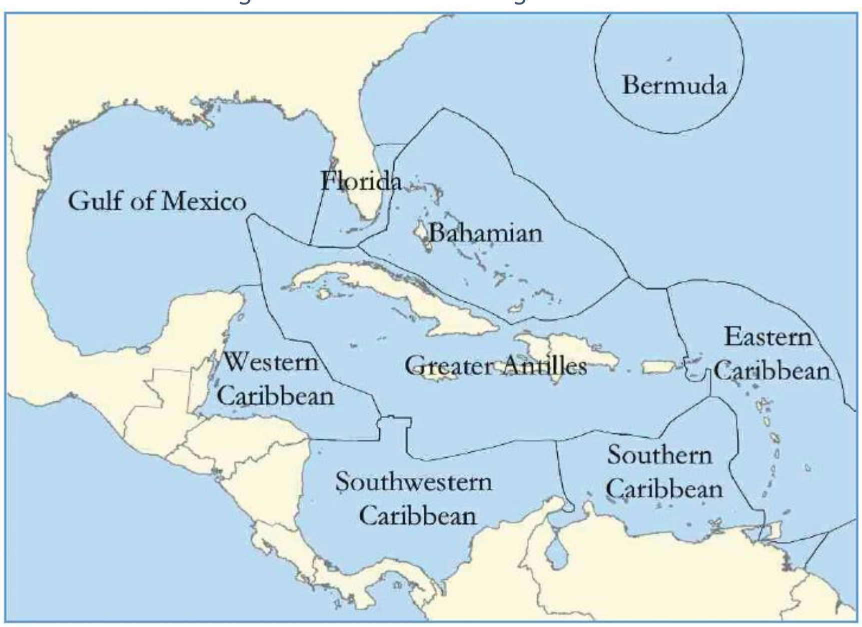
The Wider Caribbean is the geographic region that comprises the territories with coasts on the Caribbean Sea and the Gulf of Mexico, and the waters of the Atlantic Ocean adjacent to these territories. This region encompasses approximately 5,326,000 km^2 (UNEP, 2008), and it is divided in 9 sub-regions (Figure 1) that include 29 island territories[1], 14 of which are independent, and 10

[1] Bahamas, Turks and Caicos Islands, Anguilla, Antigua and Barbuda, Barbados, British Virgin Islands, Grenada, Guadeloupe, Martinique, Montserrat, Netherlands Antilles North (St. Maarten, St. Eustatius and Saba), St. Kitts and Nevis, St. Lucia, St. Vincent and the Grenadines, Trinidad and Tobago, Virgin Islands (US), St.

continental territories[2] (CARSEA, 2007; UNEP, 2014). About 41 million people inhabit this region within 10 km of the coastline, thus exhibiting some degree of dependence on coastal and marine ecosystems (Burke & Maidens, 2004). However, these ecosystems in the Caribbean are fragile and are being threatened by different anthropic activities.

The most relevant ecosystems found in the Wider Caribbean are coral reefs, mangrove forests, seagrasses, beaches, sand dunes, oyster reefs and wetlands (lagoons, swamps, marshes and estuaries) (CARSEA 2007; Vaslet & Renoux, 2016). In the following section, we describe their main characteristics, including their location, the services they provide[3], their economic and social importance, their status and the threats they are exposed to. We emphasize coastal protection ecosystem services, the mechanisms through which ecosystems provide these services and the determinants of their effectiveness. In particular, erosion and flooding control are considered here to be the most important coastal protection ecosystem services. Hence, when we talk about coastal protection, we refer to these two ecosystem services.

Figure 1. The Wider Caribbean Region and its nine sub regions.



Source: Burke & Maidens (2004)

Barthelemy, St. Martin, Bermuda, Cayman Islands, Cuba, Dominica, Dominican Republic, Haiti, Jamaica, Puerto Rico, Navassa Island, Aruba, Netherlands Antilles South (Bonaire and Curaçao).

[2] United States, Mexico, Belize, Guatemala, Honduras, Venezuela, Panama, Costa Rica, Nicaragua, Colombia.

[3] According to the Millennium Ecosystem Assessment (MEA, 2005), ecosystem services are defined as the benefits people obtain from ecosystems. This is a key concept as it specifically foregrounds the ecological functions of ecosystems and their relationship to the wellbeing of society.

Coral reefs

The Wider Caribbean has approximately 26,000 km^2 of coral reefs, which corresponds to 7% of the global total (Paulay, 1997; Spalding et al., 2001), an area equivalent to the Bahamas, Trinidad and Tobago and Puerto Rico combined. This ecosystem provides numerous services. Particularly within regulating services, those related to coastal protection are the most relevant in this region. Coral reefs are the first line of defence against shock events (Spalding et al., 2014a). On the one side, as a result of the production of large amounts of calcium carbonate, reefs generate sediment (accretion) and adjust their height in response to the increase in sea level (World Bank, 2016), reducing wave energy and height by 97% and 84% respectively (Ferrairo et al., 2014). On the other side, these ecosystems help marine and terrestrial sediments bind, reducing routine erosion, lessening flooding and wave damage during storms, and supporting clear offshore waters that are favourable to corals (Burke & Maidens, 2004).

According to Burke & Maidens (2004), the value of coastal protection by coral reefs in 2000 was between US$750 and 2,180 million in the Wider Caribbean, based on costs required to replace them by artificial means. Despite its importance, the average hard coral cover on reefs has declined by 80% in the Caribbean since 1970 (Gardner et al. 2003) due to coastal development, deforestation, poor agriculture practices and overfishing. This problem is exacerbated by pollution, sedimentation, nutrient over enrichment, and invasive species affecting Wider Caribbean coastal waters (Regional Activity Centre for the SPAW Protocol & Réserve Naturelle Nationale de Saint-Martin, 2016; Doney et al., 2012; Field et al., 2012; Jackson et al., 2014). Climate change threats, such as sea level rise and higher temperatures, increase the pressure on this ecosystem (Burke & Maidens, 2004; Cesar, 2000; Doney et al., 2012).

Mangroves

Mangroves are forests formed by trees and shrubs that have adapted to survive in coastal zones. They are complex and adaptative ecosystems that create an environment for different organisms. Although a single estimate of mangrove area in the wider Caribbean is difficult to obtain, mangroves in countries from the Central and North American region occupy an area of 22,404 km^2 (14% of total global coverage), while in South America this area has been estimated to be approximately 23,882 km^2 (15.7%) (Spalding et al., 2010).

Moreover, mangroves have the ability to provide multiple ecosystem services, including coastal protection, since they act as barriers from natural hazards and can stabilize sediments, which helps reduce erosion. For instance, a mangrove forest with a width of 100 m can reduce wave height by 13 to 66%, and a mangrove with a width of 500 m can reduce over 50 to 99% of wave height (McIvor et al., 2012a). Additionally, McIvor et al. (2012b) suggest that every kilometer of mangrove forest can reduce water levels by up to 50 cm during storm surges. A single estimate of coastal protection provided by mangrove in the Wider Caribbean is not available; however, studies in other countries such as Thailand have found a value of US$ 8,966-10,821 per hectare for storm protection services (Barbier, 2007) and US$3,679 per hectare per year for erosion control (Sathirathai & Barbier, 2001).

Clear-felling, aquiculture, urbanization and human population growth, among other threats, directly and indirectly affect this ecosystem to the point where around 25% of the global mangrove cover has disappeared (Spalding et al., 2010).

Seagrasses

Seagrasses are submarine flowering plants growing in mud and sand that require high levels of light (den Hartog & Kuo, 2006; Duarte, 1991; Ondiviela et al., 2014; Orth et al., 2006). Although they are characterized by a low diversity of species, they can grow over a wide latitudinal range. The seagrass area in the Wider Caribbean is approximately 68,157 km^2, equivalent to the size of Ireland (Creed et al., 2003, Green & Short, 2003, Zarate-Barrera & Maldonado, 2015).

With regard to coastal and shoreline protection services, seagrass capacity to stabilize soil and bind sediments plays an important role in erosion control. Additionally, seagrasses provide costal protection through wave attenuation, although in less magnitude than other ecosystems (Barbier et al., 2011; Ondiviela et al., 2014; Spalding et al., 2003; Waite et al., 2014). The wave attenuation capacity of seagrasses is limited and highly dependent on several key characteristics: first, it is restricted to shallow areas and low-wave environments (Barbier et al., 2011; CGIES Task Force, 2015; Ondiviela et al., 2014). This implies that seagrasses can protect the coast from regular waves and tidal changes but would be less effective for other more drastic climatic events. In addition to wave attenuation, it is also important to consider the sediment stabilization service provided by seagrasses, which, according to Christianen et al. (2013), is an indirect mechanism for coastal protection. Sediment stabilization occurs when roots and rhizomes bind together sediments and other organic matter (Spalding et al., 2003). These services provided by seagrasses need yet to be evaluated.

Seagrasses are not exempt of natural and anthropogenic threats. In fact, some of the major threats to this ecosystem are coastal development and the runoff of inland nutrients that pollute the environment where seagrasses live in (Green & Short, 2003). Barbier et al. (2011) have estimated a loss of 29% of seagrasses worldwide.

Coastal sandy environments

Beaches and dunes are representative sandy environments in the Caribbean. The coastline of the Wider Caribbean is 86,930 km in length (Burke & Maidens, 2004). This represents 24% of the world's total coastline length. Although tourism is the most visible ecosystem service provided by beaches, they also play a role as energy sinks, attenuating and dissipating waves, and stabilizing and accumulating sediments (Cambers, 1998; Hanley et al., 2014). In the Wider Caribbean, tourism activities are primarily located close to the coast and are heavily dependent on the tropical climate and the presence of sandy beaches and scenic coastal areas with clear seas free from pollution and abundant marine life (Cambers, 2009). Tourism in the Caribbean is very important and accounts for 5% to 85% of GDP in some countries and territories. Despite their economic value in the Wider Caribbean, beaches have unfortunately not been perceived as areas in need of management,

protection and funding, but rather as permanent features of the landscape (Cambers, 1998). Sand mining, several hard defenses, and climate change have adversely affected beach ecosystems (Cambers & Diamond, 2010).

As for dune systems in the Wider Caribbean, there is not accurate information on regional coverage, but they are located in the Gulf of Mexico, the western Caribbean and Greater Antilles. Dune ecosystems used to be sand mining sources, but what may be the most relevant socioeconomic service provided by coastal sand dunes is the absorption of energy from wind, tide and wave action (Bridges et al., 2015; Everard et al., 2010). There is limited data in literature to quantify the economic importance of these ecosystems in the Wider Caribbean in the context of coastal protection. In fact, we have only identified one economic valuation study for Mexico focusing on dunes. González et al. (2012) estimate a willingness to pay in 2010 of $65,743 ha/year. In spite of its importance, sand dunes continue being destabilized by various income-producing activities such as construction, tourism, recreational activities, farming and mineral extraction, which eliminate plant cover, as well as climate change (Hanley et al., 2014, Pontee, 2013, Hanley et al., 2014; Moreno-Casasola, 2006).

Oyster reefs

Oyster reefs are structures formed by species from the Crassostrea genus, known for their ability to construct vertically (towards their growing edge) and form dense groups (Bahr & Lanier, 1981; Grabowski et al., 2012). There is not information about their extension around the world nor in the Wider Caribbean, but it is indisputable that they play a crucial role in shoreline protection. Oyster reefs act as natural breakwaters by attenuating wave energy and promoting accretion, which can stabilize the shoreline –as well as salt marshes– from erosion and sea level rise, as they can grow at a rate which exceeds that of the water level (Coen et al., 2007; Grabowski & Peterson, 2007). This is possible for the tridimensional structures that these organisms construct, altering the bathymetry and water flows (Borsje et al., 2011; La Peyre et al., 2015; Stricklin et al., 2010), contributing to shoreline stabilization and erosion control. They can reduce shoreline erosion by over 40% (Scyphers et al., 2011; Piazza et al., 2005). Even though oyster reefs are important for coastal protection, few studies have attempted to examine their economic value. In the Wider Caribbean, Kroeger (2012) estimates that, by 2030, between US$3.0 and US$7.5 billion in economic losses would be avoided by oyster reefs every year . However, oyster reefs have rapidly declined over the years. Beck et al. (2009) estimate that, overall, 85% of this ecosystem, located in bays and ecoregions around the world, has been lost. Factors such as invasive species and overexploitation have contributed to the deterioration of this ecosystem.

Water bodies (Wetlands, Salt marshes, Estuaries, Coastal lagoons)

Water bodies include a variety of environments such as coastal wetlands, salt marshes, estuaries and coastal lagoons. Due to their varied nature, it is difficult to give a precise estimate of their extension in the Wider Caribbean. However, according to the Ramsar Sites Information Service,

there are 32,324 km^2 of marine or coastal wetlands in the countries that belong to the Wider Caribbean region (Ramsar Convention Secretariat, 2018).

Water bodies are important for their capacity to provide ecosystem services, among them the provision of food (de Groot et al., 2002), costal protection (ARCH, 2015), water quality (Horinko Group, 2015), tourism and habitat for a diverse number of species (Anthony et al., 2009). In terms of coastal protection, wetlands reduce flood peaks and regulate water flows (Silva et al., 2017). In addition, they act as a natural buffer for waves, and decrease the damages endured by coastal communities in proximity to the shoreline (Ruckelshaus et al., 2016). Similarly, salt marshes reduce flood peaks and wave energy (Shepard et al., 2011). The social and economic importance of these ecosystems can be characterized by the large number of people living close to these areas. As an example, 62% of major estuaries can be found near urban centers (UNEP, 2006). Moreover, studies suggest that their economic benefits are substantial. Costanza et al. (2008), for instance, value storm protection services provided by wetlands in the United States at US$23.3 billion per year. However, water bodies are subject to threats such as pollution form a variety of sources like industry, transport and agriculture that are degrading these ecosystems.

II. UNDERLYING ECONOMIC CAUSES OF THE DETERIORATION OF COASTAL ECOSYSTEM SERVICES

Despite of the relevance of coastal and marine zones, they face, as described in the previous section, an increasing rate of deterioration, caused by a wide variety of factors and groups of stakeholders who demand different types of goods and services provided by these ecosystems.

Economically, the deterioration of ecosystems and the depletion of the services they provide can be explained as the result of private exploitation at a rate that is not socially optimal. Economic theory suggests that markets allocate scarce resources efficiently. Under specific conditions, markets would allow to attain optimal allocation: First, there has to be many buyers and many sellers of the good or service; second, any agent participating in the market can freely enter or leave it at any time; third, there has to be adequate information available about the good or service that is being exchanged, including all of the benefits or costs that accessing it can generate; fourth, the rights for the use and exploitation of this good or service must be clearly defined, well known, and respected; and fifth, the use and exploitation of the good or service must not generate external consequences for other agents outside of the market. When these conditions are met, the market is defined as perfectly competitive and efficient in terms of resource allocation.

However, for many ecosystem services there are no markets in which they can be exchanged, and even when an ecosystem service is traded in a market, some of above conditions may not be met. Ecosystem services, particularly those provided by marine and coastal zones, are characterized by the presence of multiple market failures, such as the absence of clearly defined property rights, the existence of externalities, and the presence of asymmetric information.

Property rights

Any good or service, including ecosystem services, can be characterized by two fundamental properties: *exclusion* and *rivalry*. Exclusion refers to the extent to which it is possible to decide who receives the benefit of access to a good or service and who does not. When it is not viable to determine who accesses a good or service (or to prevent individual users from accessing it), it is said that the good is non-excludable. What defines whether a good is excludable is the marginal cost associated with creating such an exclusion; the higher the cost, the more difficult it will be to make the good excludable. Exclusion is closely linked to the concept of property rights. Property rights are defined as an authorization to which compliance can be enforced, and that can be granted to an individual, group of individuals, or the State in order to carry out particular actions in relation to a specific domain (Ostrom & Schlager, 1996). When property rights can be easily allocated and compliance with the allocation of rights enforced, exclusion will be viable. Many ecosystem services, such as coastal protection, however, are non-excludable, as they can be freely accessed. This is a leading cause of their inadequate provision and overexploitation.

Rivalry refers to the relationship between the consumption of a good or service by an individual and the possibility of its simultaneous consumption by other agents. When a good is non-rival, the marginal cost of providing it to an additional person is zero.

The combination of the properties of rivalry and exclusion allows to characterize the goods and services into four categories:

i. private goods: Easy excludability, high rivalry.
ii. club goods: Easy excludability, low rivalry.
iii. public goods: Difficult excludability, low rivalry.
iv. common-pooled resources (CPR): Difficult excludability, high rivalry

Table 1 shows some examples of this classification applied to ecosystem services in the Caribbean).

Many of the ecosystem services provided by marine and costal zones are either public goods or common-pooled resources –CPR. Given that for public goods there is no exclusion, it is not easy to assign property rights and thus markets are not an effective mechanism to allocate such resources. Specifically, since exclusion is impossible, people will be tempted to use them without paying for their consumption, and, as there is no way to exclude non-paying consumers, there will be fewer people willing to pay in the market than expected. This phenomenon of consumers who do not pay for the provision of goods or services that they use is known as the free-rider problem (Glover, 2010). On the other hand, CPR are such goods and services that are non-excludable but are rival (Feeny et al., 1990; Ostrom, 1990). A typical example of CPR are fisheries: anyone interested in participating in the activity can do so (there is no exclusion), but once he or she catches a fish, it becomes a rival good as no one else can catch the same fish and the overall supply of fish is reduced. The problem with CPR is that all users are tempted to use more of the good or service, as they cannot be sure that other users would be cautious of *their* use of it. This behavior leads to the

overexploitation and exhaustion of the resources, as the total quantity extracted exceeds the efficient and sustainable use (Ostrom & Schlager, 1996). This phenomenon was first identified by Hardin (1968) as the tragedy of the commons, which exemplifies that the lack of property rights for a rivalrous good leads to overexploitation, very low efficiency, and a critical situation for the resource itself, and for those that benefit from it.

Table 1. Classification of some marine and coastal ecosystems goods and services according to their properties of rivalry and exclusion

Type of good or service	Examples of ecosystem services	Examples in the Wider Caribbean
Private goods	• Fishing for commerce and self-consumption, including near-shore and offshore fisheries. • Medical resources. • Tourism and recreation. • Wonderment, excitement, and adventure.	• Aquaculture production in the Caribbean, including land-based aquaculture of tilapia (Oreochromis sp.) and coastal pond aquaculture of white-legged shrimp (Litopenaeus vannamei) (Lovatelli et al., 2013). However, there is a production potential of offshore mariculture in the Caribbean (Lennon et al., 2019). • Pseudopterosins are tricyclic diterpene glycosides isolated from the Caribbean sea whip (gorgonian). Pseudopterogorgia elisabethae (Gorgoniidae) have been licensed for medical use as a potential anti-inflammatory drug. • Beach recreation.
Common-pool resources	• Fishing for commerce and self-consumption, including near-shore and offshore fisheries. • Non-food resources (aquarium fish), ornamentation. • Timber. • Recreation and tourism.	• Open-access fisheries in the Caribbean. For example, landings have expanded considerably and many fishery resources are fully exploited and overexploited (Valle et al., 2011). • Marine fish, corals, and other marine invertebrates are in the ornamental (aquarium) trade. The majority of these species are collected from the wild in Indo-Pacific and Caribbean regions (Livengoo and Chapman, 2014). • In countries like Nicaragua, nearly 80% of households use firewood for cooking; mangroves are a major source of wood for such purposes. • Public Caribbean beaches.
Club goods	• Recreation and tourism. • Non-food resources (aquarium fish), ornamentation. • Fisheries.	• Entry fees at natural marine parks are usual in several marine Caribbean areas. Corales del Rosario, the San Bernardo Islands and Tayrona National Natural Parks in Colombia are examples. • In Belize, there have been several companies involved in the exports of Caribbean reef fish to supply trade. Companies are required to have permits for operations. • Fishing rights in St. Lucia with respect to recreational and artisanal fisheries (Leria, 2016).
Public goods	• Recreation and tourism. • Coastal protection against erosion and flooding. • Carbon sequestration. • Mangrove nurseries. • Nutrient cycling. • Biodiversity.	• Public legal status of many beaches in the Wider Caribbean. • "Forest Law No. 7575" in Costa Rica that completely prohibits the extraction of wood and the exploitation of mangroves, guaranteeing their provision of regulation services.

In summary, most marine and coastal ecosystem services are public goods or common-pool resources. That is, they are subject to exclusion problems, which, implies that private-use decisions made by consumers tend to shift the market outcome away from what is socially desirable. In particular, regulation services, such as coastal protection, are subject to problems of exclusion and rivalry, which – typically for a public good – means that not many people are interested in paying to maintain their provision. This situation is exemplified by the high level of deterioration of coastal ecosystems due to human activities (Barbier et al., 2011).

Externalities

An externality is defined as the external effect that production or consumption decisions unintendedly generate on third parties and for which there is no compensation (Glover, 2010; Baumol & Oates, 1988). For an externality to exist, two conditions must be fulfilled: first, the action of one agent imposes an effect on another and this effect is not considered within the market, and 2) no compensation is provided for this action. The occurrence of an externality implies that the costs and benefits for the entire society, including those agents participating in the market and those who are not, are different from the costs and benefits taken into consideration in private decision-making. In such conditions, market equilibrium is no longer efficient, and resources are not allocated optimally.

There are two forms of externalities: positive (when the external effect generates benefits for someone else), or negative (when the effects generate additional costs). They can arise from production or consumption decisions and affect other production or consumption activities.

Externalities can be found in marine and coastal areas. Urban development decisions that do not consider the effects on ecosystems, water pollution, overfishing, catching fish below the minimum size, mangrove deforestation and sand extraction, among others, are activities that create externalities. Similarly, natural infrastructure activities or ecological restoration efforts can generate positive externalities by improving the flow of other ecosystem services.

Asymmetric information

Another market failure that affects the adequate allocation of ecosystem services is the lack of comprehensive information on the benefits of these services for society. This problem, known as information asymmetry, means that, when deciding about the best possible –and sometimes exclusive– use of ecosystems, there is only partial information available regarding their benefits and costs of their degradation. This is particularly relevant in the case of ecosystem services, for which there has been historically insufficient information on the entirety of their benefit and cost flows beyond the physical quantities of the services provided, and no information about the magnitude of their importance for society. In other words, society in general has limited information on their economic value, and societal decision-making processes assign an economic value close to zero to such ecosystem services as a result, making it difficult for these services to compete with other activities or uses that preclude them. In other cases, the physical values of these ecosystem services

are completely unknown, making it more complicated to compare in terms of assigned monetary value to alternative uses.

When the benefits of preserving an ecosystem are not taken into account, policies tend to favor the creation of positive incentives to stimulate other sectors that generate a greater benefit for the society but have perverse or negative incentives to conservation. Being unaware of the benefits that nature can bring to society makes it difficult to include this value in the country's flow of assets, thus ignoring its importance and the losses that would be produced by the deterioration of such ecosystems. In this sense, economic valuation can play an important role by bringing these values to the discussion and the decision making process.

Policy failures

In addition to market failures, marine and coastal ecosystem services face what is known as policy failures, which involve the policy making process itself. They refer to the design and implementation of incentives that induce behaviors that negatively affect either ecosystems or the services they provide. Most of those incentives, known as perverse incentives, are unanticipated side effects of policies designed to attain other objectives in other sectors.

Perverse incentives affect natural environments, in part because of the above-mentioned market failures. Not recognizing the externalities produced by ecosystems due to lack of information or the absence of clearly defined property rights may lead to the creation of policies that favor other activities and, as a result, affect the conservation of natural environments and the ecosystem services they provide.

Market failures and policy failures prevent an efficient allocation of resources, including ecosystem services. By ignoring these failures, the benefits that ecosystems provide are underestimated, leading to their overexploitation and deterioration.

III. INTEGRATED COASTAL ZONE MANAGEMENT (ICZM) AND NATURAL INFRASTRUCTURE

Coastal management as a concept was originally coined in the 1960s (Clark, 1996; Sorense, 1997, as cited in Alonso et al., 2003; Sorensen and McCreary, 1990). However, it was until the 1970s and the 1980s that the implementation of coastal management strategies, particularly in the United States, took place (Saffache and Angelelli, 2010; Post and Lundin, 1996).

Global recognition of integrated coastal zone management, also known by its acronym ICZM[4], was gained when the United Nations Conference on Environment and Development of Rio de Janeiro in 1992recognized the pressures and threats to coastal zones. With the goal of minimizing coastal exploitation as well as ensuring the sustainable use of coastal resources, the Conference highlighted the relevance of ICZM and proposed the design of guidelines for its implementation (Post and Lundin, 1996). At the Rio Conference, the concept of ICZM was integrated as one of the key elements for work on marine and coastal ecosystems (Secretariat CBD, 2015) and urged coastal nations to develop ICZM plans by the year 2000 (Lemay, 1998).

Compared to previous coastal management exercises, ICZM seeks a more comprehensive approach, since it incorporates the different coastal ecosystems, all of the sectors that affect the coastal zone, and all of the stakeholders involved. As a result, it combines the economic, social and environmental aspects, seeking to reduce conflict and harmonize activities in order to reach national goals for these zones (Post and Lundin, 1996). The comprehensive nature of ICZM implies a strong institutional perspective, as it requires a governance scheme based on rules that regulate all the different actors involved in the use of coastal zones.

In the literature, it is possible to find several definitions for ICZM, which have evolved over time. It is important to clarify what we understand as ICZM. In this paper, we adopt the following definition:

> ICZM is a governance process that promotes the sustainable use of coastal resources by developing a multisector management that is continuous, holistic, harmonic, balanced, adaptive and dynamic, and uses the ecosystem approach to simultaneously favor coastal development, minimizing its negative effects, and guarantee the provision of natural coastal ecosystem services over time (Chua, 1993; Knecht and Archer, 1993; Pot & Lundin, 1996; Sorensen, 1993). The ecosystem approach of ICZM recognizes the dynamic and interconnected nature of the coastal and marine systems, as well as the interactions among them and their different uses (Adapted from Post & Lundin, 1996; Secretariat CBD, 2015).

The different definitions of ICZM highlight its multisector, multi-scale and holistic nature. As a result, ICZM requires institutional coordination and local community participation for the development of diagnoses of the state of the marine coastal zones, as well as strategy formulation and implementation. In Latin American and Caribbean countries, ICZM has manifested in the form of the establishment of protected areas, land use zoning, hybrid and nature-based infrastructure development, and property right allocation for mangroves and fisheries. Adaptive and community-based management, participatory processes, and property rights allocation for access and use are also promising strategies for LAC coastal zones.

[4] Integrated coastal zone management is also known as integrated management of marine and coastal areas, and coastal integrated management, among others.

The importance of ICZM for the achievement of sustainable economic development is reflected in the strategies of multilateral organizations. For example, the IDB, in its Eighth General Increase of Resources Report (1994), recognized the need for interventions in the LAC marine and coastal zones that support the conservation and management of their resources. The Bank acknowledges that ICZM is critical to manage, invest in, and allocate marine and coastal resources in a sustainable fashion. Over recent decades, IDB investments in the Wider Caribbean have focused on ICZM approaches to coastal planning, including natural and hybrid infrastructure. In this way, the IDB does not only require the incorporation of ICZM elements into the loan requests of the countries in the region but has also directly funded initiatives of coastal management.

Additionally, in the context of climate change, IDB has supported adaptation strategies, particularly in small-island states (Banerjee et al., 2018; Lemay, 2016; Saffachen & Angelelli, 2010). IDB projects in countries such as Barbados, the Bahamas, Belize, Trinidad and Tobago, and the Dominican Republic have incorporated ICZM dimensions, such as the development of coastal risk assessments, including monitoring and risk management; the development of green or hybrid infrastructure or the restoration of ecosystems to stabilize coasts, reduce erosion and flood risks, enhance public access and associated economic activities mainly related to the tourism sector; and the development of institutional capacity for integrated risk management in coastal zones (Lemay, 2016). Under this approach, these ICZM projects seek to reduce the vulnerability of coastal communities and protect or enhance the resilience of marine and coastal ecosystems and sustainability in the provision of ecosystem services.

Natural infrastructure as a strategy for ICZM

Coastal zones attract people. Temmerman et al. (2013) estimate that at least 40 million people and US$3,000 billion of assets are located in coastal cities exposed to flooding. By 2070, these figures could increase to 150 million people and US$35,000 billion. In the Latin American and Caribbean region, 32.6 million people were living in low elevation (<10m above sea level) coastal areas in 2010. The coastal population is expected to grow to over 96 million by 2100 (Silva et al., 2017).

Evidence shows that the impact of climate change is especially relevant in coastal zones. The main concerns relate to the fact that global warming leads to changes in the weather, water temperature and sea levels, which affect coastal zones by increasing the frequency, intensity, duration and size of storms. The consequences for the coast include more severe disasters, coastline erosion, flooding of lowlands and the deterioration and loss of marine and coastal ecosystems (IPCC, 2014; Post and Lundin, 1996; Saleh and Weinstein, 2016; Spalding et al., 2014a).

Efforts to enhance coastal resilience have traditionally relied on conventional coastal engineering that focuses on hard (gray) infrastructure to protect public and private assets facing great risks from floods and storm surges (Spalding et al., 2014a; Temmerman et al., 2013). Nevertheless, hard infrastructure poses high maintenance costs, particularly if they are going to be redesigned to withstand increasingly intense catastrophic events. Moreover, the use of hard infrastructure as a line of coastal defense has been questioned for its negative ecological effects.

The need to improve coastal resilience, while simultaneously addressing the issue of climate change, has led to innovative approaches for the protection of coastal zones and their population from erosion and flood risk (Ondiviela et al., 2014; Saleh and Weinstein, 2016). In recent years, the role of marine and coastal ecosystems for coastal protection and vulnerability reduction in coastal communities has been recognized. The inclusion of coastal ecosystems for coastal adaptation planning has recently been proposed as a result (Spalding et al., 2014a) in what is generally known as green or nature-based infrastructure. We refer as "natural infrastructure" to all green or nature-based infrastructures. Specifically:

> Natural infrastructure refers to all the natural, semi-natural or artificial constructions that mimic natural systems and processes and contribute to the conservation or restoration of biological diversity and the enhancement of ecosystem services, and at the same time support communities, economies and environmental resilience (CGIES Task Force 2015; Silva et al., 2017).

Natural infrastructure involves the protection of natural systems, as well as their restoration and rehabilitation. Typical examples of natural infrastructure include mangrove reforestation, the establishment of vegetation cover in wetlands, beach and dune nourishment, movement of dredged sediments from and to wetlands, coral reef restoration, and shoreline stabilization , among others (Beck, 2014; Saleh and Weinstein, 2016; Silva et al., 2017).

Compared to gray infrastructure, natural infrastructure has the ability to self-maintain, and offers the potential for self-repair after damaging events and the ability to adapt to climate change with minimum human intervention (e.g., maintain the same rate as sea level increase), with lower costs of construction and maintenance while simultaneously generating a range of co-benefits (Ruckelshaus et al., 2010; Sutton-Grier, 2015). Furthermore, some authors argue that natural infrastructure utilizes less basic raw materials, and, importantly, favors sustainability in the provision of other additional ecosystem services aside from coastal protection, thus achieving social, environmental and economic objectives (Temmerman et al., 2013; van der Nat et al., 2016; CGIS Task Force, 2015; Sutton-Grier et al., 2015). Some of these services are precisely the reasons why people are attracted to live in coastal zones.

Although the creation or restoration of natural coastal systems provides a good alternative for conventional coastal protection, the viability and effectiveness of nature-based approaches depend on various factors, such as the type of coastal zone to be protected, the location of the population at risk, the particular characteristics of the ecosystems involved in the scheme of coastal protection, and the characteristics of the specific event threatening these zones (Spalding et al., 2014b; Temmerman et al., 2013; World Bank, 2016).

When purely natural infrastructure is not feasible, it is possible to combine gray and natural infrastructure designs. Under this hybrid approach, the constructed infrastructure is installed simultaneously with natural infrastructure such as wetlands or oyster reefs. The natural

infrastructure provides key protection benefits for medium and small storms, whereas grey infrastructure can provide protection from larger events.

Nevertheless, one of the greatest limitations to mainstream the implementation of nature-based solutions for coastal protection is the lack of knowledge about the associated values of the services that natural coastal ecosystems provide.

Although the field of ICZM has made advances towards the comprehension of the physical and economic benefits of nature-based coastal protection interventions, a knowledge gap between the state of science and policy practice remains, thus limiting the implementation of these types of projects (Ruckelshaus et al., 2010). There are also gaps in the information required to develop rigorous analyses about the effects at a local scale. The importance of identifying and estimating the values of the benefits associated with natural coastal protection is such that the National Academy of Sciences of the United States suggested that in order to improve coastal resilience, it is necessary that all of the costs and benefits of the interventions of coastal infrastructure, including those of an environmental and social nature, be quantified and incorporated so that they can support better decision-making on the management of coasts (Sutton-Grier et al., 2015). With this in mind, we discuss approaches to conduct cost-benefit analyses, economic valuations, and impact evaluations for ICZM projects, including those that involve natural infrastructure, in the next section.

IV. ECONOMIC ANALYSIS OF COASTAL NATURAL INFRASTRUCTURE PROJECTS

Investing in natural infrastructure projects has emerged as a strategy within ICZM for coastal protection and climate change adaptation. Natural-infrastructure projects need to be at least as effective as conventional gray infrastructure projects in terms of coastal zone protection to be considered for implementation. Coastal protection effectiveness can be measured either in terms of capacity to reduce erosion and flood risk (see Table 2), or economic viability. Although much progress has been made to determine the biophysical and economic effects of gray infrastructure for coastal protection, the effects of natural infrastructure have only recently become evident. Although the costs of implementing natural infrastructure projects can be directly determined, an important challenge when adopting ecosystem-based approaches is to appropriately identify the benefits associated to improving the resilience of such intervened ecosystems and their surrounding area. Generally, when dealing with natural infrastructure, the benefits resulting from reduced erosion and flood risk are not reflected directly in markets, so they can be easily overlooked.

Table 2. Mechanisms through which some coastal ecosystems can provide coastal protection

Ecosystem	Mechanisms through which they reduce flooding and erosion
Coastal wetlands	Wave attenuation: Can attenuate energy of waves, tides, and currents Soil stabilization, sediment flow, sediment deposition and accretion Water flow and flood regulation
Mangrove forests	Wave and wind attenuation: Can attenuate energy of wind and waves Soil stabilization, sediment flow, capture of sediments

Ecosystem	Mechanisms through which they reduce flooding and erosion
	Water flow and flood regulation
Coral reefs	Wave attenuation: can reduce incident wave energy
Seagrasses	Wave attenuation: can reduce wave energy Soil stabilization and sediment flow
Beaches and dunes	Wave and wind attenuation: Can attenuate wave energy Soil stabilization and sediment flow

Source: CGIES Task Force, 2015

Also, by improving the conditions of the intervened ecosystems, natural infrastructure projects generate additional benefits, as they increase the provision of other multiple services by these ecosystems. We call these additional benefits co-benefits and they can be represented in other regulation services, such as carbon sequestration, or provisioning services, such as improvements in artisanal and commercial fisheries, as well as cultural services, such as improved possibilities for recreation and tourism.

We review the economic tools available to incorporate the flow of ecosystem-service benefits derived from natural infrastructure projects and assess their overall economic viability.

Cost-benefit analysis (CBA) is a central *ex ante* tool that offers information to decision-makers with regard to the economic viability of an intervention. An essential challenge of any cost-benefit analysis of natural infrastructure interventions is to identify the monetary benefits generated by ecosystems services, which are seldom reflected in observable market prices. An adequate CBA must quantify and value both the benefits generated by improved coastal protection and the co-benefits generated by the increased provision of other ecosystem services, in order to assess economic viability and compare the cost-effectiveness of a range of interventions. The additional benefits of natural infrastructure projects are considered positive externalities that must be appropriately assessed. To overcome this challenge, there are several methods of economic valuation of non-market goods and services. We analyze the main considerations that must be taken into account in order to carry out rigorous valuation exercises, as well as the main techniques available.

In addition to identifying whether natural infrastructure projects are economically viable or not, governments, multilateral banks, and investors as a whole, are increasingly interested in evaluating the economic, social and environmental impacts of these interventions. Impact evaluations (IE) identify which socioeconomic and environmental outcomes are attributable to a specific natural infrastructure or restoration project. We review the theoretical foundations of impact evaluations in the context of natural infrastructure projects for coastal protection. We also examine some of the different methodological approaches used to carry out impact evaluations for conservation and restoration projects, analyzing their advantages and disadvantages and the extent to which they allow us to measure the effects and attribute the changes observed to the intervention.

COST-BENEFIT ANALYSIS (CBA)

Cost-benefit analysis (CBA) is a tool for the appraisal of the economic viability of a project, program or policy. As its name suggests, the idea is to compare the benefits and costs associated with an intervention and compare them to any other alternative or the status quo. In the case of natural infrastructure, CBA determines ex ante if the benefits outweigh the costs relative to other alternatives, such as gray infrastructure, or a scenario without any intervention.

Boardman et al. (2006) define several steps to follow in any CBA. These steps and their relationship with natural infrastructure can be summarized as follows[5]:

1. Define the goals and objectives of the action or intervention, e.g., coastal protection for reducing erosion or flood risk.
2. List alternative interventions, including natural infrastructure and gray infrastructure, if appropriate.
3. Measure all costs and benefits of the intervention, including those that have a value even if they are not on the market. Shadow prices[6] also need to be incorporated if they are available.
4. Assess the flow of costs and benefits over the relevant time period, both for the scenario where the natural infrastructure intervention takes places as well as all alternative scenarios or the status quo.
5. Convert all costs and benefits into a common currency.
6. Apply an appropriate discount rate. IDB requires the flows of benefits and costs to be discounted at a rate of 12% per year. However, other discount rates can be evaluated through sensitivity analysis.
7. Calculate the net present value of the actions under consideration. The most relevant measurement is the present value of net benefits. Other measurements include the internal rate of return (IRR), and the benefit-cost ratio (BCR).
8. Perform sensitivity analysis.
9. Adopt the recommended course of action.

The principle behind the CBA is relatively straightforward. Suppose there is an intervention that an agency wants to implement in a particular region, for instance, a project to protect the shoreline from erosion or to reduce the risk of flooding in a coastal area. Two types of costs can be identified: First, the initial investment, a fixed cost equal to C_i^w. Second, maintenance costs over a time span

[5] Other references on CBA are Hanley & Spash (1993), Pearce et al. (2006) and Pearce (2006).

[6] Shadow prices are adjusted measures of market prices that take into account market failures such as monopolies or externalities. Some researchers have suggested that they are specifically important in developing countries since markets in these countries are often more distorted than in developed countries. In general, shadow prices are used to reflect the social value of a good in the cases were the observed value does not exist or deviates significantly from its social counterpart. For example, market prices of goods that pollute the environment may be adjusted upwards by the analyst in a CBA to include the external costs and consequences of such externality (Boardman et al., 2006).

of *T* years. This flow of costs needs to be brought to present value by discounting every year's costs and adding them up:

$$C_o^w = \sum_{t=0}^{T} \delta^t C_{o,t}^w$$

Here, $C_{o,t}^w$ is the operational cost at every period t; $\delta = \frac{1}{(1+\rho)}$ is the discount factor, where ρ corresponds to the adequate discount rate to be applied. C_o^w is the added present value of the flow of costs needed for the project to operate. Thus, $C^w = C_i^w + C_o^w$ reflects the present value of all the direct costs associated with the intervention. In the case of natural infrastructure, the investment costs C_i^w would correspond to the expenditures related to initial investments that are needed for the project to start (e.g., buying land, planting trees, moving sand, among others). Operational costs, C_o^w, are those needed for the continued maintenance of the investments over time during the lifetime of the project (e.g., replanting lost trees, preventing losses of material or biomass, monitoring, enforcement, maintenance).

There are other costs that can emerge from any project such as those related to the damages or losses generated by its execution. For instance, the project might affect ecosystems, reducing the provision of specific services. We refer to these costs as C^{ext}, and they are externalities or indirect costs generated from the project itself. The presence of externalities increases the costs of the project as shown in the following expression:

$$C = C^w + C^{ext}.$$

It is important to note that, if the project is not carried out, there is still a flow of costs (C_t^{wo}) that must be accounted for, brought to present value and then added up in order to estimate the costs of not carrying out the intervention, C^{wo}. These costs of inaction must be subtracted from the cost of execution, as they will be avoided once the investment is in place. In the case of natural infrastructure, these costs can relate to increased erosion or higher risks of flooding as the conditions of the ecosystems will eventually deteriorate if no action is taken. Costs can also take into account the increasing effects of climate change. These avoided costs can be considered as benefits resulting from the implementation of the project. Therefore, the total costs of the project would be:

$$C = C^w + C^{ext} - C^{wo}.$$

We present a summary of these costs in Table 3.

Once the direct costs of the project are determined, the next step is to identify the benefits it generates. These benefits can be divided in two categories: first, the benefits derived directly from achieving the objectives of the project, in this case coastal protection, and second, the benefits derived from the additional provision of other ecosystem services (provisioning, cultural and regulating services). As in the case of the costs, benefits need to be estimated for the full span of

the project and brought to present value. In Table 3, direct benefits from coastal protection are denoted by the letter *B*, and indirect benefits or co-benefits are denoted by the letter *A*.

Table 3. Costs and benefits generated by a project

Net total values	Total values with and without project	Initial investment	Operation
Net costs in present value $C = C^w + C^{ext} - C^{wo}$	With project, direct costs	$C^w = C_i^w + C_o^w$	C_i^w
	With project, indirect costs	C^{ext}	$C^{ext} = \sum_{t=0}^{T} \delta^t C_t^{ext}$
	Without project	C^{wo}	$C^{wo} = \sum_{t=0}^{T} \delta^t C_t^{wo}$
Net benefits from coastal protection in present value $B = B^w - B^{wo}$	With project	B^w	$B^w = \sum_{t=0}^{T} \delta^t B_{o,t}^w$
	Without project	B^{wo}	$B^{wo} = \sum_{t=0}^{T} \delta^t B_t^{wo}$
Net benefits from other ecosystem services (co-benefits) in present value $A = A^w - A^{wo}$	With project	A^w	$A^w = \sum_{t=0}^{T} \delta^t A_{o,t}^w$
	Without project	A^{wo}	$A^{wo} = \sum_{t=0}^{T} \delta^t A_t^{wo}$

As with project costs, it is important to recognize that there is a flow of direct benefits and co-benefits that would be obtained even if the project were not to be carried out. These benefits should be accounted for in order to identify the effect of the project and avoid overestimation of benefits of the projects, as there is some level of coastal protection that would be achieved without a project (B^{wo}). In the status quo, there is also a flow of co-benefits (carbon sequestration, fishery nursery, recreation, tourism, fishing, etc.), which are represented by A^{wo} (see Table 3).

Therefore, the net benefits from coastal protection at present value attributable to the project would be:

$$B = B^w - B^{wo}$$

Similarly, the net benefits derived from other ecosystem services (co-benefits), at present value are:

$$A = A^w - A^{wo}$$

The net aggregated benefits of the project would be $A + B$.

To summarize, the CBA of nature-based infrastructure or restoration interventions aims to provide information about the cost-effectiveness of protecting coastal communities against natural hazards

and the effects of climate change. This task requires the estimation of the net cost of design, implementation and maintenance (C), and the benefits that the ecosystems involved in the interventions generate for the intervened communities ($A + B$). The cost component of the CBA is generally estimated based on engineering and/or restoration expenditures C^{w}, the opportunity costs $C^{wo}$and the cost of negative environmental impacts C^{ext} (Corral and Schling, 2017). Regarding the latter, it is also important to keep in mind that a project can generate costs in terms of it affecting other agents as well as the provision of other ecosystem services. If so, these costs should be included as they are externalities that have to be considered in the analysis.

When accounting properly for all costs and benefits, a natural infrastructure project will be economically viable if $A + B - C > 0$. This aggregation ($A + B - C$) is known as the net present value (NPV) of the project, and, in the CBA, it is the central measurement of economic viability.

Other measures exist to identify the viability and profitability of an investment. Among these, there is the benefit/cost ratio (BCR), which in this case, would be:

$$BCR = \frac{A + B}{C}$$

If the BCR equals to or is larger than one, the project is said to be economically viable. Another commonly used measurement is the internal rate of return (IRR), which refers to the discount rate that would make the net present value of the project equal zero. If the IRR is larger than the discount rate used in the analysis, the project is viable and profitable.

Although estimating $A + B - C$ seems relatively straightforward, in practice, it could be very difficult to estimate some of these elements. In some cases, it might be impossible or too costly to estimate project benefits (*A+B*), meaning that all the benefits of a project will be unknown. In such case, one alternative is to carry out a cost-effectiveness analysis to compare the costs of natural infrastructure with those of other options. This approach assumes that the benefits are the same regardless of the type of intervention, and that the only decision to be made is based on identifying the most cost-effective alternative. Needless to say, this is a very strong assumption, particularly when comparing natural infrastructure projects with gray infrastructure interventions, as the former generates other additional benefits that gray infrastructure cannot provide.

A full CBA is the best approach to compare the economic viability of projects. When carrying out a CBA, however, other challenges may emerge. Sometimes, the costs and benefits of not implementing the project (C^{wo}, A^{wo}, B^{wo}) are ignored. They reflect the opportunity cost of not executing the project, among them the increased risk of flooding or erosion. By ignoring these costs, the project will be undervalued. In turn, overlooking the benefits that would be provided regardless of the project will also bias the results, overvaluing the project by attributing all of the benefits to the intervention.

With respect to the benefits (A, B), ecosystems can provide a large amount of services and not all of them can be included in the analysis. By selecting the most important services, the most relevant

values can be captured, and at the same time narrowing the analysis to a number of services that make the exercise doable. This can be done based on secondary information.

When some services –as is the case of provisioning services such as fish or timber- are traded in markets, benefits from ecosystems (*B*) can be partially included in the CBA. However, typically, coastal protection and other ecosystem services are not traded in the market and, therefore, do not have observable prices. Under such circumstances, the estimation of the direct benefits and co-benefits provided by natural infrastructure constitutes a challenge for the accurate and complete application of the tools for economic assessments. By ignoring these nonmarket values, the benefits from a natural infrastructure project will be undervalued and other alternatives (such as gray infrastructure) may seem more profitable. Therefore, it is necessary to use appropriate economic valuation methods for the ecosystem services provided for each specific nature-based infrastructure project for coastal protection.

Risk assessment applied to coastal protection

One of the main challenges of carrying out CBAs of infrastructure projects for coastal protection is that they should consider the effects of climate change on both coastal zones and coastal infrastructure. Sea level rise and the increase in the frequency, duration and intensity of storms, hurricanes and other extreme events are likely to increase human and economic vulnerability in coastal areas. According to the Caribbean Catastrophe Risk Insurance Facility (CCRIF, 2010), annual expected losses from wind, storm surge, and inland flooding in the Caribbean are equivalent to 6% of GDP in some countries, and climate change may increase expected losses by 1-3% of GDP by 2030. Climate change increases the risk of extreme conditions, which implies a need for infrastructure that is better adapted to the changing environment. Analytically, this requires the consideration of uncertainty in CBAs. To do so, vulnerability and risk assessment models that simulate different conditions and states of nature may be carried out.

Some vulnerability studies and risk assessments applied to coastal protection include Arkema et al. (2013), Arkema et al. (2014) and CCRIF (2010). Arkema et al. (2013; 2014) model changes in vulnerability of coastal zones under different zoning and habitats scenarios, and CCRIF (2010) analyze the potential economic impact of climate change in eight Caribbean countries. Likewise, IDB has disaster risk profiles for Jamaica (IDB, 2014) and Trinidad and Tobago (CIMNE et al., 2013), with risk models based on probabilistic formulations that incorporate uncertainty. Probabilistic risk models are constructed as a sequence of modules that quantify the potential losses from a specific event. Once the expected physical damage is estimated, the probable maximum loss (PML) can be derived for different time periods, allowing to estimate average annual loss (AAL) and technical risk premiums. These indicators are generated for use and application in future financing instruments or risk transference. In sum, vulnerability and risk assessments offer additional inputs to incorporate uncertainty associated with climate into CBAs of nature infrastructure projects in coastal zones.

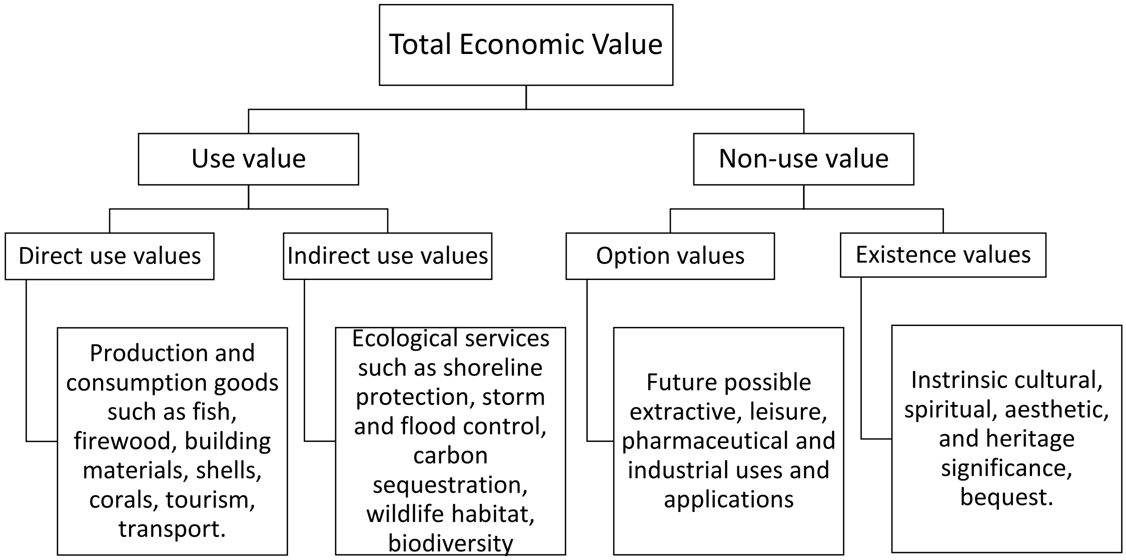
ECONOMIC VALUATION OF ECOSYSTEM SERVICES

Despite the importance of coastal and marine ecosystem services in Caribbean countries, natural systems that provide them are rapidly degrading, in part because knowledge about their economic and social benefits is limited. Economic valuation can contribute to better-informed decision making about coastal use and development, as it allows not only for the identification of the ecosystem services provided, but also for the recognition of their benefits.

A better understanding and recognition of the vital role played by ecosystems in coastal areas is a fundamental part of ICZM. Valuation can be a tool to make more informed decisions about the development, planning, conservation, and provision of public goods and services (Waite et al., 2015). We define economic valuation as the identification and physical and monetary quantification of the costs and benefits derived from changes in an ecosystem's provision of goods and services (Maldonado & Moreno-Sánchez, 2012). Economic valuation attempts to measure the value of ecosystem services in monetary units to provide a common metric that reflects the benefits of the variety of services provided by the ecosystems. Economic valuation is based on the concept of Total Economic Value (TEV).

The TEV associated with an ecosystem incorporates different dimensions of value. According to Pearce and Moran (1994), TEV can be divided into two categories (see Figure 2): use value and non-use value. Use value is, in turn, divided into direct use value and indirect use value, while non-use value is classified as either option value or existence value. Use values, as indicated by their name, are those that society assigns to ecosystems in exchange for the benefit of using or exploiting them in the present. The most evident is direct use value, which refers to goods (fish, timber, construction materials) or services (tourism, recreation, transport) that are obtained directly from exploiting ecosystems. Direct benefits can be derived via extraction (e.g., fish harvests, collection of raw materials) and are associated with provisioning services, while benefits derived from non-extractive direct interactions such as recreation, research and aesthetic enjoyment are associated with cultural services. Additionally, there are a number of services, mainly regulation services, such as coastal protection, flood control and carbon sequestration, among others, that ecosystems provide to society, although not directly. These benefits are referred to as indirect use values. Furthermore, society confers value to ecosystems not only for their current use but also for their potential future uses, such as the discovery of yet unknown pharmaceutical uses of biodiversity components, which are known as option values. At the same time, society can ascribe value to an ecosystem for the simple fact that it exists, regardless of whether it is used directly or indirectly, be it for intrinsic, cultural, spiritual or moral reasons, or by bequest. This latter group is known as existence value.

Figure 2. Different values associated with an ecosystem and its services



Source: Adapted from Emerton (1999).

Table 4 shows the relationships between the types of values according to the TEV categorization and the types of ecosystem services according to the Millennium Ecosystem Assessment classification (MEA, 2005)[7].

Table 4. Relationships between different values of an ecosystem and the services it provides

Category	Direct use value	Indirect use value	Option value	Existence value
Provisioning	Strong		Medium	Low
Regulating		Strong	Low	
Cultural	Strong		Medium	Medium
Supporting	Valued through other cosystem service categories			

Source: The authors, based on TEEB (2010).

STEPS FOR THE ECONOMIC VALUATION PROCESS

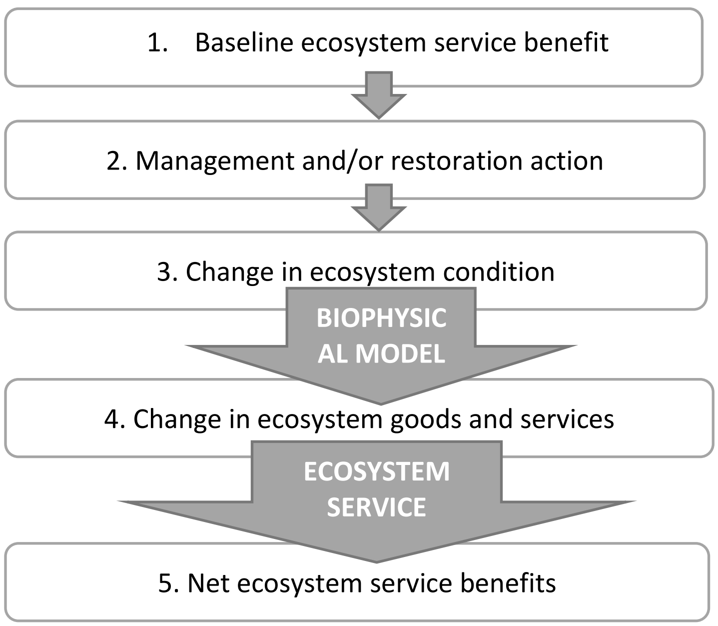
To make economic valuation operational, several considerations need to be taken into account. One central point is that valuation should be performed to assess the effects of a policy, program or an anthropogenic intervention in a given ecosystem. That implies that it is necessary to identify a status quo scenario prior to an intervention taking place and a final condition after implementation. Once these two scenarios are identified, the valuation exercise can be carried out.

[7] The Intergovernmental Panel on Biodiversity and Ecosystem Services (IPBES) proposes (i) the concept of nature contributions to people to emphasize the role that culture and local knowledge plays in defining the relation between people and nature and (ii) a new classification of ecosystem de services based on this new approach (Díaz et al., 2018).

Schuster and Doerr (2015) present the steps to measure the change in ecosystem service benefits within the framework of coastal restoration projects (Figure 3):

1. Before the intervention, some benefits from ecosystem service provision might exist that should be recognized.
2. An ecosystem management and/or restoration action (intervention) is implemented in one or more ecosystems.
3. The condition of the ecosystem (structure and function) changes as a result of the intervention.
4. The provision of ecosystem services changes as a result of changes in the ecosystem's condition.
5. Net benefits from changes in ecosystem services arise.

Figure 3. Elements for the assessment of ecosystem service benefits from a management action.



Source: Schuster and Doerr (2015, p.43)

This generic approach shows that economic valuation is not uniquely related to analysis by economists; it requires the participation of biologists, engineers and other scientists that can identify changes in ecosystem conditions and ecosystem services, and assess the relevance of these changes to society. In particular, the relationship between the changes in ecosystem conditions and the services provided can be identified using an ecological production function (EPF), whose inputs are the changes in ecosystem conditions and the output is the quantification of the net ecosystem service benefits (Schuster and Doerr, 2015). The EPF is defined as a mathematical expression that estimates the effects of changes in the structure, function and dynamics of an ecosystem on outputs with direct relevance to decision makers (U.S. Army Corps of Engineers, 2015 cited in NSTC, 2015). Schuster and Doerr (2015, p. 60) define the EPF as "the quantitative relationship between the underlying ecological function and the resulting ecosystem service." The biophysical model then establishes the link between elements 3 and 4, as shown in Figure 3. The output from this model

(the change in the ecosystem goods and services) would be the input for the ecosystem service valuation, which connects elements 4 and 5.

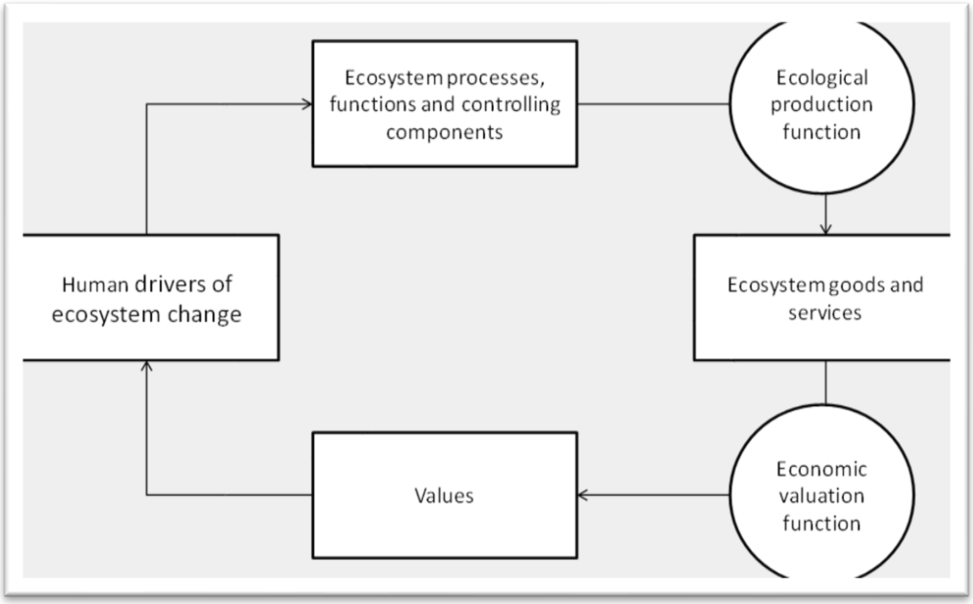
Following the approaches suggested by Schuster and Doerr (2015), Barbier (1994; 2007; 2011), Freeman (2003), the National Research Council (NRC, 2005), Polasky and Segerson (2009) and NSTC (2015), an economic valuation can be performed in the three steps:

1. First, it is important to identify and describe qualitatively those ecosystem services that would be affected as a result of the intervention, taking into consideration the physical natural environment..
2. Second, the marginal change in the provision of ecosystem goods and services resulting from the intervention needs to be quantified using non-monetary measures. This step also involves identifying who the main beneficiaries from the ecosystem service change are.
3. Finally, once the changes in the physical flows of the goods and services provided by ecosystems have been identified and quantified, a monetary value to these changes (economic valuation), which captures the importance that society gives to the provision of such services, is assigned.

This process is summarized by Barbier et al. (2011) in Figure 4. In this diagram, human drivers are the human interventions intended to affect the functioning of ecosystems. Once ecosystems are altered by an intervention, the provision of ecosystem services must be identified using EPFs. Ecosystem services are then valued with an economic valuation function.

With this approach, a good economic valuation requires an adequate identification of the changes in ecosystem services. There are several EPF-based biophysical models designed to quantify ecosystem services related to marine and coastal projects. Examples include the InVEST models – coastal blue carbon, coastal vulnerability, fisheries, habitat risk assessment, marine fish aquaculture, offshore wind energy, visitation (recreation and tourism) and wave energy-; Artificial Intelligence for Ecosystem Services (ARIES); Coastal Resilience Tool; the Federal Emergency Management Agency's (FEMA) tool and Wave Height Analysis for Flood Insurance Studies (WHAFIS); Coastal Hazard Analysis Modeling Program (CHAMP); Toolkit for Ecosystem Services Site-Based Assessment (TESSA); XBeach; and the Climate Adaptation (Climada) model, among others.

Figure 4. Steps in the valuation of ecosystem services



Source: Barbier et al. (2011)

We present a summary of some of the platforms used for identifying changes in coastal and marine ecosystem services below.

THE INTEGRATED VALUATION OF ECOSYSTEM SERVICES AND TRADEOFFS: INVEST

InVEST[8] is a software created by the Natural Capital Project as a "tool for exploring how changes in the ecosystems are likely to lead to changes in benefits that flow to people" (Sharp et al., 2018, p. 10). InVEST is based on a production function approach that quantifies and values ecosystem services in relation to changes in ecosystem structure (Guerry et al., 2012). In order to link the production function to the benefits provided to people, InVEST identifies the ecological functions provided by ecosystems (supply), then links these functions to the demand, considering the beneficiaries of the ecosystem services (service), and finally includes social preferences to calculate the economic and social metrics (value) (Sharp et al., 2018). In this way, the models are designed to identify, quantify, map and value the benefits provided by ecosystems.

From the set of models provided by InVEST, there are some that are suitable for marine and coastal ecosystems, among them the habitat risk assessment, coastal blue carbon, visitation, wave energy production, offshore wind energy production, marine finfish aquaculture, and fisheries models, as well as the module for coastal vulnerability. Table 5 shows some examples of the InVEST models that have been applied to marine and coastal ecosystems in the Wider Caribbean.

[8] See Sharp et al. (2018) for InVEST user guide.

Table 5. Some applications of InVEST models to marine ecosystems in the Wider Caribbean

Model	Applications in the Wider Caribbean
Habitat risk assessment	Arkema et al. (2014; 2015)
Blue carbon storage and sequestration	Chmura et al. (2003), Arkema et al. (2017)
Fisheries	Arkema et al. (in review), Clarke et al. (2013), Guannel et al. (2014), Arkema et al. (2015; 2017)
Coastal vulnerability	Arkema et al. (2013), Ruckelshaus et al. (2016), Verutes et al. (2017)
Recreation	Ruckelshaus et al. (2016), Arkema et al. (2015; 2017), Verutes et al. (2017)
Coastal protection	Ruckelshaus et al. (2016), Arkema et al. (2015; 2017), Verutes et al. (2017)

ARTIFICIAL INTELLIGENCE FOR ECOSYSTEM SERVICES (ARIES)

ARIES is an open-source technology capable of selecting and running models to quantify and map all aspects of ecosystem service provision, including biophysical generation, flow and extraction by sinks and beneficiaries.[9] It models the provision of multiple ecosystem services (World Bank, 2016) and the complex relationships between ecosystems and human activities and values. In particular, with ARIES it is possible to model the provision of carbon sequestration and storage, aesthetic services, flood regulation, subsistence fisheries, coastal flood regulation, sediment regulation, crop pollination, water supply and recreation (Martinez-López et al., 2019; Bagstad et al., 2011). The methodology combines spatially explicit models of ecosystem service provision and use with dynamic flow models to describe the distribution of benefits across the landscape. ARIES includes two coastal and marine models: subsistence fisheries and coastal flood regulation. It is relevant to highlight that ARIES does not have an economic section for valuation, concentrating on the biophysical modeling of ecosystem services and leaving the calculation of economic value and its implications to the end user.

COASTAL RESILIENCE TOOL

The coastal resilience tool was developed by The Nature Conservancy (TNC) and it consists of various web apps that use a specific approach (assess risk vulnerability, identify solutions, take action and measure effectiveness), a web mapping tool and a network of practitioners around the globe (TNC, 2016). At present, it comprises 17 web apps that cover topics such as community planning, coastal defense and economics of coastal adaptation, among others. The main objectives of the tool are to i) visualize the potential impact of sea level rise, storm surge, hurricanes and inland flooding; ii) combine coastal habitat, exposure and socioeconomic factors, to analyze where habitat management is most effective; iii) analyze the effects of natural and engineered (gray) solutions; and, iv) compare risk and vulnerability indicators around the world (TNC, 2016).

[9] https://www.ipbes.net/policy-support/tools-instruments/artificial-intelligence-ecosystem-services-aries

COASTAL HAZARD ANALYSIS MODELING PROGRAM (CHAMP)

CHAMP is a software created by the Federal Emergency Management Agency (FEMA) to conduct coastal flood hazard assessments. The program allows users to enter data, perform coastal engineering analyses, view and tabulate results, and collect summary information for representative transect locations along a coastline.[10] CHAMP evaluates storm-induced erosion, wave height and wave run-up associated with coastal flood hazard assessments (FEMA, 2007; 2013).

TOOLKIT FOR ECOSYSTEM SERVICES SITE-BASED ASSESSMENT (TESSA)

TESSA is a toolkit that allows for a high level of stakeholder engagement to generate information at site scale about ecosystem services values, using locally gathered information at particular protected areas, important sites for biodiversity or project locations (BirdLife International, 2018; Peh et al., 2013). It is based on a comparison of site evaluations in two alternative states: before and after restoration or conversion.

TESSA is designed to be used online and in the field. It is relatively low-cost compared to other tools and is accessible to non-experts and conservation practitioners. It does not require high-level technical skills and collects locally-relevant data that can be quantitative and/or qualitative (BirdLife International, 2018). Importantly, TESSA offers guidance for carrying out the monetary and non-monetary valuation of some of the ecosystem services it assesses.

Currently, the toolkit evaluates the following types of services: global climate-regulating, water services (supply, quality, and flood reduction), harvested wild goods, cultivated goods, nature-based recreation, cultural services and pollination services and coastal protection (Peh et al., 2017).

The module for coastal protection in the TESSA toolkit describes the methods and approaches for the assessment of the hazard reduction potential of a wetland site (mudflat, salt marsh or mangrove). The aim of the module is to help assess the value of coastal protection services provided by ecosystems against wind-waves, water levels and erosion related hazards (Peh et al., 2017).

XBEACH

XBeach is a two-dimensional open-source numerical model developed by UNESCO-IHE, Delft University of Technology, Deltares and the University of Miami, whose main function is to simulate hydrodynamic and morphodynamic processes that occur in coastal areas during storms (Nederhoff, 2015; Roelvink et al., 2015).

The hydrodynamic process simulates the refraction, shoaling and breaking of short-wave transformation; the generation, propagation and dissipation of long wave transformation; as well as wave-induced unsteady currents, overwash and inundation (Roelvink et al., 2015). The

[10] https://data.femadata.com/NationalDisasters/Hurricane%20Sandy/RiskMAP/Public/Public_Documents/Workmaps/Fact_Sheet_NJ_CHAMP.pdf

morphodynamic process models bed load, suspended sediment transportation, dune face avalanching, bed update, and breaching (Roelvink et al., 2015). XBeach carries out a single numerical step by calculating the gradients in radiation stress in the short wave module, it then uses those changes in the gradient stress to estimate velocities and surface elevation through the flow module, that will, later on, lead to sediment transportation and updating of the bed level in the morphodynamics modules (Nederhoff, 2015). It is important to highlight that the software allows the user to enter vegetation (e.g., mangroves) and hard infrastructure data since they have a relevant damping effect on waves (Roelvink et al., 2015; van Rooijen et al., 2015).

CLIMATE ADAPTATION (CLIMADA)

Climada is an open-source probabilistic risk and economic model that implements the economic climate adaptation (ECA) methodology and helps decision-makers understand the impact that climate has on the economy (Bresch et al., 2018). The model is based on four elements: assets, damage functions, hazards and adaptation measures (which includes natural and gray infrastructure) (Bresch, 2015).

According to Bresch et al. (2018), through probabilistic modeling, Climada estimates expected economic damage, using the resultant variable to measure today's risk, as well as measuring increases in economic growth and risk as a consequence of climate change. It then creates a portfolio of adaptation measures and assesses the cost-benefit and potential aversion damage for each one. Finally, an adaptation cost curve is built to show the user the prevention, intervention and insurance measures of the portfolio previously created.

METHODS TO ESTIMATE ECONOMIC VALUES OF ECOSYSTEM SERVICES

Once a biophysical model has generated information about the change in ecosystem services, the next step is to assign a value to these ecosystem services. Economists have relied on several techniques to assess these values. The use of a specific technique depends on, among other factors, the particular ecosystem service to be valued (provisioning, regulation, cultural), the type of value (use vs. non-use value) and the agents that –in the end– will benefit from the provision of a particular ecosystem service.

In general terms, valuation techniques can be classified as revealed-preference methods and stated-preference methods. Revealed-preference methods use information already provided or 'revealed' by markets. Because these methods use information from markets directly, they are called direct methods. For instance, a direct method observes how much timber is obtained from a forest and values it based on the market price of timber. On the other hand, indirect methods involve the valuation of goods and services that are not traded in the market. These approaches make use of information from substitute or complementary goods and services that are marketed is for value inference. For instance, travel-cost methods use information on holiday expenditures as a means to approximate the consumer surplus of demand for trips to a natural park as a way to value it. Because revealed-preference methods use information from markets, they are generally effective to capture

use values, particularly direct values. However, these methods are not suitable to assess non-use values.

Stated-preference methods, conversely, involve the creation of hypothetical markets where individuals can state the value of a given change in conditions associated with the provision of ecosystem services. In other words, these methods capture values without needing markets to reveal a price. Because of this, they typically employ surveys where a hypothetical scenario is presented to respondents and they have to state their willingness to pay for a given change in the conditions of that scenario. These methods have gained special attention over the last few decades as they are the only way to capture non-use values (and even some use values).

Table 6 presents some of the most common methods used for valuation and their classification according to the type of information available.

Table 6. Some methods of economic valuation and their classification.

Methods	Revealed preferences	Stated preferences
Direct	Market prices Production function Auctions	Contingent valuation Choice experiments Conjoint analysis
Indirect	Prices of alternatives or substitutes Collection and preparation value Averting behavior models Avoided damage/defensive expenditures Replacement costs Hedonic prices Travel cost Expected damage function	

All of these methods for economic valuation are relevant to natural infrastructure, conservation and restoration of marine and coastal ecosystem interventions. We discuss them briefly below.

REVEALED-PREFERENCE METHODS

The first group of methods are those based on revealed preferences. They are used primarily to capture provisioning services, as well as cultural services such as recreation.

Valuation based on market prices of ecosystem goods and services

Market prices are the most straightforward way to assess value for ecosystem services. Market prices typically reflect what people are willing to pay for a given good or service, and therefore the value that they place on them (Emerton, 1999). The value of goods and services provided by ecosystems such as fish, timber and firewood, recreation activities such as diving, and tourist activities such as visits to historic buildings and monuments, can be approximated with market price data. This method can also be used to quantify the value of marine products that are extracted by local communities and that are used within the household for self-consumption, such as fish and

timber). As long as these goods and services have a market, their price represents expenditures saved or potential to earn income (Emerton, 1999).

Although market prices are useful to estimate economic values, they pose some limitations that should be considered: first, a number of goods and services, such as those that are only used for subsistence purposes and never sold, are not traded in the market; second, market prices are distorted because of taxes, subsidies, monopolies, externalities, or other interventions and do not reflect the true value; and third, estimated values only capture revenues and do not incorporate the costs of service provision, and therefore cannot be considered as economic values (Schuhmann, 2012). Ideally, with complete information from markets, economic value estimates could be obtained with the consumer and producer surpluses. However, their estimation is complicated due to data availability and uncertainty regarding the shape of the curves of supply and demand.

This approach is useful to estimate the direct-use values of provisioning and cultural services. In some cases, it can also be used to indirectly evaluate regulating services, such as the value of carbon sequestration or the contribution of mangroves to fisheries via providing nursery habitats.

Prices of alternatives or substitutes

Although some products from marine and coastal ecosystems have no direct markets, they do have close substitutes that are traded. For instance, local coastal communities would need to buy protein if fish and seafood were not available. Similarly, building materials such as bricks or wood need to be bought on the market if coral or mangroves did not provide these materials. The prices of these substitute goods can be used as a proxy for the value of the goods obtained from ecosystems (Emerton, 1999).

Collection and preparation value

In some cases, products have no market prices and no market substitutes. One way of valuing these products is to consider the value of time and other scarce inputs allocated to obtaining them, as these inputs do have a market price (e.g., wages) (Emerton, 1999). This approach is useful to estimate direct use values of provisioning services.

Averting behavior models

These models assume that people will change their behavior and invest money to avoid an undesirable outcome. The model analyzes the rate of substitution between changes in behavior and expenditures and changes in environmental quality in order to infer the value of certain non-market environmental attributes (NRC, 2005). For instance, if flood risk is increased, a household may move upland to avoid the damage caused by flooding. These models are known as the avoided damage approach or defensive expenditures, because they use estimates of the expenditures that would be incurred to prevent, diminish or avoid harmful effects associated with the loss of natural resources (Schuhmann, 2012).

In these models, the behavior of the individual in the market as well as the expenditures incurred in mitigating the effects of the change in environmental quality are analyzed to obtain value estimates of the environmental service (Sundberg, 2004). This method provides information about the demand for the environmental service and estimates of its use value as a result. Given that the service being valued may have regulating functions, the approach can be used for obtaining indirect values estimates, such as flood control.

The limitation of this approach is that it lead to problems of underestimation or overestimation of economic value (NRC, 2005). The value will be underestimated if analysts fail to include the inconvenience of undertaking the averting behavior (construction, relocation, etc.) Similarly, value will be underestimated if households cannot fully remove the risk of damage from flood or erosion with the averting behavior. Averting behavior will overestimate the economic value when joint production is present, e.g. when the averting behavior produces other ecosystem services such as improved recreation possibilities as a result of control barriers being built.

Replacement costs

This technique is based on the notion that if the ecosystem service were not available it would be needed to be replaced by alternative means (Emerton, 1999). This method is closely related to the defensive expenditure method, as it also relies on the perfect substitutability assumption. In this case, however, the replacement cost method uses the cost of the perfect substitute to estimate the value of the environmental good. Investment and maintenance costs should be included in the replacement cost (Sundberg, 2004).

Coastal protection ecosystem services can be valued by assessing how much it would cost to set in place measures to prevent or mitigate the damages arising from the loss of ecosystems. For example, flood control barriers, such as breakwaters or sea walls, might be needed to prevent the negative impacts associated with the loss of flood control services provided by wetlands, and these expenditures represent the value of coastal services. Likewise, the shoreline protection function of coral reefs or mangroves could be replaced by the construction of groins and barriers. These replacement costs reflect expenditures saved by the presence of coastal ecosystems and thus, seem to be an option for valuing natural infrastructure projects. However, this approach is not based on market behavior, and uses costs as a measure of benefits. This implies that the ratio of costs to benefits of an ecological service would be equal to one (Barbier, 2007). In addition, this method is not based on preferences, and therefore, there is no certainty that these replacement or treatment activities would be the best option for the agents involved. In that sense, it cannot be used as a measure of economic value (NRC, 2005; Barbier, 2007). Nonetheless, several authors, among them Shabman and Batie (1978), WRI (2009) and Sundberg (2004), argue that this method can be a valid measure of economic value only if three conditions are met (i.e.,):

1. The natural service can be replaced with a man-made alternative that is equivalent in quality and magnitude to the ecosystem service.

2. The costs of that substitute are known or estimable and the substitute represents the least costly means of providing the service and replacing the ecosystem service.
3. Society is willing and able to incur the costs associated with the replacement.

Travel cost

Travel cost is a method suitable for valuing recreation services. Travel cost studies attempt to infer non-market values of ecological services by using the travel and time costs that an individual incurs to visit a recreation site (Bockstael, 1995). Given that marine ecosystems hold a high value as a recreational or leisure destination (e.g., sailing, swimming, sunbathing, diving, snorkeling or birdwatching), this approach is useful to estimate the value of these cultural services. The travel cost method collects the costs incurred when visiting a marine location, such as gasoline, bus fares, accommodation, tour guides, food expenses, as well as the cost of the time spent during such visits and the frequency of trips. These costs allow estimating a demand function for a particular site in order to calculate the consumer surplus, which reflects the value allocated to this site. This method is difficult to apply when there are other destinations available and/or the individual makes multifunctional trips. A way of overcoming this challenge is by using the approach based on the random utility model (RUM) (Haab and McConnell, 2003).

Hedonic prices

Hedonic methods analyze how the different characteristics of a marketed good, including environmental quality, might affect the price people pay for it. This type of analysis provides estimates of the implicit prices paid for each feature (NRC, 2005). The most common application of hedonic methods in environmental economics is to real estate sales (Palmquist, 1991; 2003). The assumption is that property values reflect a stream of benefits, some of which are attributable to the environmental good. The task is to isolate the value attributable to the environmental good or service. Hedonic pricing can be used to establish some of the values associated with coastal protection as properties in proximity to an area where natural infrastructure interventions aimed at reducing erosion or lowering flood risk are likely to hold a higher value because these services are considered as a benefit. In general, a hedonic analysis is a statistical procedure that identify the premium people pay for less erosion or lower risk of flooding, which is the revealed value for these ecological services. It can also be used for other services such as coastline landscape (aesthetic values) or better opportunities for recreation.

Production function

Production function approaches assume that an ecosystem service is an input for the production of a marketed good or service that yields utility. Therefore, changes in the availability of the ecosystem service can affect the costs and supply of the marketed good, the return of other factor inputs, or both (NRC, 2005). Dose-response and change-in-productivity models can be considered to be special cases of this approach in which the production responses to ecosystem services changes are simplified.

This method links the impact of a change in environmental conditions to the provision of particular goods or services by using a model describing the production relationship (Schuhmann, 2012). The application of this method requires an appropriate understanding of the relationship between the environmental resource and the resulting impacts on the production of the good or service of interest, which is closely related to the ecological production functions associated with the aforementioned biophysical model.

Expected damage function

The expected damage function approach is a special category of production function model, and it assumes that the value of an asset that yields a benefit in terms of reducing the probability and severity of economic damage is measured by the reduction in the expected damage (Barbier, 2007).

According to Barbier (2007), the essential step in the implementation of this approach is to estimate how changes in the asset affect the probability of occurrence of the damaging event. This approach is applied, under certain conditions, to value ecological services that also reduce the probability and severity of economic damages. This approach is appropriate when valuing regulating services such as those related to coastal protection.

STATED-PREFERENCE METHODS

As mentioned before, these methods do not depend on information from actual markets and have the advantage of capturing non-use values, such as option and existence values. Stated-preference approaches rely on the construction of scenarios that offer different policy actions, and are based on surveys where respondents are asked to state their preferences concerning these options. Economic value is derived from the hypothetical markets created in the survey (Carson, 2000). To create these hypothetical markets, the options presented to respondents are associated with a payment. In this way, a marginal rate of substitution between the environmental alternative and the payment can be used to estimate the value associated with the selected alternative. Two of the most common stated-preference methods are contingent valuation and choice experiments.

Contingent valuation

The foundation of a contingent valuation is to create a realistic, albeit hypothetical, market where individuals value a good or service. The most direct and widely accepted approach is the referendum format. With this approach, the respondent is offered a binary choice between two alternatives: the status quo (no change in conditions) and an alternative policy implementation that would imply a cost for the respondent, generally in the form of a tax or fee. The cost communicated to the respondent is randomly picked from a vector of possible values. The respondent must select one of the two options and the information collected from several participants exposed to different costs is used to estimate people’s willingness to pay for the policy or the hypothetical scenario (Carson, 2000). This method is useful to estimate use and non-use values of changes in scenarios due to natural infrastructure interventions that alter one or more ecosystem services.

Choice experiments

While contingent valuation is useful for deriving value estimates for ecosystem changes, choice experiments are more useful in terms of determining the value associated with particular characteristics or attributes of ecosystem modifications, such as hard infrastructure interventions. This approach is increasingly gaining favor in the literature as it avoids many of the inherent biases associated with contingent valuation methods (Schuhmann, 2012). In a choice experiment, individuals are given a hypothetical setting and asked to choose their preferred option among several alternatives in a choice set. In this setting, each alternative is described by a number of attributes or characteristics, and a monetary value is included as one of the attributes. When making choices, individuals have to make trade-offs between the levels of the attributes in the different alternatives presented in a choice set (Alpizar et al., 2001). Unlike other valuation methods, choice experiments allow multidimensional attribute changes to be valued simultaneously and can be used to generate estimates of the relative value of multiple attributes (Huybers, 2004). The key to the experiment design is the variation of the alternatives across scenarios. By observing the changes in stated choices due to the variation in any given alternative's characteristics, the effect of the attributes on the choices made can be derived (Schuhmann, 2012).

BENEFIT TRANSFER

Benefit transfer is the process of applying economic value estimates from a previously assessed case (the study site) and applying the monetary values estimated to assess the value of a quantified effect in a different study (the policy site) (Pearce and Özdemiroglu, 2002). Although primary data and studies provide the most accurate information on the role of a specific ecosystem, benefit transfer (also known as value transfer) techniques may work as acceptable alternatives when there is a lack of time, money, and technical expertise, or when logistical obstacles to primary valuation prove insurmountable. The two main approaches to this practice are benefit estimate transfer and benefit function transfer (Schuhmann, 2012). Benefit estimate transfer directly applies summary estimates of environmental benefits (usually average values) from the study site to the policy site, while benefit function transfer applies an empirical model of benefits to the policy site.

Benefit transfer can be applied using the results of studies of both revealed and stated preference models. However, they tend to be less accurate than other methods and it is recommended to use this approach only as the first step of a valuation exercise to compare values estimated from other studies and to have a reference point for more sophisticated studies.

ECONOMIC VALUATION APPLIED TO COASTAL PROTECTION

Although ecosystem service valuation is not a new concept, the body of literature related to coastal and marine ecosystem valuation is limited to support effective policy making (Barbier, 2013). In particular, studies on coastal protection services such as buffering homes and roads from flooding, reducing wave energy from storms, and erosion control/shoreline stabilization, are scant in contrast with other marine and coastal ecosystems services such as fishing or recreational services.

Barbier (2013) highlights two important challenges in valuing ecosystem services provided by coastal ecosystems: first, knowledge to appropriately link the changes in the ecosystem structure and the production of valuable ecosystem services is inadequate; and second, further work on probabilistic risk models that incorporate the uncertainty related with climate change is needed.

We now provide a review of the evidence for the economic value of coastal protection from marine and coastal ecosystems, summarizing120 coastal protection economic valuation studies from around the world. We find 101 studies that describe the contribution of marine and coastal ecosystems to coastal protection, and 19 that examine the economic value of reducing coastal erosion, restoring beaches and improving infrastructure in coastal areas. Our summary considers previous literature reviews on coastal protection services such as Barbier (2015), Barbier et al. (2011), Chong (2005), De Groot et al. (2012), Schumann (2011), Mahvar et al. (2018), Salem & Mercer (2012), and World Resources Institute (2011), but the number of studies analyzed here is significantly higher. Our review also covers technical reports, academic studies and scientific articles available on the Internet and takes into account the location of the studies, the economic valuation methods applied and the most valued ecosystems with respect to coastal protection services.

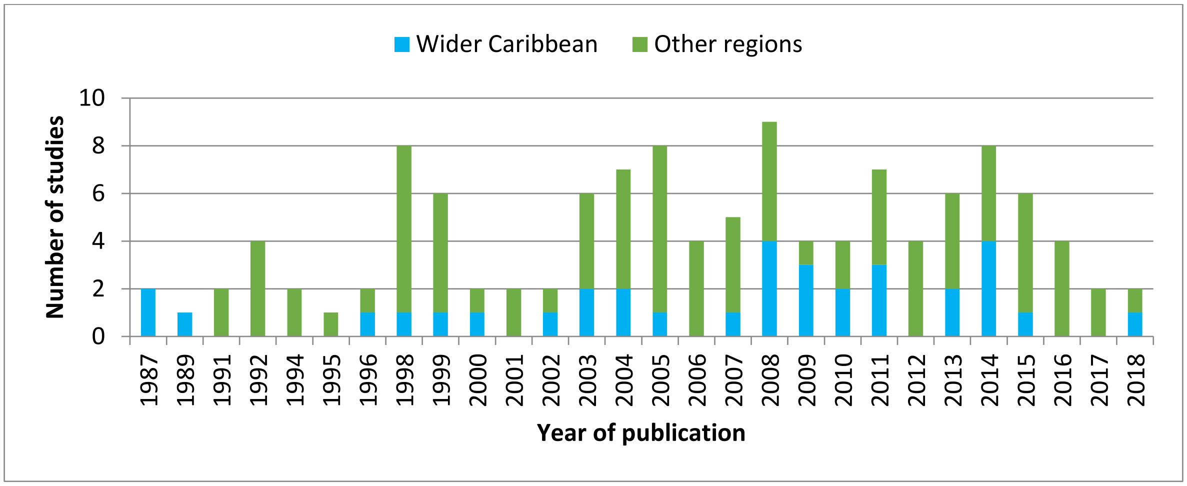
WHERE HAVE ECONOMIC VALUATIONS FOR COASTAL PROTECTION SERVICES BEEN APPLIED?

The existing literature on the value of coastal protective services is limited both by the number of studies and their geographic coverage (Sanchirico et al., 2015). Most valuation studies come from Asia (30%) and the Wider Caribbean (28%). It is important to highlight that about half of the studies for the Wider Caribbean examine the United States' Gulf of Mexico. Studies from other places in the United States account for 23%, whereas another 10%focuses on Europe. Global studies account for 4% of the total. In North America, economic valuation studies on the Mississippi and Louisiana regions stand out, while in Asia, most of the studies are carried out for Sri Lanka, the Philippines and Thailand.

In terms of the geographical coverage of economic valuation studies, 54% of studies focus on Small Island Developing States (SIDS). This is not surprising considering that SIDS are disproportionately exposed to the effects of sea level rise and hurricanes. Although there has been important progress in the development of economic valuations of coastal protection services, our review shows that, with the exception of the United States, over the past 30 years few studies have quantified the economic values associated with coastal protection in the Wider Caribbean (Figure 5).

Of the 39 territories comprising the Wider Caribbean, only 10 have been included in economic valuation studies of coastal protection services provided by the marine and coastal ecosystems, with the Unites States being the most relevant, with most studies conducted for Louisiana. In addition to the USA, Belize is the only continental country that has been included in an economic valuation study of coastal protection services, and Jamaica has been the target of the most studies applied in the region.

Figure 5. Economic valuation studies of coastal protection service provided by coastal and marine ecosystems in the Wider Caribbean and the rest of the world, 1987-2018



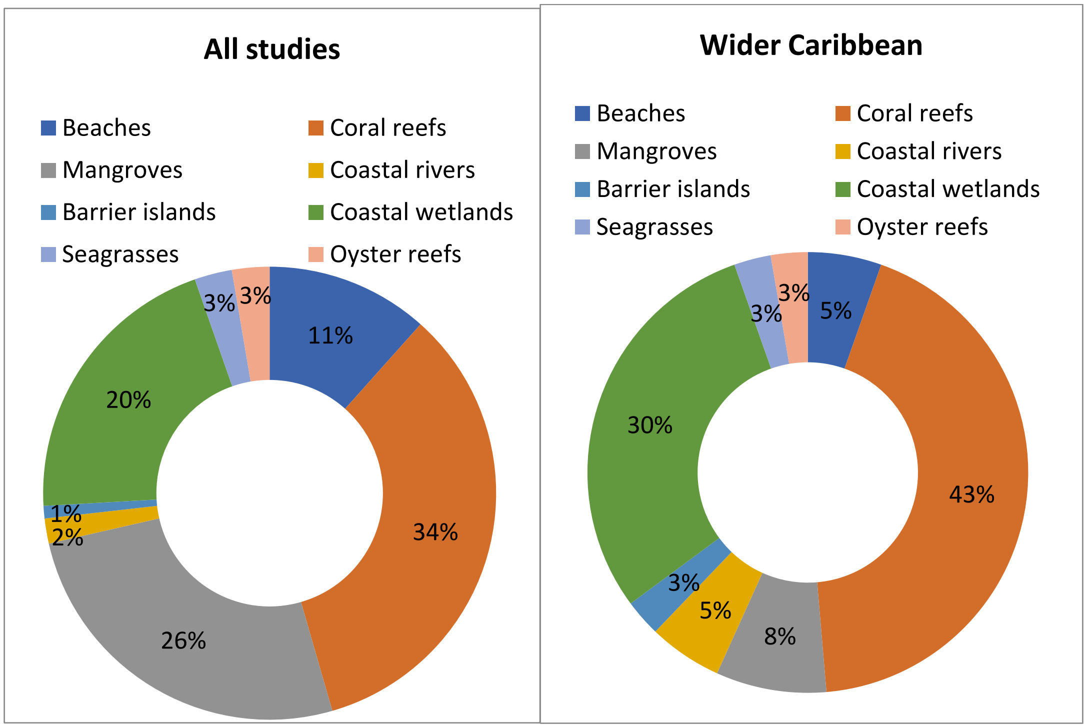
Source: The authors.

WHAT HAS BEEN VALUED?

According to Barbier et al. (2011) and Kroeger (2012), there is generally more and better data on the protective services provided by coastal wetlands, mangroves and coral reefs than on those provided by seagrasses, beaches and dunes. Studies that value the coastal protection service of coral reefs are the most common, both for the Wider Caribbean (Figure 6, right panel) and other regions (left panel). After coral reefs, 26% of the estimated economic values refer to the coastal protection service provided by mangroves. This figure increases to 47% if mangroves are included within the coastal wetlands category. In the Wider Caribbean, studies including mangroves are only 8% of the sample, although count for as much as 39% if wetlands are added to this category. This difference is due to the fact that most of studies analyzing mangroves have been carried out in Asia.

These studies estimate the coastal protection value related to erosion control (46%), extreme events (40%) or flooding protection (13%). The studies in the Wider Caribbean emphasize extreme events more (51%), while those focusing on erosion control and flooding comprise 37% and 11%, respectively.

Figure 6. Distribution of economic valuation studies on coastal protection service provided by coastal and marine ecosystems.



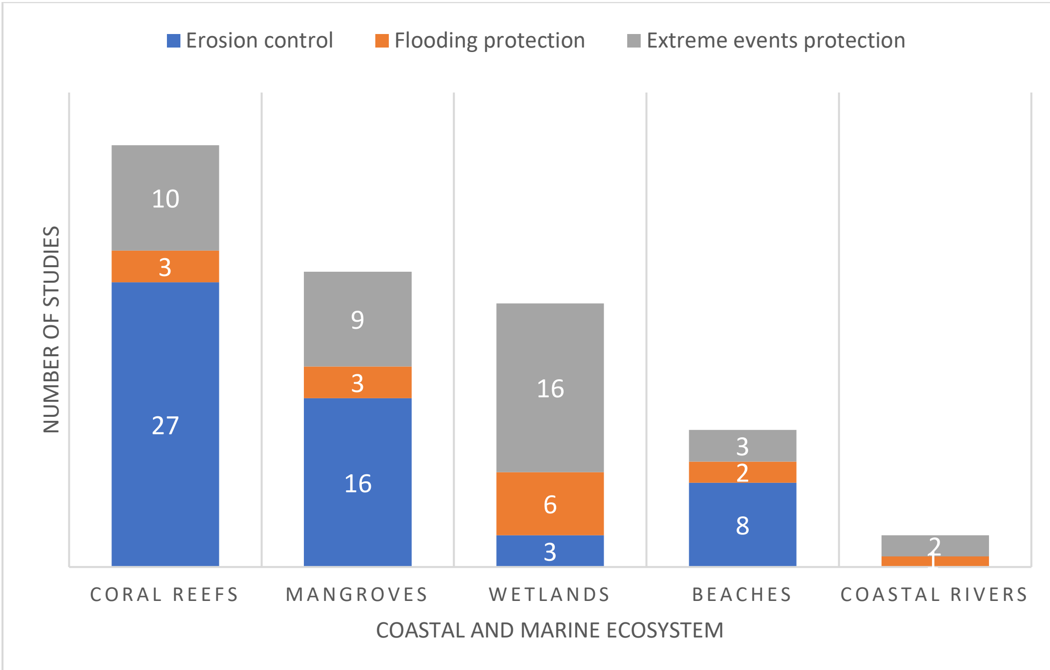
Source: The authors

Out of the 40 coral reef studies reviewed, 27 (68%) value erosion control, with 10 estimating value of protection from extreme events (25%) (Figure 7). Mangroves are the second largest ecosystem valued for its coastal protection service. There are 16 value estimates that address erosion control or shoreline protection (57%). Protection from extreme events such as typhoons, storm surges and hurricanes are addressed by 9 studies (32%). Lastly, the economic valuation of coastal protection from flooding by mangroves is included in 3 studies (11%).

On the other hand, around the world, estimates of the value of the coastal protection service of beaches are mainly related to the provision of erosion control. According to our review of the literature, 13 studies examine the coastal protection service provided by beaches, 8 related to erosion control, 3 to extreme event protection and 2 to flooding protection. In contrast, there are more than 20 economic studies that explore the benefits of protecting beaches for recreation and the value of coastal properties.

In contrast to the number of studies valuing coastal protection services provided by coral reefs, mangroves, coastal wetlands and beaches, only four studies focus on oyster reefs and seagrasses. All of them deal with erosion control.

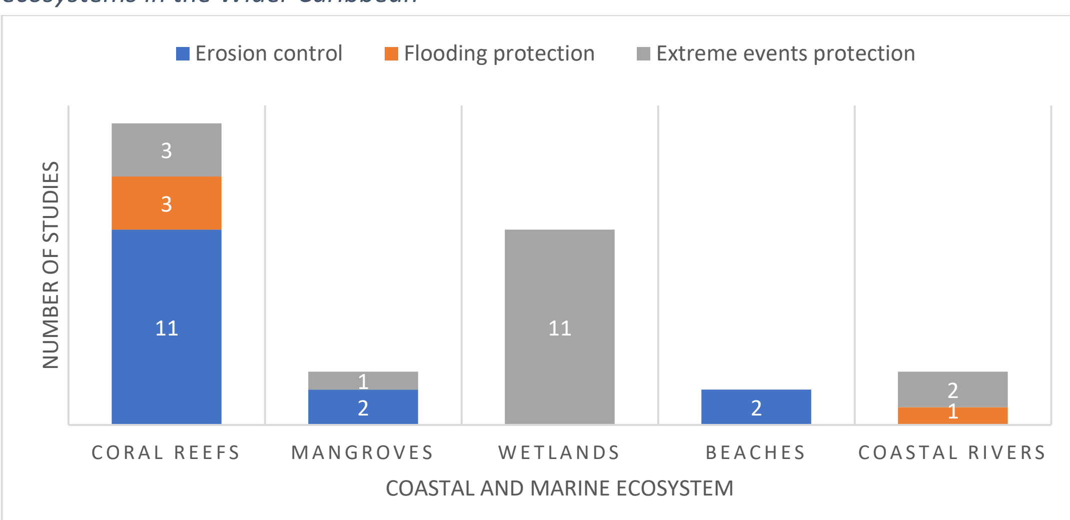
Figure 7. Distribution of economic valuation studies by type of coastal protection and ecosystem



Source: The authors

In the Wider Caribbean, coral reefs have been the most valuated ecosystem, with 17 studies conducted, primarily focusing on valuating their erosion control service (Figure 8). The second most popular ecosystem for valuation studies on coastal protection is wetlands, with all of the 11 studies reviewed focusing on extreme events protection.

Figure 8. Distribution of economic valuation studies by type of coastal protection for different ecosystems in the Wider Caribbean

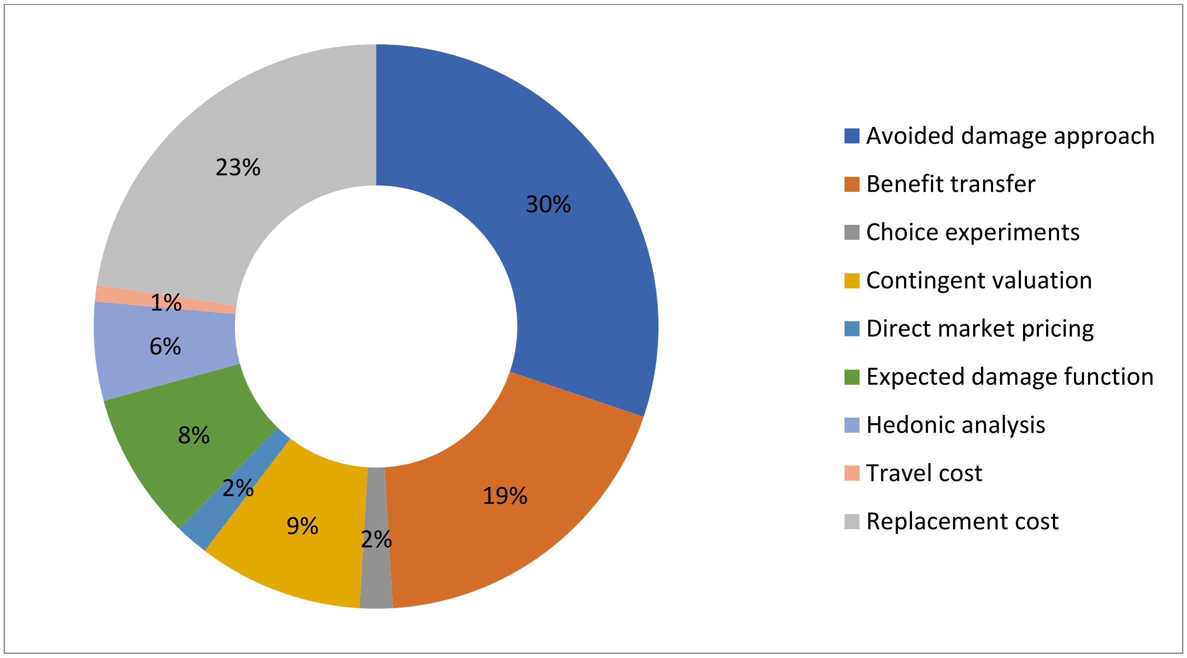


Source: The authors

HOW HAVE THEY BEEN VALUED

The principal methods for deriving coastal protection values is the avoided damage approach (30%) and the replacement cost method (Figure 9). The economic values are estimated mainly on hard infrastructure cost (e.g., seawalls, stone-piled embankment, groins) and natural infrastructure (e.g., beach nourishment) (Leo et al., 2016; Haites et al., 2002). On the other hand, the expected damage function (EDF) approach, along with the hedonic analysis method and travel cost method, only account for 16% of the studies. All of these are revealed preference methods. Barbier (2007) argues that relative to the replacement cost method, the EDF approach provides a more robust measure of the value of coastal protection services, especially for large-scale assessments. We find that 9% of the studies apply the EDF approach, with a focus on the USA, Belize, the Philippines, and Thailand.

Figure 9. Distribution of economic valuation studies around the world by valuation method



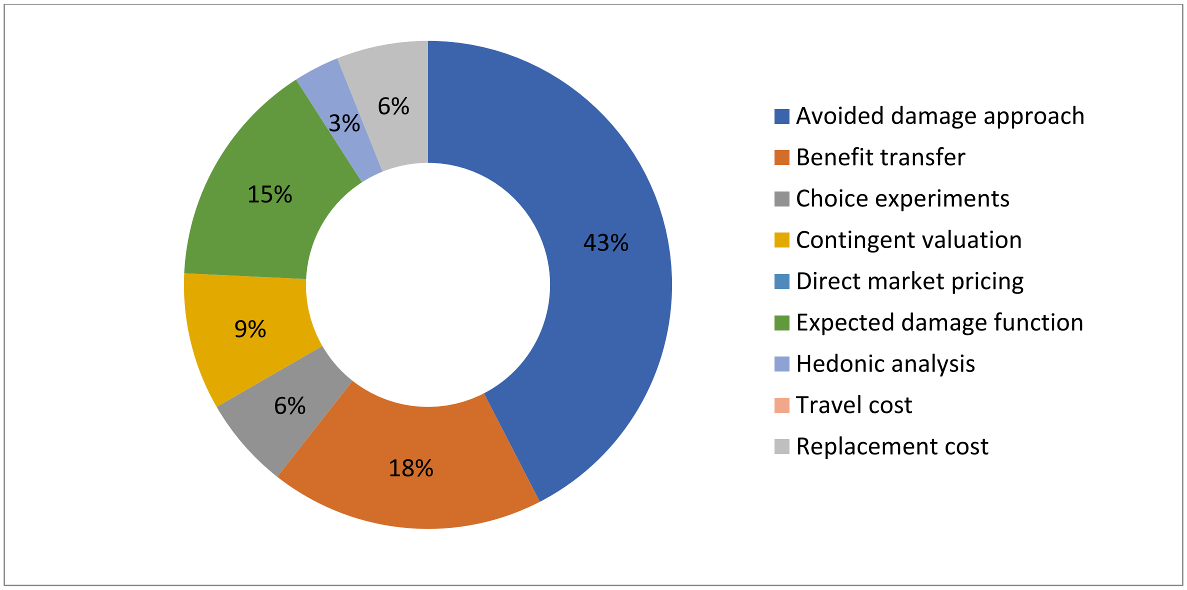
Source: The authors

With regard to stated-preferences methods, contingent valuation is the most typically employed. Stated-preference methods are regularly used to assess protection, management or restoration interventions of marine coastal ecosystems in order to improve the coastal protection services they provide. The economic valuations studies that include evaluation of natural infrastructure rely primarily on contingent valuation, replacement cost and avoided damage cost.

Benefit transfer has been used notably in North America and Asia, and it is often applied to costal protection provided by coral reefs.

Considering the studies on the Wider Caribbean, avoided damage approach continues to be the most recurrently used (43%) (Figure 10). Stated-preference methods are used in 15% of the studies, while benefit transfer approach is used in 16% of them.

Figure 10. Distribution of economic valuation studies in the Wider Caribbean, by valuation method



Source: The authors

WHAT ECONOMIC VALUES HAVE BEEN FOUND?

Results from the valuation of ecosystem services are presented in a very heterogeneous way. Some studies report aggregated values for ecosystem, while other present estimates per unit area. Many studies present the results in year-equivalent estimates but others present them in net present value for a given time horizon and discount rate. In addition, some studies estimate the results in per capita terms, while others present results at the municipality and county scale. Different currencies and study periods were also observed.

Some studies, such as Cesar (2003), estimate aggregate economic values for services. His analysis values the protection service that coral reefs provide against extreme events. According to the results, coral reefs generate annual benefits of coastal protection of US$ 5,047 million in Southeast Asia, US$ 720 million in the Caribbean, US$1,595 million in the Indian Ocean, US$579 million in the Pacific, US$268 million in Japan, US$172 million in the United States, and US$629 million in Australia, totaling US$9,009 million globally.

With regard to coastal protection provided by mangroves, Salem and Mercer (2012) estimate a mean of US$ 3,166/ha/year for coastal stabilization and flood control. They used benefit transfer meta-analyses considering 31 studies related to coastal protection. For coastal wetlands, economic values range from US$583/ha/year to US$11,672/ha/year in the reduction of extreme events such as hurricanes.

ECONOMIC VALUATION APPLIED TO CO-BENEFITS

Identifying suitable economic valuation methods to quantify the value of co-benefits, i.e., the ecosystem services provided by ecosystems beyond those related to coastal protection, is also an

important element of a robust economic valuation. We analyze the three ecosystem service categories defined by the Millennium Ecosystem Assessment: provisioning, regulating (other than coastal protection) and cultural services (MEA, 2005).

Our analysis is based on a literature review of economic valuation studies. Most of them examine cases located in the Wider Caribbean region. However, some studies developed in other regions are also included to show examples of suitable economic valuation techniques to valuate co-benefits from natural infrastructure projects for coastal protection. Given the heterogeneity among studies with respect to their location, context, type of service and methodology, it is generally difficult to compare studies, however, they give a sense of the magnitude of the value of different ecosystem services.

PROVISIONING SERVICES

Provisioning services are sometimes traded on a market. This is the case of fish and other raw materials like timber and wood. As a result, provisioning services might be valuated directly using market prices. Several examples of economic valuation of provision services using market prices are available (Burke et al., 2008; Cooper et al., 2009; Islam and Ikejima, 2010; Kroeger, 2012; Leeworthy and Wiley, 2000; Rönnbäck, 1999; Salem and Mercer, 2012). In these studies, the gross revenue of commercial and non-commercial fishing activities is computed using financial analysis methods. To assess the value of one specific service, some costs such as maintenance, labor and taxes might be subtracted so that net revenues are obtained. However, economic benefits can be greater than net revenues, and a multiplier to account for wider economic benefits from each industry might be used (Burke et al., 2008). A similar approach to the valuation of fisheries using revealed preferences techniques is provided by Burke and Maidens (2004), who make use of the avoided damage cost method to show the lost monetary value in fishery production by coral reef degradation in the Caribbean.

Another common approach to evaluate the role of ecosystems' provisioning services is the use of production function modeling in order to explain how changes in coastal ecosystems can affect the supply of a final good. This method has been primarily used to study outcomes in the market of fish species. Examples include Barbier and Strand (1998), Barbier (2007), Barbier et al. (2002), Barbier (2003), Cuervo-Sanchez et al. (2018), Freeman (1991), Lynne et al. (1981) and Swallow (1994). In essence, these methods assume that the well-being of ecosystems enhances the productivity of some economic activities, and they can be valued as a factor or input of production as a result (Barbier, 2007).

Stated-preference methods have also been used to valuate provisioning ecosystem services, albeit less frequently. The advantage of using this method is the possibility of capturing non-use values as well as use values. For example, Hargreaves-Allen (2008) makes use of the contingent valuation method to capture the value of fishing access to marine reserves in Belize. A few other studies use avoided damage cost (Burke & Maidens, 2004), benefit transfer (Hargreaves-Allen, 2010; Kroeger, 2012) and meta-analysis (Salem & Mercer, 2012).

REGULATING SERVICES

The literature on the economic valuation of regulating services provided by coastal ecosystems – besides coastal protection– focuses on carbon sequestration services. The existent studies on the economic benefits from carbon sequestration provided by coastal ecosystems usually incorporate two general steps. First, the ecosystem's capacity to capture and store carbon is determined. Second, the economic benefits from carbon sequestration are obtained by using estimates of the monetary social value or market prices of sequestered carbon whenever possible. Examples of these studies –that usually rely on simulation and modeling (production function) – include Barbier et al. (2011), Chumra et al. (2003), Kaito et al. (2009), and Zarate-Barrera and Maldonado (2015). Avoided damage cost approaches have also been used to value water purification services.

CULTURAL SERVICES

The main cultural services provided by coastal ecosystems that have been evaluated in the literature are related to aesthetic values and the capacity to provide locations for recreation and tourism. The economic valuation studies of cultural services make use of varied economic valuation techniques, including revealed-preference methods (such as market prices, hedonic prices and travel cost methods) and stated-preference techniques (such as contingent valuation and choice experiments). The majority of reviewed studies use the contingent valuation method (33%), but other methods also appear to be important; for example, hedonic price models to evaluate aesthetic values (15%), and market prices and travel cost methods to evaluate recreational values (18% and 15% respectively).

Economic valuation of coastal ecosystems using market prices has focused on income generation derived from tourist activities. The main tourist activities evaluated include diving, snorkeling, sport fishing and the cruise industry. In addition, these types of studies consider expenditure on different goods such as food, gasoline, lodging and park fees (Cooper et al., 2009; Wielgus et al., 2010; Johns et al., 2001; Leeworthy and Wiley, 2000). Although less common in the Caribbean, economic valuation studies using revealed preferences employ hedonic pricing models that evaluate the aesthetic characteristics of properties adjacent to coastal ecosystems (Landry et al., 2003; Landry and Allen, 2016; Landry and Hindsley, 2011; Pompe, 2008; van Beukering et al., 2011). Finally, travel cost methods constitute an important approach for evaluating the benefits of tourist activities that ecosystems can provide (Parsons et al., 2013; Pendleton, 1995; van Beukering et al., 2011; Wielgus et al., 2010).

In addition to revealed-preference methods, stated-preference methods can be used to evaluate the recreation and tourism services provided by coastal ecosystems. Most of these studies use the contingent valuation method, focusing on tourists' willingness to pay for quality recreation on marine protected areas (MPA) and tourism benefits derived from improvements and enlargements of MPA (Barrera and Maldonado, 2013; Dixon et al., 1993; Hargreaves-Allen, 2008; Rivera-Planter & Muñoz-Piña, 2005; Trejo, 2005; Thur, 2010; Uyarra et al., 2010; Wielgus et al., 2010; Wright, 1995). On the other hand, to evaluate particular characteristics of marine and coastal zones, including MPA,

studies using choice experiments focus on other attributes such as biodiversity conservation and livelihoods of local communities (Montañez-Gil & Maldonado, 2014).

IMPACT EVALUATION OF CONSERVATION PROJECTS

An impact evaluation (IE) is a systematic method for the collection, analysis, and assessment of information to measure the causal effect of a specific policy, program or intervention in relation to stated goals, with regard to a credible counterfactual scenario (McKinnon et al., 2015; Ferraro & Hanauer, 2014; Bottrill & Pressey, 2012). In addition to identifying the effects of a particular intervention, an impact evaluation should provide explanations and understanding of the conditions under and mechanisms through which such effects arise (Woodhouse et al., 2016; Ferraro & Hanauer, 2014). Annex 1: Impact evaluation approaches provides a more in-depth description of the different impact evaluation methodologies.

When referring to a conservation project, impact evaluations can assess the effects on both ecological conditions and socioeconomic outcomes attributable to the actions of the conservation intervention. In recent years, donors, multilateral aid agencies, conservation NGOs, and other stakeholders in the international conservation community have engaged in promoting and favoring evidence-based interventions and recognized the relevance of carrying out impact evaluations. The reason is threefold: first, it is critical to identify whether interventions fail or succeed and why. In particular, more knowledge on what works for both ecosystem conservation and human well-being is required. Second, the international conservation community needs to learn how to design and implement cost-effective programs that demonstrably make an efficient use of the limited budgets allocated to conservation projects to achieve desired conservation outcomes. Third, greater transparency and accountability to donors and society needs to be offered by demonstrating positive returns on conservation investments (Woodhouse et al., 2016; Baylis et al., 2016; Roe et al., 2013; Bottrill et al., 2011; Margoluis et al., 2009; Ferraro & Pattanayak, 2006). In addition, conservation donors and organizations are also interested in emphasizing the effect of conservation projects on socioeconomic outcomes, acknowledging the contribution of conservation interventions to the maintenance or improvement of ecosystem services on which humans depend (Bottrill et al., 2014).

Despite the fact that the conservation community has begun to recognize the relevance of impact assessments, impact evaluations of conservation projects remain scarce (Curzon & Kontoleon, 2016; Margoluis et al., 2009). Financial, temporal, logistical, methodological and ethical challenges related to carrying out rigorous impact assessments have hindered the practice of impact evaluations. Other reasons also explain the modest number of available impact evaluations of conservation projects. First, it is difficult to carry out evaluations when it is not possible to control for external factors (e.g., extreme climate events). Second, there is typically selection bias associated with many conservation projects (e.g., the establishment of protected areas or the devolution of rights to local communities for the management of common pool resources; these locations are by nature usually not randomly selected). Third, the appreciation and understanding of the need for counterfactual thinking in the

evaluation of conservation interventions remains limited at best. Fourth, baseline and historical data are not generally available. Fifth, the challenges associated with the nature of conservation interventions make the use of orthodox impact evaluation approaches inviable (e.g., the difficulty or impossibility of finding a valid counterfactual). Finally, understanding and integrating the interactions between natural and social systems in evaluation frameworks is a complex endeavor (Banerjee et al., 2016; Curzon & Kontoleon, 2016; Bottrill et al., 2014; Roe et al., 2013; Ferraro, 2009; Ferraro & Pattanayak, 2006).

In spite of these challenges, impact evaluations of natural infrastructure interventions for coastal protection are fundamental for an effective ICZM policy for at least four reasons (Banerjee et al., 2016; Baylis et al., 2016; Margoluis et al., 2009; Gertler et al., 2011): First, they constitute standard requirements for loans and grants from multilateral development institutions as they provide concrete indicators on the benefits generated by their investments. Second, they assess whether an intervention reached tangible and intended outcomes offer lessons and insights to improve the design of future and cost-effective interventions. Third, they ensure value for money and accountability and provide evidence that supports policy making. Finally, they offer information on the possibilities of scaling up successful programs.

IMPACT EVALUATION APPLIED TO CONSERVATION INTERVENTIONS

In this section, we present a review of 51 impact evaluations that assess conservation policies, projects or interventions with environmental and/or socioeconomic outcomes. We highlight studies from the Wider Caribbean but relevant studies from other regions are also included.

Our review shows that different research designs were employed to carry out impact evaluations of conservation interventions, ranging from experimental (4%) and quasi-experimental (75%) to non-experimental (8%) and qualitative (10%), as well as mixed methods (4%). The majority of impact evaluations of conservation interventions focus on biophysical outcomes (43%). Although bio-physical indicators provide useful evidence to external stakeholders, they may fail to capture the multidimensional nature of conservation policies, especially when conservation activities may relate to people and their well-being. Among the biophysical indicators related to environmental outcomes, we find deforestation and forest cover (51%), biodiversity conservation (12%) and erosion (2%). Research has also been conducted to study the impact of conservation interventions on socioeconomic outcomes. This line of research typically focuses on welfare and economic activity impacts. Examples of socioeconomic indicators are fishing income and net earnings from commercial fisheries, economic growth, and food security, as well as health and mortality rates.

We now discuss several impact evaluations on marine protected areas – as an ICZM strategy – and on coastal natural infrastructure, highlighting main findings and challenges.

IMPACT EVALUATIONS OF MARINE PROTECTED AREAS

Most of the impact evaluations we reviewed assess continental ecosystems, with only 5 out of 51 studies focusing on marine areas. There are no experimental evaluation designs that evaluate management policies on marine ecosystems. This is not surprising considering that these policies (e.g., marine reserves) are not designed in a way that facilitates the measurement of their causal impact (Smith et al., 2006). For instance, the implementation of a marine reserve rarely restricts an area to the activities of one group of fishermen while others are allowed to fish, thereby impeding estimation of the counterfactual outcomes that would have occurred without the presence of the reserve (Reimer & Haynie, 2018). Accounting for a counterfactual may be possible in area-based fishery management approaches such as territorial used rights in fisheries (TURFs). However, Mascia et al. (2017) highlight that an impact evaluation may be easier to conduct where harvested species are sedentary, because this minimizes spillover effects, or otherwise in a situation where an intervention affects a subset of the social-ecological systems resource creating viable control sites.

In addition, Reimer and Haynie (2018) mention that marine reserves do not typically occur in isolation. Other factors that influence fishing-related outcomes change simultaneously, such as total allowable catches, prices, abundance, climate change, etc. Thus, by simply using outcomes from before and after the implementation of a marine reserve one may not isolate the effect of the restrictions from other simultaneous changes. This is what Ferraro et al. (2018) refer to as no interference. However, Bucaram et al. (2018) attempt to experimentally assess the impact of a marine protected area (MPA), with the analysis of the Galapagos Marine Reserve (GMR). The authors conduct an evaluation of three types of pelagic tuna species and the influence of the MPA in terms of productivity for the industrial tuna fleet in the GMR, the exclusive economic zone (EEZ) and a control area. Using several quasi-experimental methods, the authors show that the MPA did have a positive effect on productivity.

Miteva et al. (2015) examine whether MPAs in Indonesia were effective in avoiding mangrove loss and emissions of blue carbon between 2000 and 2010 and whether the effectiveness differs across types of protection using quasi-experimental techniques (a combination of propensity score and covariate matching, differences-in-differences, and post-matching bias adjustments). Their results show that MPAs have led, on average, to a 10% reduction in mangrove loss from 2000 to 2006.

IMPACT EVALUATIONS OF NATURAL COASTAL INFRASTRUCTURE

There are very few impact evaluations of natural coastal infrastructure and its effects on the protection from extreme climatic events and other natural events. Moreover, studies often use methodologies that are not usually considered under the scope of traditional impact evaluations. It is difficult to determine whether effects occur due to a natural process or because of human activities (Hunt et al., 2014). In addition, both natural infrastructure and the negative effects they are trying to prevent are being influenced by human activities. As a result, impact evaluations of natural infrastructure projects for disaster prevention must consider anthropogenic variables in order to truly assess the effects of nature-based solutions.

Qiu and Gopalakrishnan (2018) use Hurricane Sandy as a natural experiment to estimate the perceived risk reduction value of shoreline stabilization. Throughout a DDD model (triple difference estimates), they show that investments in beach nourishment result in a price premium of 11.7%-16.5% for oceanfront homes located on a nourished beach.

Corral and Schling (2017) evaluate the impact of a shoreline stabilization policy in Barbados over medium-term economic growth. Their study concentrates on the evaluation of a policy that prevents the retreat of the shoreline and enhances beach amenities using solutions that include natural infrastructure (restoration of coastal habitats). As an approximation for economic growth, the authors use a measure of remotely sensed nighttime lights with a deblurring methodology. The authors apply the synthetic control method first proposed by Abadie and Gardeazabal (2003) as a way to systematically choose comparison units (beach sites) combined with bootstrapped confidence intervals to allow for precise quantitative interference in this small-sample study setting. Their results show a difference in annual local GDP of 11.7% between treated beaches and the synthetic counterfactual.

It is worth noting that the use of such non-traditional methods and data sources provide new opportunities for the evaluation of conservation projects that face the above-mentioned limitations to traditional evaluation techniques.

V. CHALLENGES, INNOVATIONS AND RECOMMENDATIONS

The main objective of this document is to support, disseminate and mainstream the practice of economic analyses of coastal protection and restoration investments in the Wider Caribbean region. We give a special emphasis to natural infrastructure and coastal ecosystem restoration activities for their recognition as important mechanisms for ICZM.

In this section, we present the challenges that have been identified with respect to the use of cost-benefit analysis (CBA), economic valuation (EV) and impact evaluation (IE) as tools for the economic assessments of natural infrastructure projects, coupled with a series of innovations associated to the use of these tools. In some cases, these innovations can help overcome the identified challenges, while in others they imply an even greater challenge. To overcome these challenges, we present, along with relevant innovations, a series of recommendations that are applicable to the Wider Caribbean.

We develop a framework for the economic analysis of natural infrastructure and coastal ecosystem restoration projects, which is depicted in Figure 11. Although we focus on natural infrastructure and coastal ecosystem restoration, we recognize that this framework can be adjusted and applied to other ICZM policy alternatives as, for example, managed realignment. Our framework is intended to be a step-by-step process suited for the three types of analysis (CBA, EV and IE).

The first panel in Figure 11 (shadowed in gray color), indicates that before conducting an economic analysis, the information available on the project should be identified. It is important to identify the different technically viable alternatives of infrastructure (i.e., natural, hybrid or gray), as well as the biophysical and socioeconomic studies that support them.

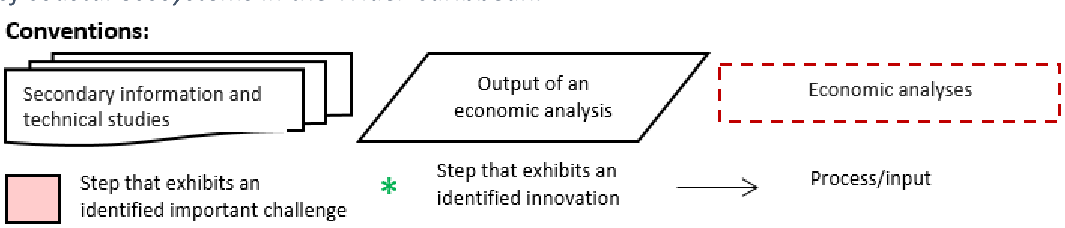
Following this, the necessary inputs for the analysis have to be identified (green panel in Figure 11), including information on the lifespan of the project, the planning horizon, its associated costs (both investment and operational), the relevant physical, biophysical and socioeconomic indicators, the intervened ecosystems, and the involved stakeholders.

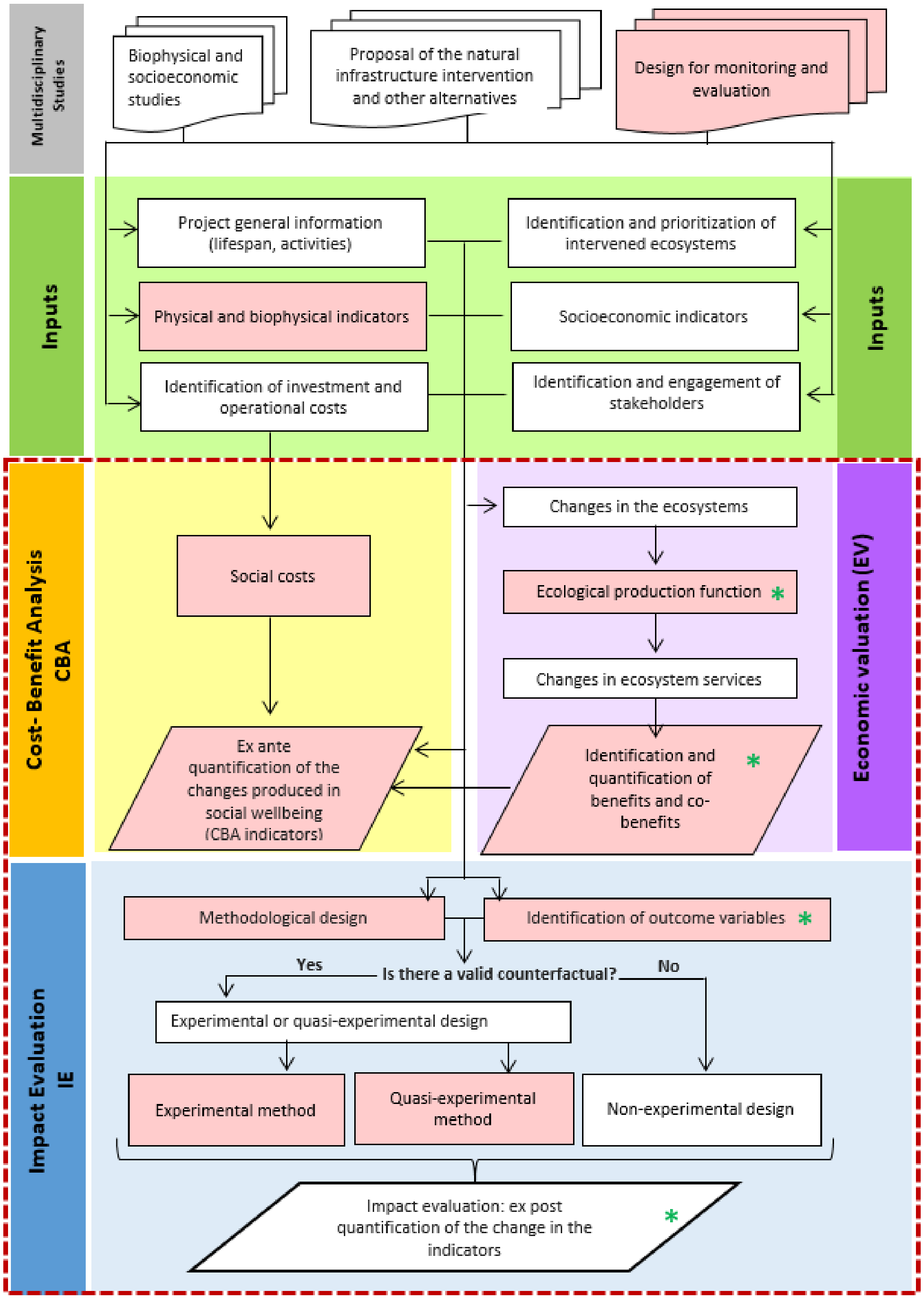
With this information, the economic analysis begins. First, the CBA allows for an ex ante evaluation of the intervention, quantifying the wellbeing that could be obtained by implementing a project, taking into account all costs and benefits from a social perspective. In many cases, however, the economic benefits of a project are not directly or monetarily observable. Thus, it is important to consider economic valuation in order to quantify these benefits and include them in the cost-benefit analysis. Second, an IE assesses the effects of the intervention on different indicators after its implementation. This type of analysis is valuable when it comes to identifying whether an intervention achieved the expected socioeconomic or biophysical results.

It is important to mention that although the three methods for economic assessments are located in the general process described in Figure 11, they have different objectives and can be carried out independently.

The remainder of this section discusses the main lessons learned in this research, highlighting the challenges to be faced (marked in red in Figure 11) and the innovations proposed in the literature to overcome them (marked with a green asterisk in Figure 11).

Figure 11. Framework for the economic analysis of natural infrastructure projects and restoration of coastal ecosystems in the Wider Caribbean.





UNDERTAKING THE COST-BENEFIT ANALYSIS OF NATURAL INFRASTRUCTURE PROJECTS

As shown in Figure 11, the purpose of the CBA is the ex-ante quantification of the change in the social welfare resulting from the implementation of a project, with respect to the initial situation, or different alternatives. Even when the CBA appears methodologically simple to carry out, there are several challenges that need to be addressed for a robust analysis that captures the complete set of benefits and costs associated with any given intervention (Table 7).

Table 7. Challenges of carrying out CBA for natural infrastructure projects.

Challenge	How to address it?	Possible to overcome?
Including the opportunity cost of not executing the project.	• Identify -at least conceptually- the opportunity cost of the project, in order to have an idea of its magnitude and its effect (overestimation or underestimation) on the net benefit. • Approximate the opportunity cost using studies on the economic impact of climate change (benefit transfer).	Short-term
Selecting an appropriate discount rate for natural infrastructure projects.	• Perform sensitivity analysis in the CBAs. •	Short-term
Considering the benefits from a wide array of ecosystem services.	• Prioritize the most relevant ecosystem services, based on secondary information or focus group exercises with stakeholders.	Short-term
Estimating the benefits and co-benefits that emerge from ecosystem services not traded in markets.	• Collect data for processes, ecosystem structure and ecosystem service production. A good example is the data sources associated to InVEST performance (shown in Table 23). • Train natural science scientists, engineers, practitioners, and policy makers in economic valuation so that they understand the needs and characteristics of the information required to conduct economic valuation studies. • Promote multidisciplinary research. • Promote the creation and improvement of databases and platforms with reliable and periodic information on relevant biophysical and socioeconomic indicators. • Combine methods for the identification of ecosystem services production and valuation of these services.	Medium to long-term

The first challenge refers to the need to *include the opportunity costs* associated with not implementing the project. It is not uncommon to find CBA where the costs of not carrying out a project are ignored. They reflect the opportunity cost of not executing a project, such as the increased risk of flooding or erosion with its corresponding effects on communities. Ignoring these costs underestimates the net benefit of a project. Similarly, overlooking the benefits that would be provided regardless of the project may bias the results upwards, overestimating the net benefit of the project by attributing all of the benefits to the intervention.

The costs and benefits of a project are usually observed over a long period. In this case, it is important to convert these values to a common currency, and to discount the cost and benefit flows. The *choice of the social discount rate* is one of the critical elements in the CBA. Particularly for natural infrastructure projects, the choice of a discount rate has important implications over time.

For example, using an exponential discount rate implies that for natural infrastructure and other long-term projects, the future becomes less and less valuable, putting them at a disadvantage vis-à-vis projects with shorter time horizons (such as grey infrastructure). This can introduce distortions when alternatives are selected (Campos et al., 2016). Drupp et al. (2015) recommend a mean (median) long-term social discount rate of 2.25% (2%) or any within the interval of 1% to 3%; US EPA (2013) use rates between 4% and 4.857%, whereas Nordman et al. (2018) suggest a 3.5% discount rate. CBAs of IDB projects apply a 12% discount rate.

A third challenge is *the quantification of all the costs and benefits engendered by the project*. With respect to the benefits, ecosystems can provide a large amount of services and sometimes not all of them can be included in the analysis because of time, budget or technical constraints. It is important to capture the most important values by selecting the most relevant services in each context. This exercise can narrow the analysis to a number of services that makes the CBA exercise practical. This can be done based on secondary information and with the participation of relevant stakeholders. Regarding costs, it is also important to consider shadow prices in order to capture the social costs of the project. In the CBA panel, Figure 11 shows that these costs and benefits are used to quantify the change in wellbeing associated with an intervention.

Perhaps the main challenge facing the CBA of natural infrastructure projects is *to estimate the costs and benefits of non-market ecosystem services*. When some services –as is the case of provisioning services such as fish or timber- are traded in markets, benefits from ecosystems can be partially included in the CBA. However, typically, coastal protection and other ecosystem services are not traded on the markets and, therefore, their prices are not observable. Under such circumstances, the estimation of the direct benefits and co-benefits provided by natural infrastructure constitutes a challenge for the CBA. By ignoring these nonmarket values, the benefits from a natural infrastructure project will be undervalued and other alternatives (such as gray infrastructure) may seem more profitable. It is therefore necessary to use appropriate economic valuation methods for the ecosystem services provided for each specific natural infrastructure project for coastal protection.

The main challenges to tackle in using economic valuation as a tool to estimate benefits of non-market ecosystem services are discussed in the following section.

UNDERTAKING THE ECONOMIC VALUATION OF COASTAL PROTECTION SERVICES

The purpose of an economic valuation of a natural infrastructure project is to estimate the benefits obtained from natural infrastructure due to changes in coastal protection and other ecosystem services (co-benefits) such as fishing, recreation, increases in property values, and so forth.

Given that natural infrastructure projects are likely to be affected by climate change, economic valuation of such projects faces a number of challenges (Table 8) associated to the ecological production function, as shown in Figure 11.

The first challenge is that the knowledge available to establish linkages between the changes in the ecosystem structure and the production of valuable ecosystem services is inadequate (Barbier, 2013). In other words, we often do not know how variations in ecosystem structure, functions and processes -whether in terms of quality or quantity- resulting from a natural infrastructure project lead to changes in the flow of ecosystem services. This relationship is typically established through ecological production functions, which was defined in section 4. An ecological production function establishes a relationship between ecosystem services (products) and changes in the ecological structure (inputs), which result as a result of an intervention to ecosystems. The main challenge associated with ecological production functions is to have accurate and reliable information to establish this relationship.

Table 8. Challenges of carrying out economic valuation for natural infrastructure projects

Challenge	How to address it?	Possible to overcome?
Linking the changes in the ecosystem structure and the provision of ecosystem services (ecological production function).	• Collect data for process, ecosystem structure and ecosystem services provision for the definition of ecological production functions. • Train natural science researchers in economic valuation, so that they understand the needs and characteristics of the information required to conduct economic valuation studies. • Promote multidisciplinary research. • Provide training on coastal modeling tools.	Medium to long-term
Incorporating uncertainty related with the changing conditions caused by climate change.	• Promote multidisciplinary research. • Provide training on coastal modeling tools and the evaluation of climate models.	Medium-term
Selecting and applying the most appropriate economic valuation method, according to the ecosystem and ecosystem service to be valued and the availability of information and resources.	• Improve and provide access to databases with reliable and up-to-date information. • Promote the creation and consolidation of databases that include relevant socioeconomic information with minimum quality standards. • Train scientists to use coastal modeling tools. • Recognize and disseminate information about databases available and data requirements. •	Medium to long-term
Carrying out economic valuation for policymaking.	• Train policy makers so that they understand the different uses of economic valuation: ○ To justify the choice between natural and gray interventions. ○ To support funding requests for natural interventions. ○ To train policy makers in understanding the different uses of economic analyses for assessing coastal management and adaptation strategies for segments of coasts. ○ To justify increases in taxes or budget distribution.	Medium-term

Challenge	How to address it?	Possible to overcome?
	○ To support the design of regulatory alternatives adopted by agencies.	
Prioritizing ecosystem services	• Strong engagement with stakeholders	Short-term
Addressing ecosystem connectivity and synergies	• Reviewing valuation methodologies to allow including values of synergies between ecosystems	Short-term

The use of coastal modeling tools are useful when dealing with this challenge. They help identify ecosystem services in relation to changes in ecosystem structure. For example, InVEST identifies the ecological functions provided by ecosystems (supply), then links these functions to the demand, considering the beneficiaries of the ecosystem services (service), and finally includes social preferences to calculate the economic and social metrics (value) (Sharp et al., 2018).

Other modeling tools include the Artificial Intelligence for Ecosystem Services (ARIES) (World Bank, 2016), the coastal resilience tool (TNC, 2016), the coastal hazard analysis modeling program (CHAMP) (FEMA, 2007; FEMA, 2013), Toolkit for Ecosystem Services Site-Based Assessment (TESSA) (BirdLife International, 2018), XBeach (Nederhoff, 2015; Roelvink et al., 2015), and Climate adaptation (CLIMADA) (Bresch et al., 2018).

At this particular point, it is necessary to consider the non-linearity of the ecological functions. Barbier et al. (2008) suggest incorporating the nonlinear ecological function concept into economic valuation, particularly for coastal protection services provided by coastal interface ecosystems. This is an important innovation since it is usually assumed that ecosystem services change linearly in relation to changes in habitat or environmental variables (Canadell, 2000; Burkett et al., 2005; Barbier et al., 2008). Additionally, it allows the possibility to propose intermediate conservation strategies coherent with ecosystem-based management strategies (Barbier et al., 2008), which emphasize reconciliation between economic development and conservation of critical ecosystem resources and services, an approach embraced by ICZM.

Uncertainty caused by the effect of climate change on ecosystem structure and functions is another challenge that needs to be taken into consideration. Some modeling tools have been used to address climate change uncertainty. For instance, Arkema et al. (2013) use the coastal vulnerability model of InVEST to create an exposure index for the US coast, simulating five scenarios of sea level rise. With this model, they identify future vulnerability under different sea level scenarios. Arkema et al. (2014) also used INVEST to estimate the risk posed to habitats from multiple stressors in Belize. The model evaluates the risk for ecosystems in terms of exposure and consequence. Additionally, CCRIF (2010) estimate the economic losses from climatic risks in eight countries in the Caribbean, evaluating the cost-effectiveness of 20 adaptation measures. They use an approach similar to the one used by insurance companies, considering hazard, value and vulnerability modules as inputs.

A third challenge is *the selection of the economic evaluation method* to be used in each case and the information available for its rigorous application. The selection of the approach to be used depends on the type of ecosystem service to be valued and the type of value it generates (use value -direct

or indirect- or non-use value -option or existence-). Revealed preferences methods serve to capture use values, primarily direct values, but also, in some instances, indirect ones.

In some cases, when time is short or resources are scarce, or as a first approximation to the defined valuation exercises, the benefit transfer approach is used. However, benefit transfer may lead to erroneous or inaccurate interpretation of results, so applying this technique requires recognition of its limitations.

It is important to consider that the economic valuation of the coastal protection service has an important limitation compared to other ecosystem services, as it can become context-specific. This means that, given the interdependence of ecosystems, and at the same time their uniqueness, changes in the level of coastal protection is not associated exclusively with one ecosystem (even if the change comes from the modification of only one of those ecosystems), but may come from several, depending on the diversity and interaction between ecosystems in a given area. Furthermore, as discussed in Chapter 1, the levels of protection provided by these ecosystems depend on their characteristics. Given the context-specific condition of biophysical studies, benefit transfer becomes an even more restricted method for valuing coastal protection services.

As mentioned at the beginning of this section, empirical estimates of the benefits stemming from changes in the shoreline environment have focused primarily on recreation and hedonic housing values. Two innovations can be identified in the processes related with the valuation of these services.

The first innovation involves the use of instrumental variables for endogeneity bias in the hedonic price. Beach nourishment is one of the most frequent erosion control strategies for beaches around the world. Usually, hedonic price methods are applied to estimate the value of beach width that is reflected in property values. However, economic valuations usually ignore the presence of endogeneity bias: coastal property prices are influenced by beach width and nourishment decisions (which influence the beach width) also depend on benefits from increasing width, then the width of a beach becomes endogenous in the exercise. To overcome this challenge, Gopalakrishnan et al. (2011) use instrumental variables, such as distance to continental shelf and beach quality attributes to estimate the implicit price of beach width.

The second innovation involves the use of geo-economic models for beach erosion management. Models have recently been developed to explore the interactions of geomorphological processes with economic decision-making about shoreline management (Gopalakrishnan et al., 2016; Gopalakrishnan et al., 2017). These models consider a community that wants to stabilize its shoreline by deciding whether, when, and how much to implement infrastructure for beach management. Thus, the community is faced with the choice of whether to build structures (e.g., how often to nourish the beach and extent how far out to build the beach) in order to maximize a stream of net benefits subject to the evolution of the geophysical system and institutional constraints.

Another challenge relates to the difficulty in using the results of economic valuation exercises in the design of public policy. Part of this difficulty is due to the way in which assessment results are communicated to decision-makers, but also due to the lack of decision-makers' training in this area. In principle, it is important to know that CBAs can help decision-makers allocate scarce resources to competing social demands, but they generally do not take into consideration the benefits provided by coastal ecosystems. At this point, economic valuation plays a fundamental role because it can identify the benefits that are usually associated with non-market values (Waite et al., 2014).

The identification of stakeholders and their engagement in the process represents another challenge for the economic valuation of ecosystem services. Stakeholders might or might not be able to influence decisions. When designing an economic valuation, and performing an intervention, it is important to think about how each of the stakeholders might be affected (gains and losses) and to identify possible stakeholder conflicts and trade-offs (Waite et al., 2014; Schuster & Doerr, 2015).

Some works have also considered the engagement of stakeholders as an important step for the identification, prioritization and valuation of ecosystem services (Sheil & Liswanti, 2006; Lynam et al., 2007). These studies represent efforts to consider the importance that local resource users assign to ecosystems and the services they provide. For example, Moreno-Sanchez and Maldonado (2011) use an approach named participatory valuation, which is based on weight distribution methods to evaluate the importance that local communities assign to different landscape units and ecosystem services in the coastal hydrological complex of Ciénaga Grande de Santa Marta in the Colombian Caribbean. In another case study, Maldonado et al. (2019) apply a similar approach where they combine participatory valuation and choice experiment to evaluate different characteristics of a conservation policy for mangroves in Ecuador. These two studies represent an innovation for the economic valuation of ecosystem services with local communities. Although the first case does not stand as an economic valuation exercise itself, it provides an effective methodology to engage primary stakeholders and understand their preferences and priorities about ecosystem services. In the second case, the participative approach contributes to facilitate the understanding of the valuation exercise that is proposed to local communities.

Finally, a last challenge for the economic valuation on coastal ecosystems is related to the existence of synergies and trade-offs among ecosystems -and among ecosystem services. According to Barbier et al. (2011) the interactions among coastal ecosystems could generate services that in the aggregate are higher than those provided by individual ecosystems. This ability is referred to as the connectivity of coastal ecosystems, which produces synergies that enhance the provision of ecosystem services with respect to the one that would be produced by an isolated single habitat.

UNDERTAKING THE IMPACT EVALUATION OF CONSERVATION PROJECTS

Despite the fact that the conservation community is increasingly interested in carrying out IE of conservation interventions, challenges for the application of rigorous research designs remain (Baylis et al., 2016). In Table 9, we present some of the most relevant challenges discussed in the literature, as detailed below.

The first challenge considers the *complexities* related to the setting where the interventions take place. In many cases, conservation projects are carried out under dynamic conditions characterized by uncertainty (i.e. climate change) and high complexity (i.e. integration of biological and social systems) (Margoluis et al., 2009). Additionally, ecosystems are complex systems with non-linear dynamics at various spatial and temporal scales (Baylis et al., 2016). The complexity associated with socio-ecological systems and the lack of knowledge about their interactions makes it difficult to establish conceptual causal linkages among intervention activities, outcomes and impacts. Sometimes even data requirements are unclear (Margoluis et al., 2009).

Table 9. Challenges of carrying out impact evaluation for conservation interventions

Challenge	How to address it?	Possible to overcome?
Complexities related to the setting where intervention take place.	• Collect data for process, ecosystem structure and ecosystem services production, including the use of innovative data sources (remotely sensed, etc). • Robust knowledge of empirical strategies at the disposal of the analyst. • Train decision makers (government officials, policy makers, local community) in impact evaluation so that they understand the information requirements.	Long-term
Small scale of initiatives and funding for evaluation	• Allocate resources to evaluate interventions. • Train decision makers (government officials, policy makers, local community) in the importance of running rigorous evaluation of interventions. • Focus on short- to medium-term outcomes to evaluate if the project is effective at that level, and promote longer-term evaluations a few years after project execution has ended.	Short-term Medium-term
Logistical challenges of the evaluation implementation	• Include impact evaluation studies into the design of natural infrastructure projects. • Identify available data and gaps. • Clearly state the expected results in natural infrastructure projects. • Use of innovative data sources, such as from remote sensing.	Short-term
Methodological challenges: non-random location/ selection of the intervention; evaluation outcomes vs. intervention units and scales; confounding factors; spatial spillovers	• Design natural infrastructure projects that include socioeconomic and biophysical variables as outcome results. • Monitor socioeconomic and biophysical variables throughout the project (even after the project). • Mainstream impact evaluation practices among policy makers and the importance of the projects to be implemented. • Design better methodological approaches to estimate causal effects (use of synthetic control methodology, for instance)	Medium-term Long-term (Depending on the intervention)

The second challenge in the overall process of impact evaluation involves the *small initiatives and funding* for this type of analysis. Conservation interventions are often implemented by small organizations with multiple funding sources and are carried out gradually over many years. In many cases, changes in outcome variables are small or they can only be evident in the long run. Additionally, the scarce resources allocated to conservation projects –as well as time constraints– restrict the possibilities of using funds to design and apply impact evaluation studies (Baylis et al., 2016; Bottrill et al., 2011; Curzon & Kontoleon, 2016; Margoluis et al., 2009).

Third, understanding planning and data requirements is essential for the development of a rigorous impact evaluation and overcome the *logistical challenges* of implementing the evaluation in a rigorous and timely manner. Usually, this process is not incorporated into the intervention design and, when it is, implementation challenges and/or impromptu modifications of the project impede the adequate collection of baseline data (Banerjee et al., 2016; Bottrill & Presey, 2012; Margoluis et al., 2009). Many impact evaluations of conservation interventions emerge once the project has finished and no baseline data are collected (Banerjee et al., 2016; Bottrill et al., 2011; Ferraro, 2009; Margoluis et al., 2009). Additionally, conservation projects generate both ecological and socioeconomic impacts. This requires the combination of ecological, geographic, demographic and socioeconomic data to accurately measure relevant effects. This challenge can be dealt with in the short term if the impact evaluation is part of the design of the intervention and if databases and gaps are previously identified.

Once the complexity of the conservation project is assessed and the expertise of the team is settled, the impact evaluation should be designed, taking into account the *methodological challenges*, related among other things to differences in desired evaluation outcomes and the intervention units and scales. As shown in Figure 11, this is one of the two central components of the methodological design of an impact evaluation. Outcome and impact variables can be physical, biological, social, or economic. Since ecological processes are associated with spatial units, it is necessary to account for the appropriate spatial scale when evaluating the impact of conservation interventions. However, even when the evaluation employs an adequate spatial scale, measuring impacts can be complex because of the presence of spatial spillovers; that is, that outcomes in some observations are affected by interventions in neighboring units. Not only do spatial spillovers affect outcomes, but they also yield biased impact estimators. Spillovers might be the result of both ecological processes and behavioral responses of target communities in conservation interventions (Baylis et al., 2016; Ferraro, 2009).

In this context, the identification of the most adequate evaluation design can be quite challenging. This decision depends on the existence of a valid counterfactual. If a control group can be identified, experimental or quasi-experimental methodologies can be used. However, analysts need to be aware of potential spatial spillovers, confounding factors, the choice of controls, and randomization limits. The use of non-traditional evaluation techniques and data sources, such as the synthetic control method or remotely sensed data, may present an opportunity to overcome the identified challenges in the future.

Regarding the presence of confounding factors, the choice of controls, and randomization limits, the literature warn that appropriate control groups that discard rival explanations may be difficult to find (Roe et al., 2013). Sometimes it is not possible or desirable to randomly select and separate treatment and control groups (Banerjee et al., 2016; Baylis et al., 2016). Randomization is viable with large samples and high replication; however, large samples and replication may be difficult to attain when the intervention covers a large geographical area or when the evaluation involves unique treatment units without units of comparison, such as restricted habitats of endemic species (Ferraro, 2009). Additionally, in many cases it is unethical or not politically viable to randomly assign, for example, conservation regulations, restoration projects or economic incentives for conservation. On approach that can help mitigate this risk is using geographical units to evaluate impacts. This approach is common in impact evaluations of interventions in natural environments (see for example Corral & Schling, 2017; and Robalino et al., 2015).

In some cases, units from which the outcome of an intervention will be measured are mobile (the individuals that are affected by the intervention do not necessarily reside in the site where the project is implemented), and therefore it becomes impossible to prevent their exposure to the intervention. Such is the case of tourists in beach restoration projects (Banerjee et al., 2016). In other cases, confounding factors are not easily observed, and the use of instrumental variables is limited.

When it is not possible to identify a suitable counterfactual, an evaluation based on non-experimental methods remains an option. In this case, a before and after comparison can provide information on the possible impacts of an intervention, although not necessarily attributable to the project. In large area conservation projects, this may be the only possible means of evaluation.

Another challenge is that relative to the *time lag between the implementation of the project and the manifestation of program effects*. In conservation projects, the time lag between the implementation of the intervention and the observation of biological impacts is often very long and extends beyond the life span of the intervention itself (Bottill et al., 2011). Changes in ecological variables and effects on human wellbeing can occur over many decades. Attributions to a particular intervention may be difficult to establish as a result (Ferraro, 2009; Ferraro & Pattanayak, 2006; Margoluis et al., 2009).

SUMMARY RECOMMENDATIONS FOR POLICY MAKERS

Given the framework and procedures discussed, as well as the challenges and the innovations we identified in the literature for the economic analyses of coastal interventions and ecosystem restoration investments, we present some recommendations to be considered when implementing these analyses:

1. *Start the planning of the economic evaluation process at the stage of project design together with the definition of an analytical framework*. The design of natural infrastructure interventions implies a multidisciplinary work that involves identifying the socioeconomic

context that is intervened, as well as the ecosystems and services being altered. This identification requires the engagement and participation of a diversity of stakeholders. Many impact evaluations of conservation interventions emerge once the project has finished and fail to take into account evaluation requirements at the beginning of the intervention. This is an obstacle for the adequate collection of baseline data. This can be overcome if economic analysis is adequately incorporated into the design of the intervention.

2. *Establish an environmental economics supporting team*. A multidisciplinary approach is fundamental for the formulation, implementation and evaluation of natural-infrastructure interventions. However, for the economic evaluation of the intervention, the team must be led by environmental economists that have knowledge and awareness about the challenges and innovations discussed, and that can follow the steps defined and proposed in this note for each type of economic assessment in a rigorous fashion.
3. *Disseminate economic analyses among stakeholders*. Given that this is a relatively new topic, it is important to guarantee that key stakeholders have access to all the materials pertaining to an economic analysis, including the approaches and the results –both, at baseline and the end of project. In this way, it is possible to obtain feedback and guarantee validity of the interventions.
4. *Strengthen mechanisms and entities in charge of collecting and making available data related to ICZM*. Socioeconomic, biophysical and ecological information systems in marine and coastal zones must be supported, consolidated and encouraged. The information provided by these systems is relevant for the identification, monitoring and modelling of changes in ecosystem services and stakeholders' wellbeing as a result of interventions related with coastal management, including natural infrastructure. The success of these systems depends on effective articulation between statistical agencies and other government units, as well as universities and other research institutions. Besides the strengthening of information systems, it is important to guarantee the adequate and opportune access to data.
5. *Train policy makers and practitioners on the usefulness of economic tools for the analysis of natural infrastructure projects*. Agencies and officers in charge of formulating and implementing projects, programs or plans related to the coastal and marine zones, including natural infrastructure, must be aware of the methods for economic analyses, both ex-ante (CBA) and ex-post (IE), as well as the economic valuation of ecosystem services, their scope, outcomes, advantages and disadvantages.

ANNEX 1: IMPACT EVALUATION APPROACHES

An impact evaluation that is both well designed and soundly applied should be able to answer the following question: What would have happened in the absence of the intervention, program or policy? That is, it would be able to recreate an unobservable counterfactual scenario, allowing to assess changes that can be attributed to the intervention (Ferraro & Pattanayak, 2006).

An impact evaluation must therefore demonstrate the causal relationship between the intervention and the outcome, discarding alternative or rival explanations of intervention outcomes (**Error! Reference source not found.**). This requires the knowledge of what would have been the outcome if the intervention had not taken place (Ferraro, 2009). In other words, an impact evaluation should be able to attribute the identified changes to the particular intervention that is being evaluated with high confidence and reliability (Woodhouse et al., 2016). This counterfactual analysis identifies the extent to which impacts are the result of the intervention a vis-à-vis confounding factors or biases (White, 2006). Typical confounding factors include contemporaneous elements correlated with the treatment intervention and outcomes and selection bias when intervened units are selected or self-selected, based on characteristics that affect the outcome (Ferraro, 2009).

Box 1. Key concepts in impact evaluation analysis

Counterfactual: The outcome that would have happened if there had been no conservation intervention (Ferraro & Pattanayak, 2006).

Selection bias: A bias in estimating a program's effect that occurs when the participant and control groups differ from each other because of factors that also affect the program's outcomes. Such differences often arise when program units (species, acres, people, etc.) volunteer to participate in the program or are purposively inducted into the program. As a result, outcome differences between the participant and control groups may arise from differences between the groups rather than the program itself (Ferraro & Pattanayak, 2006).

Internal validity*:* When a correlation implies a causal relationship rather than hidden biases (Ferraro, 2009; Ferraro and Hanauer, 2014). That is, when there is high accuracy in the attribution of impacts to the intervention being evaluated (Roe et al., 2013).

External validity: When an implied causal relationship can be generalized to other settings (people, places, time) (Ferraro & Hanauer, 2014; Ferraro, 2009).

Construct or measurement validity*:* The appropriateness of the variables or indicators that are used to measure impacts (Roe et al., 2013). Selected indicators or variables should actually be measuring the outcome that they are intended to measure (Ferraro, 2009).

Exogenous variable: A variable in a model or system that is causally independent of other variables in the model or system (Ferraro & Pattanayak, 2006).

Endogenous variable**:** A variable in a model or system that is causally dependent on other variables in the model or system (Ferraro & Pattanayak, 2006).

The counterfactual analysis is crucial for isolating the impacts of interventions and obtaining an unbiased estimate of an intervention's performance (Berry et al., 2012, cited in Curzon & Kontoleon, 2016). Given that the counterfactual cannot be observed, the key challenge of an impact evaluation is to find a valid and appropriate counterfactual scenario (Curzon & Kontoleon, 2016). However, as Ferraro (2009, p. 75) states, "this counterfactual world can be inferred only indirectly through evaluation designs that control for confounding factors." Confounding effects can be isolated using baselines, covariates and control groups. Baselines capture conditions before implementing the intervention, which control for conditions that may affect the program's effectiveness. Covariates are observable socioeconomic, biophysical or institutional factors that also influence the outcome. Control groups are individuals, communities, or areas that have not been treated but are similar (on average) to those that have experienced intervention (Ferraro & Pattanayak, 2006).

Common evaluations of conservation interventions are based on (i) the comparison of outcomes before and after the intervention and (ii) the comparison of outcomes with and without the intervention. However, both types of evaluations could potentially lead to biased estimates of impacts, because the former does not control for other time-varying factors (contemporaneous correlated factors), and the latter assumes that zones where the conservation intervention is implemented exhibit similar characteristics to those zones with no intervention. In general, however, conservation programs are exposed to selection bias. In addition, the comparison of outcomes with and without intervention assumes that there are no spillover effects from intervention on unexposed areas (Curzon & Kontoleon, 2016; Miteva et al., 2012).

Error! Reference source not found. presents the main impact evaluation approaches for conservation interventions, based on the classification of Margoluis et al. (2009).

Table 10. Types of evaluation designs (Margoluis et al., 2009)

	Quantitative			Qualitative
	Experimental	Quasi-experimental	Non-experimental	Qualitative sampling options
Definition	Random assignment of subjects to treated (experimental) and untreated (control) groups.	Similar to the experimental approach but without random assignment.	Draws inferences about the effect of a treatment on subjects, where assignment of subjects into a treated versus control group is outside the researcher's control.	Qualitative evaluation design options focus almost exclusively on the sampling framework and not on statistical power or how exposed and non-exposed cases are compared. Individual cases are weighted more heavily because the evaluator is not looking for population-based trends.
Advantages	Approximates counterfactual conditions, strong	Easier to set up than true experimental designs, fairly strong	Least expensive quantitative design,	Generally, less expensive than experimental and

	Quantitative Experimental	Quasi-experimental	Non-experimental	Qualitative Qualitative sampling options
	evidence for causality.	evidence for causality.	easier to implement.	quasi-experimental designs, provides rich anecdotal evidence.
Limitations	Expensive, often not practical, ethical issues, high expertise.	Moderately expensive to expensive.	Observes the state of the world without manipulating it, but has less power to detect causal relationships.	Analysis is more difficult, subjective interpretations.
Validity	Internal: high; random assignments, the gold standard for internal validity. External: low; artificial setting limits the ability to generalize to other settings.	Internal: moderate; inability to randomly assign controls, lack of control over variables. External: moderate; "natural experiments" allow some generalization.	Internal: low; no randomization, no controls. External: moderate; natural settings make generalizations stronger.	Internal: low; no randomization, no controls, researcher interpretation, interviewee perception, recall accuracy. External: low; if cases are carefully selected and analyzed over extended periods, can be moderate.
Examples	*Randomized control trial (RCT):* researcher randomly assigns items into control and experimental groups. Measurements taken before and after intervention.	*Matched controls:* intervention group matched with controls selected by researcher. *Regression-discontinuity:* pretest/posttest design in which participants are assigned to program or comparison groups on the basis of a cutoff score on a preprogram measure. *Statistically equated controls:* exposed and unexposed groups or items compared by means of statistical controls. *Generic controls:* exposed group or items compared with outcome measures available	*Pretest/posttest*: subjects measured before and after intervention. *Time series:* large aggregates taken from a large population and compared before and after intervention. *Cross-sectional studies for non-uniform programs*: subjects differentially exposed to intervention are compared to statistical controls.	*Stratified purposeful sampling:* stratifying samples within samples by selecting particular cases that vary according to a key dimension, thus facilitating comparison. *Extreme or deviant case sampling:* learning from highly unusual manifestations of issue of interest (e.g., outstanding successes and notable failures, top of the class or dropouts) *Theory-based or operational construct sampling:* sampling subjects on basis of their potential manifestation of a theoretical construct so as to elaborate and

	Quantitative			Qualitative
	Experimental	Quasi-experimental	Non-experimental	Qualitative sampling options
		on general population.		examine the construct.

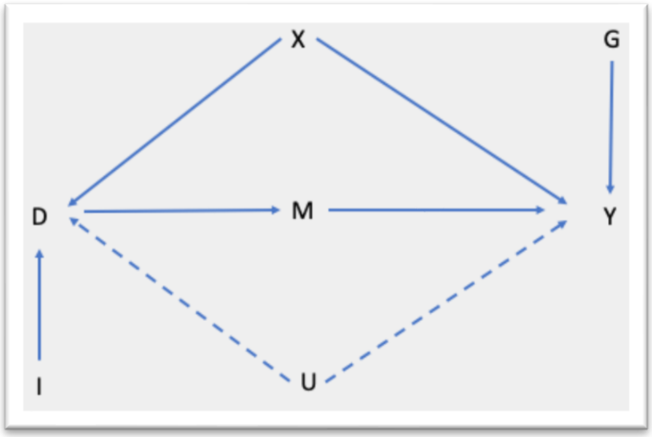
Quantitative evaluation approaches are useful to isolate effects of particular variables of interest. Margoluis et al. (2009) classify them into three categories: experimental, quasi-experimental and non-experimental. These three types of evaluation designs allow researchers to make inferences and statistical generalizations from the sample to population when units are adequately defined, and the samples are selected without bias (Margoluis et al., 2009).

The most rigorous evaluation methods rely on experimental and quasi-experimental designs as they credibly measure counterfactual outcomes, exhibiting high internal validity (Roe et al., 2013; Ferraro & Pattanayak, 2006; Ferraro, 2009). These two approaches ensure causal inference of the intervention.

Ferraro and Hanauer (2014) classify the designs aimed at causal inferences into four categories. This categorization is depicted in **Error! Reference source not found.**, where *D* represents the intervention, *Y* is the outcome (impact), and M corresponds to the mechanisms through which the intervention generates outcomes. However, a correlation between values *D* and *Y* may not necessarily reflect a causal relationship between them; in the same way, the absence of a correlation between *D* and *Y* would not necessarily reflect the absence of a causal relationship (Ferraro and Hanauer, 2014). Rival explanations (confounders) can be observable (*X*) or unobservable (*U*). *I* represents variables that, while affecting the intervention, would not affect the outcome and *G* variables that affect only the outcome.

A randomized control trial (RCT) is an experiment in which a particular intervention or treatment is assigned randomly among individuals, communities or zones (Ferraro & Pattanayak, 2006). In other words, in an RCT, units are randomly assigned into treatment (with intervention) and control (without intervention) groups. Data is collected in both the treatment and control groups, before and after the policy (Curzon & Kontoleon, 2016). In this way, potential rival or alternative explanations of outcomes (confounders) are balanced among treatment and control units, and any difference in outcomes between them can be unambiguously attributed to the intervention (Ferraro & Pattanayak, 2006).

Figure 12. Base design for a causal inference analysis and rival explanations (Ferraro & Hanauer, 2014)



If it is not possible to apply randomization, the second-best approach to carry out an impact evaluation of conservation projects is the use of a quasi-experimental approach that characterizes the counterfactual and isolates the causal effect of an intervention. Quasi-experimental approaches create, through statistical methods, comparison groups that are valid under determined assumptions about potential selection bias. This counterfactual baseline is used as a control (McKinnon et al., 2015; Khandker, Koowal & Samad, 2010)..

The authors' second category comprises the designs that depend on conditioning strategies, in which it is assumed that treatment assignments are affected only by observable variables for which one can collect data and control in the analysis (*X* in **Error! Reference source not found.**) (Ferraro, 2009). By controlling for the relationships between *D* and *Y*, they are eliminated as rival explanations. Pearl (2009) calls this the back-door criteria. This category includes matching and fixed-effect estimator methods.

Through matching, the intervention group is matched to observationally very similar units (controls) that are not affected by the intervention. In doing so, matching design mimics experimental designs but it does not assign units randomly to exposed and comparison groups (Margoluis et al., 2009). Matching assumes that similarities in the observed characteristics between treated and control groups translates into similarities in unobservable characteristics, correlated with the outcome and the intervention assignment, or that unobservable attributes are negligible sources of bias (Imbens and Wooldridge, 2009). Control units provide valid estimates of the counterfactual outcomes. Propensity score matching is one of the most used matching methods applied in research (Margoluis et al., 2009; Ferraro & Pattanayak, 2006).

The third category identified by Ferraro and Hanauer (2014) are designs that depend on naturally occurring sources of variation in *D* (**Error! Reference source not found.**) that are unrelated to potential outcomes. These designs assume that treatment assignment takes place on variables that are both observable and unobservable to the analyst (Ferraro, 2009). Following **Error! Reference**

source not found., an observable variable *I* is assumed to be unrelated to potential outcomes, except through its effect on *D*. This category includes methods such as instrumental variables and regression discontinuity designs.

The fourth category proposed by Ferraro and Hanuer (2014) corresponds to designs that depend on identifying mechanisms (*M* in **Error! Reference source not found.**) through which the effect of *D* on *Y* can be estimated via a two-step process that evaluates (i) the effect of *D* on *M* without bias, and (ii) the effect of *M* on *Y* without bias. Pearl (2009) refers to this approach as the front-door criterion.

Nevertheless, some authors state that experimental and quasi-experimental approaches are not so adept at dealing with complex and multidimensional issues such as wellbeing or poverty, leading to measurement validity problems. Another drawback related to experimental and quasi-experimental designs is that sometimes it can be difficult or even impossible to find appropriate control groups, and occasionally it is not desirable to work with control groups due to ethical reasons. These reasons support the recommendation to use mixed evaluation approaches (Roe et al., 2013).

Non-experimental designs are useful when it is not possible to identify, define or have access to an adequate comparison group or when evaluators do not have resources, time or the technical and statistical expertise to conduct experimental or quasi-experimental approaches. Non-experimental designs use a simple comparison between the performance before and after the intervention (pre-test and post-test, and time series designs) (McKinnon et al., 2015). As non-experimental designs imply neither randomization nor controls, they have less power to attribute causal relationships.

Qualitative evaluations generally focus on perceptions about the changes produced by the interventions and on changes in the attitudes and behavior of individuals and households affected by conservation projects. Although qualitative information does not allow for statistical analyses, as sampling is not population-based, and rarely involves the use of comparison groups (Margoluis et al., 2009), it does permit analysts a deep understanding of particular subjects (Ferraro & Hanauer, 2014) that play an important role for (i) eliminating rival explanations, (ii) understanding an evaluation's limitations (i.e., external validity), (iii) identifying causal hypotheses to test, and (iv) identifying factors that may serve as moderators. Results from qualitative analysis could also inform future quantitative evaluation surveys and help explain quantitative evaluation results (Margoluis et al., 2009).

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