The value of biodiversity and ecosystem services in the EU's Outermost Regions and Overseas Countries and Territories

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Summary

The EU overseas entities are well-known hotspots of terrestrial, freshwater and marine biodiversity. They support unique ecosystems, which are home to an estimated one third of the globally threatened species, including many endemic species. However, their ecosystems are threatened by invasive species, climate change, and habitat loss – the latter of which is often induced by human activities. In addition, biodiversity conservation in ORs/OCTs turns out to be challenging, due to the complex jurisdictional matters among the EU, Member States, and overseas territories levels.

The economic valuation of biodiversity and ecosystem services can be a tool to express the multiple societal benefits of intact ecosystems and their ecological functions. Estimated values can be used in awareness raising campaigns and in the design of policy instruments, such as payments for ecosystem services, at local and regional level. Within this report, a total of 39 valuation studies containing 110 individual value estimates have been identified and evaluated, in order to provide an overview about

the magnitude of the estimated economic value derived from (sub)tropical biodiversity in ORs and OCTs. It was found that the value of individual ecosystem services differs widely among the covered regions and ecosystems. The reasons can be found in different value perceptions of the local populations, and in the particularities (related to design and implementation) of the individual valuation studies.

The policy impacts of environmental valuation studies depend on a range of aspects, including the reliability of the valuation methods applied, the integration of stakeholder perceptions into the study design, the communication and dissemination efforts made by the researchers, and the receptivity of the targeted policy makers. This report presents case studies from Bonaire, St. Maarten and Belize, where valuation results were taken up by local policy makers, and discusses the factors which contributed to that uptake. The case studies reveal that, in order to be perceived as relevant and useful by policy makers, valuation studies should be designed and implemented in a participatory manner. Taking into account the stakeholders' perceptions turns out to be essential, even more if the objective is to design or adapt policy instruments based on the results of the valuation study.

While more effective protection of biodiversity in ORs and OCTs will depend on a range of factors, particularly available funding mechanisms, the economic valuation of biodiversity and ecosystem services can provide arguments for the integration of biodiversity aspects into other policy domains, such as agriculture and tourism. By informing public debates and local and regional policy making, it holds potential for contributing to smarter and more sustainable management of tropical and subtropical biodiversity in ORs and OCTs.

Publishable Summary

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Part I – Introduction

Within the framework of the EU FP7 project NetBiome-CSA, Task 3.2 gathered and reviewed the available literature on the socio-economic benefits of biodiversity and ecosystem services in the EU's Outermost Regions (ORs) and Overseas Countries and Territories (OCTs). A non-exhaustive review of peer-reviewed and grey literature was conducted to elicit the socio-economic value of biodiversity and ecosystem services. A number of good-practice case studies were identified, with the aim to illustrate the assessment of the socio-economic benefits of (sub)tropical biodiversity and ecosystem services in ORs/OCTs, as well as the uptake of the valuation results in local or regional policy-making. This report will lay the basis for a framework to build capacity on the socio-economic valuation of biodiversity and ecosystem services in ORs/OCTs. Specifically, the contents of the report will be used in NetBiome-CSA training sessions for local policy-makers and other stakeholders (Task 4.2), in order to inform them on the evaluation of the socio-economic benefits of biodiversity and ecosystem services to local communities in ORs/OCTs. Furthermore, the inventory of the identified valuation studies (*valuation database*) and the selected good-practice case studies will be integrated into the project Information System of the Biodiversity Management Toolbox (task 5.2). Table 1 below shows how the task description, as laid down in the Description of Work (DoW), has been implemented in this Report.

Table 1 - Overview of tasks covered in D3.1.

Task 3.2 outline in the DoW	How the tasks have been implemented
• *obtain insights on the methods applied to assess the economic value of biodiversity and ecosystem services*	A non-exhaustive review of peer-reviewed literature and technical reports was conducted in order to identify relevant valuation studies which have been carried out in ORs/OCTs and beyond. The identified valuation studies have been evaluated with regard to the valuation methods applied (Chapter IV.3). In addition, Part III.2 of this Report presents a general overview of existing market and non-market valuation methods.
• *provide an overview of the magnitude of the estimated economic value of (sub)tropical biodiversity*	A literature review identified 39 valuation studies in ORs/OCTs and beyond, which contain a total of 110 individual benefit estimates. The literature has been compiled in an Excel database. The results of the literature review are presented in Annex 1 and are summarised in Part IV.2 of this Report. The valuation database will be updated and expanded throughout the duration of the NetBiome-CSA project.
• *identify case studies where the results of the assessments have been used to inform local policy-making*	When information was available, the identified valuation studies were evaluated with regard to their impact on local and regional policy-making. This led to a selection of three good-practice case studies on the use of environmental valuations in local and regional policy-making. The case studies are presented in Part V of this Report.

The rationale behind this report is that economic valuations of biodiversity and ecosystem services can inform policy-making in the context of an (extended) cost-benefit analysis (CBA). CBA is still the standard tool for comparing the costs and benefits – and thus the welfare aspects – of a project or government decision. In a traditional CBA, costs and benefits are expressed in monetary terms. Thus, in order to integrate natural capital aspects into a traditional CBA, environmental values need to be expressed in monetary terms. This report therefore focuses on identifying and discussing the *monetary* values of biodiversity and ecosystem services in ORs/OCTs, thus relying on the concepts and methods from traditional environmental and resource economics.

However, the monetary valuation of natural resources and CBA in general are sometimes criticized. One debated aspect is the reliability/uncertainty of the valuation methods applied (Part III of this Report will discuss the pros and cons of the most common valuation methods). Another discussed aspect concerns the "concept of value" which is applied, and thus the relevance of the monetary estimates produced in the context of a valuation study. The latter aspect links to an ethical debate about whether natural resource management should (solely) be based on the idea of efficient resource allocation. Critics argue that issues concerning justice, fairness, morals and other ethical values need to be integrated into management decisions, and that a traditional CBA falls short of taking them into account. All these aspects are valid and need to be considered when evaluating the relevance of monetary valuations and their role in decision-making.

Ecological economists have acknowledged the shortcomings of traditional cost-benefit analysis and the monetary valuation of natural resources. They developed tools which can complement and replace CBA. For instance, multi-criteria-decision analysis (MCDA) has been promoted as a participatory tool to evaluate the impacts of management and policy decision. MCDA differs from a traditional CBA in the respect that not all evaluation criteria need to be monetised, but some can also be expressed in quantitative or qualitative terms. Within an MCDA, decision-makers and stakeholders can also rank the evaluation criteria, so that justice or distributional issues (usually expressed in qualitative terms) would receive a higher weight in the evaluation than costs and benefits expressed in monetary terms. This example shows how monetary valuations can play a role, even outside the neoclassical economic approaches such as CBA.

Following this introduction, the report is structured as follows: Part II discusses the rationale behind the economic valuation of (sub)tropical biodiversity and ecosystem services by describing the relevant policy context in ORs/OCTs, by introducing the concept of ecosystem services and by bringing forward arguments for the economic valuation of natural resources. Part III explains existing concepts and methods for the economic valuation of biodiversity and ecosystem services, focussing on the Total Economic Value (TEV) framework and available market and non-market valuation methods. Part IV then provides evidence on the value of biodiversity and ecosystem services in ORs/OCTs, by summarising and analysing the results of a non-exhaustive literature review. Part V presents three selected case studies, which serve as best-practice examples for the uptake of valuation results by local and/or regional policy-makers. Conclusions are then drawn in Part VI.

Part II – Rationale behind the economic valuation of biodiversity and ecosystem services

1. Governance of natural resources in ORs/OCTs

In addition to its 28 Member States, the European Union includes 34 overseas territories which are associated with the EU based on the provisions of Part IV of the Treaty on the Functioning of the EU and the items laid down in the Overseas Association Decision of 27 November 2001. Consisting of eight Outermost Regions (ORs) and 26 Overseas Countries and Territories (OCTs), these territories have constitutional ties with Denmark, France, the Netherlands, Portugal, Spain or the United Kingdom and receive support from the EU for economic and social development. While the ORs are treated like EU Member States, the OCTs depend constitutionally on one of the aforementioned EU Member States but are not directly subject to Community law. Substantial differences exist amongst the OCTs regarding their degree of autonomy but they are all sovereign countries, parliamentary democracies and islands with small populations.

The EU overseas entities support unique ecosystems which are home to an estimated one-third of the globally threatened species (Kettunen and Bezerra, 2008), including many endemic species. Four of the five French biodiversity hotspots, for example, are located in overseas territories along with an estimated 90% of the biodiversity found within the UK and its territories combined (Foreign & Commonwealth Office, 2012). This globally significant biodiversity is essential for the continued provisioning of the ecosystem goods and services supporting the local populations as well as for both the local and EU economies. While ecotourism and fisheries activities illustrate the critical role of biodiversity in supporting sustainable development in the regions, the EU appreciates the importance of upholding access to maintained fisheries grounds, marine genetic resources, mineral exploration and a foothold in the high seas in three oceans (IUCN, 2012).

At EU level, recognition of the need to conserve biodiversity in EU overseas territories as part of European biodiversity commitments has led to the integration of these territories in various legislative items. The Commission Communication COM(2009)623 on “Elements for a new partnership between the EU and the overseas countries and territories (OCTs)” lists five axes of cooperation for an OCT/EU partnership, including to cooperate with OCTs on environmental issues and disaster risk reduction. OCTs and ORs have also been included in the Council of the EU conclusions of 19 December 2011 on the Integrated Maritime Policy (IUCN, 2012) and in the EU Biodiversity Strategy to 2020, emphasizing the potential of the BEST initiative to promote biodiversity conservation and sustainable use.

Box 1 – The EU Biodiversity Strategy: six targets and twenty actions.

1. The full implementation of the EU nature legislation;
    - A1: **Complete** the **establishment for the Natura 2000 Network** and ensure **good management**.
    - A2: Ensure adequate **financing of Natura 2000**.
    - A3: Increase stakeholder **awareness** and involvement and improve enforcement.
    - A4: Improve and streamline **monitoring and reporting.**
2. Better protection and **restoration of ecosystems and the services** they provide, and
    - A5: **Improve Knowledge** of ecosystem and their services.
    - A6: **Set priorities** to restore and promote the use of green infrastructure.
    - A7: Ensure **no net loss of biodiversity** and ecosystem services.
3. More sustainable **agriculture and forestry**;
    - A8: Enhance **direct payments** for environmental public goods in the EU Common Agriculture Policy.
    - A9: Better target **rural development** to biodiversity conservation.
    - A10: Conserve Europe's agricultural **genetic diversity.**
    - A11: Encourage forest holders **to protect and enhance forest biodiversity.**
    - A12: Integrate biodiversity measures in **forest management plans.**
4. Better management of **EU fish stocks** and more sustainable fisheries;
    - A 13**: Improve the management** of fished stocks.
    - A 14: **Eliminate adverse impacts on fish stocks**, species habitats and ecosystems.
5. **Combat Invasive Alien Species.**
    - A 15: Strengthen the **EU Plant and Animal Health Regimes.**
    - A 16: Establish a **dedicated legislative instrument** on Invasive Alien Species.
6. Contribute to averting **global biodiversity loss.**
    - A 17: **Reduce indirect drivers** of biodiversity loss.
    - A 18**: Mobilise additional resources** for global biodiversity conservation.
    - A 19: 'Biodiversity-proofing' **EU development cooperations**.
    - A 20: Regulate access to **genetic resources** and the fair and equitable sharing of benefits arising from their use

Furthermore, Commission Communication COM(2012)287 on "The outermost regions of the European Union: towards a partnership for smart, sustainable and inclusive growth" takes particular note of the need to support biodiversity and ecosystem services and identifies paths for sustainability across an array of traditional sectors (e.g. tourism, agriculture and rural development, fisheries, etc). At national level, the UK has developed an Overseas Territories Biodiversity Strategy in 2009 and published a White Paper on the Overseas Territories in 2012; the later sets out its overall approach to OCTs and outlines its role in supporting them to meet the requirements of the Convention on International Trade in Endangered Species, the CBD and the Convention on Migratory Species.

The application of EU policy to the Overseas Countries and Territories is not, however, without criticism. In January 2012, while the 10th OCT-EU Forum recognized the environment, trade and regional integration as key areas for future cooperation, the current Chairmanship prioritized green growth in education, innovation and research over the development of biodiversity strategies. Furthermore, while the ORs implement EU policies such as Cohesion, Birds and Habitats Directives and Common Agriculture Policy and are eligible for EU Structural and Cohesion, agricultural and LIFE+ funding, the OCTs lack a focused framework for conservation guidance and are not eligible for LIFE+ funding. OCTs are instead primarily funded through the European Development Fund, which often favours initiatives targeting economic growth and development over biodiversity conservation (Kettunen and Bezerra, 2008).

Although the importance of biodiversity for the territories is acknowledged, conservation targets often remain unmet. Contributing factors in addition to those above include the nations remoteness (adding to the cost of environmental projects), vulnerability to economic shocks, limited access to technical expertise, difficulties to build and maintain infrastructure or sustainable energy supply. Biodiversity is additionally threatened by invasive non-native species, climate change and habitat loss (Defra, 2009). Marine conservation in particular is subject to complex jurisdictional matters among the EU, Member State and overseas territories levels (IUCN, 2012). In general, one can observe that the degree of environmental protection and governance varies among the ORs/OCTs. For instance, an assessment of environmental protection frameworks in the UK overseas territories finds that “there are areas of best practice in many Territories, which can act as a beacon for others to emulate, but that many OCTs still have significant gaps in their environmental governance which urgently need to be addressed” (FIELD and RSPB, 2013). The current Nature Policy Plan for the Caribbean Netherlands acknowledges that “limitations in terms of capacity, funding and political support turned out to be the chief challenges” for the implementation of the environmental policy objectives that had previously been defined.

The fact that certain conservation targets remain unmet becomes apparent in discussions with local and regional stakeholders. A stakeholder consultation carried out under Task 3.1 of NetBiome-CSA identified five broad social-ecological challenges in ORs/OCTs for which actions needs to be taken:

- **Implementing species and habitat conservation and management**, including an improved understanding of the drivers affecting biodiversity and the definition of priorities for local biodiversity preservation, supported by practical guidelines for policy-makers;
- **Defining a large-scale and holistic approach for spatial planning**, including smarter and coordinated territorial policies and a sound dialogue between researchers and policy-makers for informed decision-making;
- **Avoiding and mitigating anthropogenic impacts on biodiversity**, including pressures caused by urban expansion, mining and other industrial activities, and pollution trough the use of chemicals in agriculture;
- **Designing smart and sustainable agricultures practices**, including an assessment of the sector’s impact on local ecosystems and, at the same time, considering the potential disappearance of agricultural land and its negative impacts on food security;

- **Tackling bio-security and invasive pests,** including the recognition of invasive alien species as the main direct driver of biodiversity loss and a potential cause for socio-economic losses.

2. Towards a new paradigm: the concept of ecosystem services

The identification of environmental objectives is based on a broad range of aspects and a variety of (vested) interests which influence the policy arena. One aspect in the process of environmental policy formulation is the measurement of a) the value of environmental resources, and b) the value of the marginal changes of environmental quality (Kahn, 2005). The basic idea behind valuing natural resources is that functioning ecosystems provide welfare benefits to human society and that, in turn, the degradation or loss of ecosystems leads to welfare losses. In an influential publication, Daily (1997) described the societal dependence on natural ecosystems and the services they provide. In the same year, Costanza et al. (1997) published a controversial paper in *Nature* on the value of the world's ecosystem services and natural capital. For the entire biosphere, the authors estimated the value to be in the range of US$ 16-54 trillion per year.

Until the beginning of the new Millennium, the debate about the services which nature provides and their economic value remained mainly an academic debate outside of most policy debates; this changed with the Millennium Ecosystem Assessment (MA). The MA (2005) was a global study initiated by the United Nations which aimed at providing an overview of the status of 24 key ecosystem services at global level and at assessing the consequences of ecosystem change for human well-being. Between 2001 and 2005, a number of assessment reports have been published, which showed that the world ecosystems were in a process of degradation. It was concluded that out of 24 ecosystem services, "only four have shown improvement over the last 50 years, fifteen are in serious decline, and five are in a stable state overall but under threat in some parts of the world" (MA, 2005). The innovative MA approach was that people was viewed as integral parts of ecosystems. The synthesis report states that "a dynamic interaction exists between them and other parts of ecosystems, with the changing human condition driving, both directly and indirectly, changes in ecosystems and thereby causing changes in human well-being" (MA, 2005). The analysis of the effects of ecological change on human well-being within the MA centres on the concept of ecosystem services. Figure 1 depicts the linkages between categories of ecosystem services and components of human well-being.

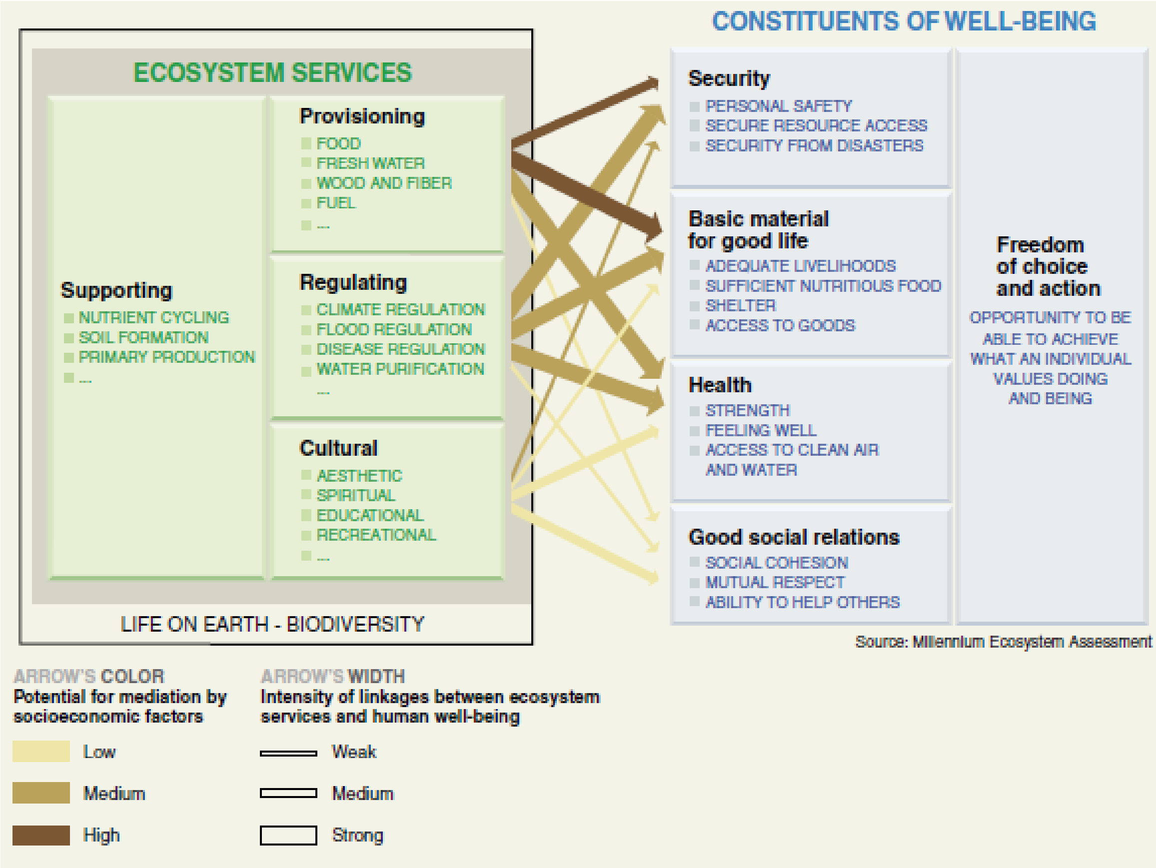


Figure 1 - The links between ecosystem services and the constituents of human well-being according to MA (2005).

The MA defines ecosystem services as the benefits that nature provides to human society. These include provisioning services, regulating services, cultural services, and supporting services. They contribute to our social and economic well-being – consisting of security, the basic materials for a viable livelihood, freedom and choice, good health, and good social-cultural relations – by providing us with food, natural fibres, steady supply of clean water, regulation of pests and diseases, medicinal substances, recreation, and protection from natural hazards (MA, 2005). Healthy ecosystems, thus, provide a broad range of socio-economic benefits to human society.

Ecosystem degradation and declining ecosystem service provision, on the other hand, pose an economic risk to society. Ecological change, leading to reduced ecosystem quality, may cause changes in the quantity and quality of ecosystem services provided. These changes may affect ecosystem functioning, human health, and economic activities that are dependent on the provision of ecosystem services. A reduced provision of ecosystem services as a result of ecological change thus results in socio-economic costs to be borne by human society. By estimating changes in production, costs of replacement, hedonic prices and by applying contingent valuation or an ecosystem services approach (cf. Part III.2), the scope of these costs can be determined ex post. Haines-Young & Potschin (2010) came up with what has been termed the “ecosystem services cascade” (see Figure 2). It describes the relationship between biodiversity, ecosystem functions and human well-being. This concept differentiates between ecosystem services, the benefits they provide to human society and the values which are attached to it.

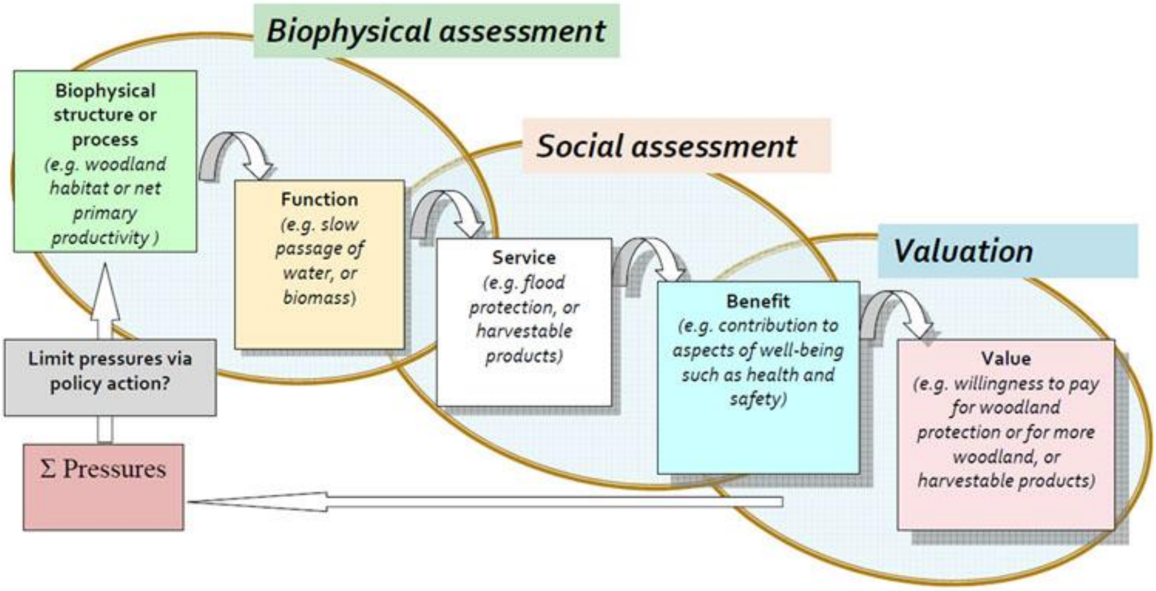


Figure 2 - The relationship between biodiversity, ecosystem function and human well-being according to Haines-Young & Potschin (2010).

This rationale has been taken up by another global effort which aims at capturing the value of the world's ecosystems: the study on The Economics of Ecosystems and Biodiversity (TEEB). The objective of TEEB is to provide an overview of existing approaches for the economic valuation of biodiversity and ecosystem services. The goal is "to highlight the growing cost of biodiversity loss and ecosystem degradation and to draw together expertise from the fields of science, economics and policy to enable practical actions" (TEEB, 2010). The blueprint for TEEB was the Stern Review on the Economics of Climate Change, which was published in 2006 and highlighted the effects of global warming on the world economy (Stern, 2006). The Stern Review received broad media coverage and contributed to an increased awareness of the negative effects of climate change among the wider public. A year later, TEEB was designed to reach the same goals in regard to another pressing problem – the ongoing loss of global biodiversity.

Since the launch of the initiative, five TEEB Study Reports have been published, each targeting a different audience:

- **TEEB Ecological and Economic Foundations** explains the fundamental concepts and state-of-the-art methodologies for economic valuation of biodiversity and ecosystem services;
- **TEEB in National and International Policy Making** provides analysis and guidance on how to value and internalize biodiversity and ecosystem values in policy decisions;

- **TEEB in Local and Regional Policy Management** provides analysis and guidance for mainstreaming biodiversity and ecosystem values at regional and local levels, copiously illustrated with case study examples;
- **TEEB in Business and Enterprise** provides analysis and guidance on how business and enterprise can identify and manage their biodiversity and ecosystem risks and opportunities.
- **Mainstreaming the Economics of Nature** provides a synthesis of the approach, conclusions and recommendations of TEEB

In addition, a number of independent TEEB studies have been or are currently being carried out, focusing on the assessment of natural capital in individual countries, sectors, or biomes. Worth mentioning in this context is the national TEEB programme in the Netherlands, which has been initiated in 2011 by the Dutch government. In the framework this programme, a number of valuation studies have been carried out or are in preparation, including two three which focus on the Dutch overseas territories: "TEEB for the Caribbean Netherlands – Bonaire", "TEEB for the Caribbean Netherlands – Saba" and "TEEB for the Caribbean Netherlands – Saint Eustatius". The project "What's Bonaire's Nature Worth?" (2011-2012) investigated how the local ecosystems contribute to Bonaire's economy and human well-being. The associated valuation studies estimated the value of more than ten different ecosystems services in monetary terms, including local cultural and recreational values, international tourism values, fisheries values, non-use values, coastal protection values, and the functional value of the island ecosystem services. An overview of these values is provided in Part VI.2 and Annex 1 of this report.

3. Environmental valuations in the policy process

Valuing both the environment and the changes in the level of environmental quality are of central importance to environmental policy formulation, as it puts the costs of obtaining certain environmental goals into perspective. Navrud and Pruckner (1997) identify five different uses of environmental valuations in decision-making: cost-benefit analysis (CBA) for both project evaluation and regulatory review, natural resource damage assessment, environmental costing (i.e. externalities), and environmental accounting. Focusing on EU water policy, Thaler et al. (2013) highlight that international, national, and regional environmental policies and strategies explicitly acknowledge the importance of environmental costs and benefits, and the need to integrate them into the policy-making process. The same is true for other policy domains.

Outlining the benefits of ecosystems and their services can provide economic arguments for the preservation, sustainable management and restoration of these ecosystems. So-called **ecosystem-based**

approaches, which target an integrated management of land, water and living resources that promotes conservation and sustainable use in an equitable way[1] and which aspires to maintain the natural structure and functioning of ecosystems. These approaches address the crucial links between climate change, biodiversity, ecosystem services and sustainable resource management, and thus have the potential to simultaneously contribute to several policy aims and local needs. Ecosystem-based approaches also maintain existing carbon stocks, regulate water flow and storage, maintain and increase resilience, reduce vulnerability of ecosystems and people, help to adapt to climate change impacts, improve biodiversity conservation and livelihood opportunities and provide health and recreational benefits (Perez et al., 2010; Naumann et al., 2011)

The value of the different social and environmental benefits that can be obtained by implementing ecosystem-based approaches is of particular importance, not only to justify the spending e.g. for the preservation and sustainable management of natural ecosystems, but also to select such approaches instead of traditional engineered approaches, for instance in the area of climate change adaptation. There is evidence that indicates that the majority of projects using ecosystem-based approaches can be considered as beneficial, from an economic point of view, if one takes account of their long-term welfare benefits. In this respect, ecosystem-based approaches may be more cost-effective than traditional engineering approaches (Jones et al., 2012; Naumann et al., 2011; Doswald and Osti, 2011). The following box provides examples highlighting the cost-effectiveness of ecosystem-based approaches.

Box 2 – Comparison of costs and benefits of ecosystem-based approaches and hard engineering options.

Maldives: Disaster risk reduction through coral reefs

Coral reefs and other coastal ecosystems in the Maldives provide critical protection to coastal communities from storms and erosion, substantially reducing storm-related damages and saving lives. Tropical storm events are likely to increase in terms of frequency and consequences with the increasing impact of climate change. These developments reveal the need to protect the reefs and prevent their on-going degradation (resulting e.g. from overfishing or coral mining) through the establishment of marine protected areas. Such actions would cost ca. US$34 million in start-up and ca. US$47 million/year to maintain their critical protection service. Apart from this reducing the risk of natural disasters, this action could also generate ca. US$10 billion per year in co-benefits through tourism and sustainable fisheries. An irreversible degradation and therewith loss of the coral reefs would require to build hard infrastructure such as seawalls, breakwaters and other forms of coastal protection. Costs for such infrastructure have been estimated at US$1.6 billion–2.7 billion.

(Source: Jones et al., 2012; Moberg and Rönnbäck, 2003; Emerton et al., 2009; Mohammed, 2007)

Turks and Caicos Islands: Disaster risk reduction through coral reefs

The protection against erosion and wave damage provided by natural buffers (coral reefs) in the Turks and Caicos Islands has been estimated at US$16.9 million/year. Constructing dykes and levees as hard engineering solution would cost instead US$223 million, which corresponds to 8 % of the gross domestic product.

(Source: Jones et al., 2012; Conservation International, 2008; Henry, 1993; Batker, 2005)

[1] CBD COP5, Decision V/6 (see http://www.cbd.int/ecosystem/)

Beyond their integration into cost-benefit analyses, environmental valuations can play an important role in the design of policy instruments, particularly **market-based instruments** (MBI). Policy interest in MBI for environmental policy has been growing since the 1980s (Eftec et al., 2010).

Box 3 – Definition of market-based instruments.

The EEA defines market-based instruments in the following way: *"Market-based instruments seek to address the market failure of 'environmental externalities' either by incorporating the external cost of production or consumption activities through taxes or charges on processes or products, or by creating property rights and facilitating the establishment of a proxy market for the use of environmental services."* Market failure, in the **case of biodiversity**, originates from the nature of the goods and services provided by biodiversity.

The main problems are:

(i) biodiversity related goods and services are often public goods,

(ii) the use or conservation of biodiversity is associated with external effects, and

(iii) an asymmetry of information between those paying for conservation measures and those carrying them out sometimes exists.

(Source: EEA (based on UNEP's definition): http://glossary.eea.europa.eu/terminology/concept_html?term=market-based%20instrument; Bräuer et al., 2006)

MBI offer policy-makers new ways to reach conservation objectives in a more cost-effective way, as they use market forces and signals to pass on incentives and address market failures. Moreover, MBI can complement traditional regulatory measures, for example by generating revenue to fund public conservation management (Bräuer et al., 2006). They can be categorised as either price or quantity based instruments. In addition, instruments aimed at improving the operation of existing markets – so-called 'market-friction' instruments – are in some cases included as market instruments (Coggan and Whitten, 2005). The categories of instruments are illustrated in Figure 3 below.

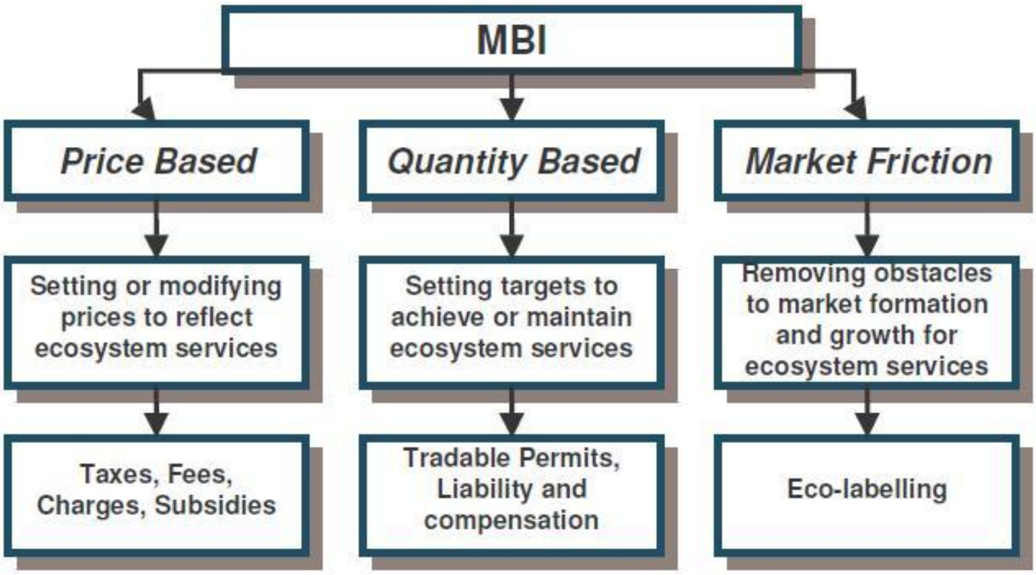


Figure 3 - Functional mechanisms of MBI (based on Coggan and Whitten, 2005).

Price based instruments include direct positive incentives in the form of subsidies/support or tax breaks and negative incentives in the form of taxes, charges and fees. These incentives can be attached to environmentally beneficial or damaging activities. It is expected (but cannot be guaranteed) that individuals will respond by adopting the behaviour which costs them least and that the use of resources will be improved. **Quantity based instruments,** also known as indirect incentives, create a market by distributing permits to carry out an activity associated with specified resource uses or environmental damage. Examples include the trade for rights to log woodland or emit a certain volume of pollutant. The total amount of damage should be controlled through these instruments. These types of MBI may be more likely to cause long-term behavioural changes but also need the greatest amount of administration. **Market frictions,** such as food certification and labelling schemes, aims to change the manner in which the current market works by reducing transaction costs and providing more information. Consumers valuing biodiversity conservation will pay more for a sustainably produced food product and thus allow producers to gain higher revenues and compensate for the higher cost of production (Bräuer et al., 2006).

In regard to the preservation of biodiversity, all standard types of MBI – taxes/charges/fees, subsidies and tradable permits – are in use, mainly for habitat and ecosystem conservation but also for the protection of specific species. The European Commission Green Paper on market-based instruments for environment and related policy purposes (COM(2007) 140 final) states that MBI can be efficient instruments to encourage landowners and land users to maintain natural ecosystems (e.g. wetlands, forest or mangroves); or to compensate for the unavoidable damage to biodiversity and ecosystems caused by infrastructure development projects, by creating similar habitats in nearby or in other areas to ensure no net loss of biodiversity. The following table provides some examples of MBI used in biodiversity protection.

Table 2 - Examples for market-based instruments in biodiversity protection.

Category	Type if instrument	Examples
Price based	Taxes, fees and charges	Charge for premature harvesting of forests, charge for wildlife use, tree cutting charge, hunting and fishing permits, fees to visit national parks, pesticide and fertiliser taxes
	Subsidies/ support, grants and funds	Subsidy for ecological livestock production, subsidy for wetland management, countryside stewardship scheme, subsidies for afforestation, maintenance of forests, natural resources management, protected areas, purchase of ecological areas
Quantity based	Tradable permits	Tradable fishing quotas, tradable hunting quotas, wetland banking, green offsets for sustainable regional development, tradable logging permits
	Liability and compensation	Compensation according to the Habitats Directive articles 6, 12 and 16: e.g. creating new habitats (to compensate for the loss of habitats through development projects), Compensatory remediation according to the Liability Directive (2004/35/EC) to compensate for a temporary loss of natural resources and environmental damages
Market friction	Eco-labelling	Forest certification, eco-labelling local foods, agricultural eco-labelling

In order to develop MBI, an economic foundation and valuation is needed. This is particularly relevant for the development of compensatory remediation, mitigation and compensation measures and payments for ecosystem services. In the case of environmental damage, the Environmental Liability Directive (2004/35/EC) aims to compensate for a temporary loss of natural resources pending their recovery. When defining remedial measures, the directive advocates the use of a resource-to-resource or service-to-service equivalence approach. Similarly, the Environmental Impact Assessments (85/337/EEC) and the Strategic Environmental Assessment (2001/42/EC) propose specific mitigation and compensation measures if damage to species and habitats was caused, resulting e.g. in the construction of highways causing irreversible damage to species and habitats. For the definition of adequate measures, the value of the respective habitat and its species has to be calculated.

One of the most prominent and strongest examples for the integration of the economic value of biodiversity and habitats into market-based instruments are the so-called payments for ecosystem services (PES). PES are incentives offered to farmers, foresters or landowners in exchange for managing their land to provide specific ecosystem services (e.g. water regulation and provision of drinking water, control soil erosion, carbon sequestration through sustainable forest management). They have been defined as “a transparent system for the additional provision of environmental services through conditional payments to voluntary providers” (Taconi, 2012). Therefore, PES promote the conservation and sustainable management of natural resources in the market place. Payments are being calculated, for example, based on the monetary value of ecosystem services provided by the land users, and also in some cases according to the income foregone due to intensification of land use.

Part III – Concepts and methods for the economic valuation of biodiversity and ecosystem services

1. The Total Economic Value (TEV) framework

The typology of benefits from biodiversity and ecosystem services provided by the concept of Total Economic Value (TEV), which consists of two main categories: use value and non-use value (e.g., Pearce and Turner, 1990; Hanley and Spash, 1993). Figure 4 provides an overview of the value types which exist within the TEV approach.

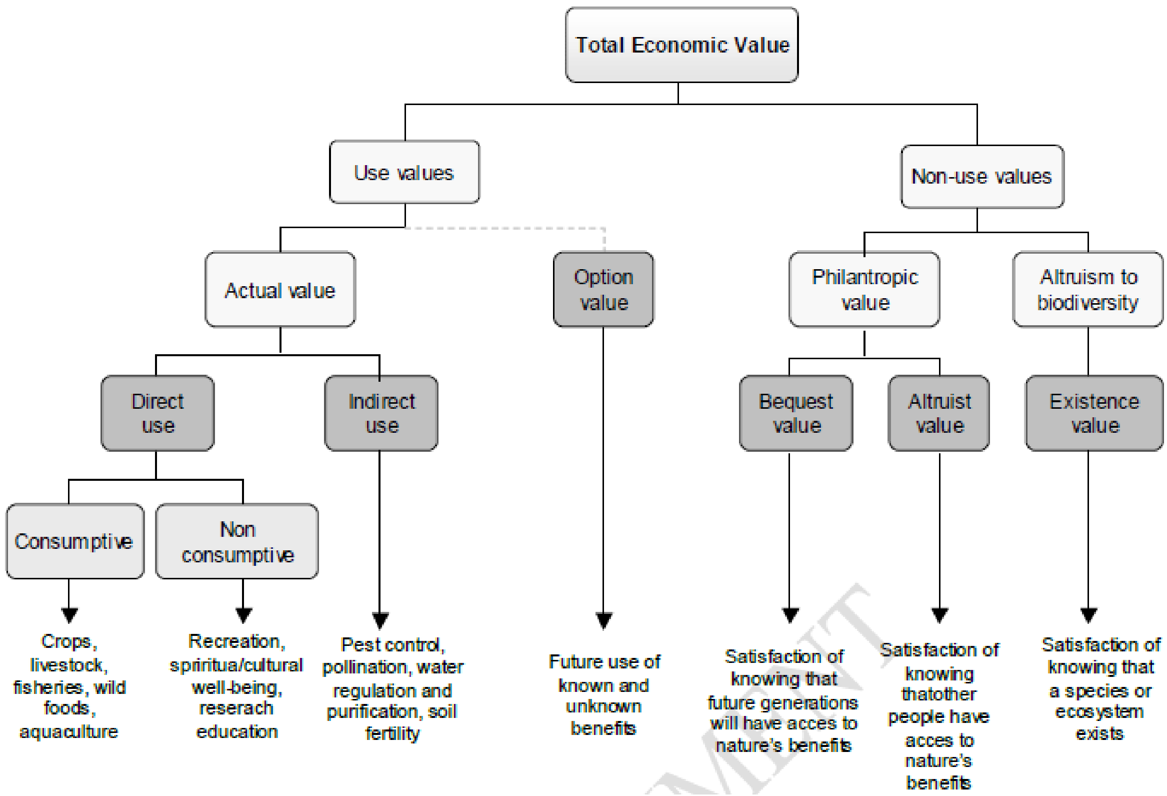


Figure 4 - Value types within the TEV approach, according to Pascual and Muradian (2010).

Fundamentally, TEV recognizes the distinction between the value that individuals derive from using the environmental resources, i.e. use values, and the value that individuals derive from the environmental resource even if they themselves do not use it, i.e. non-use values (Birol et al., 2006). Use values can be direct use values, such as when an individual makes actual use of the environmental asset improved; for example, fishing where it was not possible to catch a fish before the improvements in water quality took place; indirect use values, such as the benefits derived from ecosystem functions gained, for example, where recreational activities are created or enhanced due to water quality improvements, individuals can benefit in the form of increased recreational opportunities; and finally option values which measure individuals' preferences with regard to enjoying the improved resource in the future (make use of it). Non-use values are often called existence values, defined as the economic value placed by people for improvements to the quality of a river due to some moral and/or altruistic reasons, or for the mere pleasure of knowing that the water in the river has been enhanced.

The different value categories can be linked to the ecosystem services classification, as presented by the Millennium Ecosystem Assessment (MA, 2004). All categories of ecosystem services provide option values because each service may be used at a later moment in time, although currently undetermined. Direct use values can be assigned to the category of provisioning services, such as the supply of freshwater and fish. Indirect use values are typically assigned to the category of regulating services because these are not enjoyed directly, but affect individuals' welfare. Non-use values are typically assigned to the category of cultural services.

Table 3 - Matching the MA ecosystem service typology to categories of TEV.

MA service	Direct use	Indirect use	Option value	Non-use value
provisioning	x		x	
regulating		X	x	
cultural	x		x	x
supporting	No final ecosystem service, hence valued through the other categories			

Willingness To Pay (WTP) and Willingness To Accept (WTA) are the two standard measures of economic value. WTP is the appropriate measure in the situation where an agent wants to acquire a good. Minimum willingness to accept compensation is the appropriate measure in a situation where an agent is being asked to voluntarily give up a good. Whether WTP or WTA is the correct measure depends on the property right to the good. If the consumer does not currently have the environmental good and does not have a legal entitlement to it, the correct property right is WTP. If the consumer has a legal entitlement to it and is being asked to give up that entitlement, the correct property right is WTA.

For marketed goods, theoretically the difference between the two measures should generally be small and unimportant, as long as income effects and transaction costs are not large. For non-marketed goods, this may not be the case, as the difference between WTP and WTA is also dependent upon the substitutability of the non-marketed good for goods available on the market. Furthermore, there is huge theoretical and empirical research comparing the values derived by either WTP or WTA. WTA has been found to provide higher values than WTP, sometimes of up to 2 and 5 times higher depending on the product and method used (Melichar and Ščasný, 2004). This difference mainly shows that individuals' value is gaining something very differently from losing that same thing (Spash and Vatn, 2007).

2. Methods for the economic valuation of biodiversity and ecosystem services

This section will introduce existing methods for the economic valuation of biodiversity and ecosystem services. The valuation method to apply depends on the specific ecosystem service to be assessed. Some valuation methods are more appropriate than others for valuing particular ecosystem services, and for the elicitation of specific value components (Pascual and Muradian, 2010). Table 3 shows the links between specific methods and value components. It classifies methods according to their capacity of using conventional, surrogate or hypothetical markets for the estimation of use and non-use values.

Table 4 - Relationship between valuation methods and value types according to Pascual and Muradian (2010).

Approach		Method	Value
Market valuation	Price-based	Market prices	Direct and indirect use
	Cost-based	Avoided cost	Direct and indirect use
		Replacement cost	Direct and indirect use
		Mitigation / Restoration cost	Direct and indirect use
	Production -based	Production function approach	Indirect use
		Factor Income	Indirect use
Revealed preference		Travel cost method	Direct (indirect) use
		Hedonic pricing	Direct and indirect use
Stated preference		Contingent Valuation	Use and non-use
		Choice modelling/ Conjoint Analysis	Use and non-use
		Contingent ranking	Use and non-use
		Deliberative group valuation	Use and non-use

2.1 Market valuation methods

Market valuation methods use information from conventional markets, are based on physical linkages, and derive value indirectly using various statistical sources and the dose-response function (e.g. availability of water leads to an increase in fish populations). Some examples of market valuation methods are:

- ***Change in the Output or Input of a Marketed Good***

This method can be used when an environmental function affects the production and/or cost function of a certain good. In the productivity change method (PCM), change in an environmental attribute leads to changes in the output of the marketed good. For instance, a decrease in water quality due to pollution can have an adverse impact on fish stock in terms of quantity and/or quality. Damage due to water pollution can be estimated as a loss of fish production or involved incremental costs spent in order to mitigate the adverse effect of water pollution on the fish stock. A special case of PCM is the substitute cost method, in which the money saved using environmental goods (e.g. forage to feed livestock) instead of a priced input (sorghum) is a measure of the benefits of a certain environmental good or service.

- ***The Production Loss Method: Human Capital Approach (HCA)***

HCA can be regarded as a special case of the productivity change method applied to a very special good that is the workforce of a human being. This method is based on a macroeconomic vision of the role of the individual as an agent contributing to the activity of the economic system. The mortality effect is then valued through his/her productive contribution. The value of preventing a fatality at a given time is equal to the future productive loss evaluated as the discounted sum of the earnings that the individual would have otherwise earned. However, a problem related to this approach is that this method is inconsistent with principles of welfare economics, as it is not taking into account agents' preferences. Due to considering only the productive aspect of the individual, this method underestimates the value of life compared with estimates derived from WTP approaches (Melichar and Ščasný, 2004).

- ***The Loss of Consumption Method***

Another method trying to derive a monetary value for statistical life or mortality effect is the loss of consumption method. This approach, again, is based on a macroeconomic vision of any individual as a consumer and in case of premature death, the loss of consumption possibilities is estimated. The estimation of value of a statistical life or related mortality effects is mostly based on households final consumption (OECD, 2002).

- ***Cost-of-illness (COI)***

The cost-of-illness method is applied in monetary valuation of morbidity effects within health impact assessment. COI measures the pure economic benefit associated with a change in health status that consists of i) treatment costs, and ii) loss of productivity.

- ***Replacement Costs***

The method focuses on costs spent in order to abate, restore or replace a previously damaged marketed or non-marketed good due to degradation of a certain environmental quality. One example of the method can be found in Pretty et al. (2003). This study assessed the extent of the total external costs associated with agricultural practices in the UK in terms of changes in water quality: benefits of water quality improvements were estimated by calculating the total costs for water companies of removing agricultural diffuse pollutants.

2.2 Non-market valuation methods

There are two very well differentiated groups of non-market valuation methods: those based on revealed or stated preferences. Revealed preference techniques are based on the observation of individual choices in existing markets that are related to the ecosystem service that is subject of valuation. Stated preference techniques, on the other hand, simulate a market and demand for ecosystem services by means of surveys on hypothetical changes in environmental quality (Pascual and Muradian, 2010). Revealed preference methods can be divided into the Hedonic Pricing Method (HPM) and the so-called household production function approach (Kolstad, 2002). This approach consists of the Travel Cost Method (TCM) and the Averting Behaviour Method (ABM). Stated preference methods include the Contingent Valuation Method (CVM) and Conjoint Analyses (CA).

- ***Hedonic Pricing Method (HPM)***

The basic assumption of HPM is that the market value of a good is affected by many attributes, including environmental quality. If one is able to isolate the particular effects of specific environmental attributes on the price, it is possible to derive an implicit or surrogate price of the attribute. The method consists of two steps: first, hedonic price function is derived from real observations (the relation between a real market price and the quality of the environmental attribute is estimated) and the implicit price function is derived from the hedonic price function given by the first derivative of the house price function with respect to the environmental attribute; second, based on the estimated implicit price function, the inverse demand function is derived (in that implicit price is regressed on various observed socioeconomic and environmental variables); finally, consumer surplus can be calculated from the inverse demand function.

- ***Travel Costs Method (TCM)***

This method is commonly applied to valuing site-specific goods related to provision of a certain environmental resource. TCM is mostly applied to valuing the recreational value of forest, countryside, or any other landscape. TCM can, however, provide a value only for the direct use value and is not appropriate for non-use values (i.e. valuing the bequest or existence value of nature or individual species). The basic approach is to elicit data on visitors’ total expenditures undertaken in order to visit a site, including the entrance fee, travel costs and time spent travelling. Then, their demand curve for the service provided by the site is derived. The travel costs needed to reach the site can be considered the implicit or the surrogate price of the visit.

- ***Averting Behaviour Method (ABM)***

The averting expenditures or averting behaviour method uses revealed preferences on conventional markets and is based on behavioural linkages. This approach assesses the value of non-marketed goods through the real expenses spent by households or producers for a certain marketed good or service in order to: i) prevent an environmental impact, or ii) prevent a utility loss by environmental degradation, or iii) change their behaviour to acquire greater environmental quality.

- ***Contingent Valuation Method (CVM)***

CVM introduces hypothetical situations to a (representative) sample of a population, situations which are often presented in a questionnaire in order to elicit willingness to pay or willingness to accept compensation for a contingent product. In principle, a CVM survey can consist of three parts: first, basic information about the contingent product is offered to the respondent; then the WTP/WTA is elicited; and finally, the socio-economic characteristics or respondent attitudes are examined. Average (mean and median) WTP/WTA is calculated that could be weighted in order to get the representative value for the entire affected population.

The use of different elicitation formats has been explored in the literature, as when individuals are asked to place a single value in order to extract their willingness to pay for an environmental quality improvement, the values given are normally an underestimation of their real willingness to pay or their real preferences. This is because individuals reflect the value they place on a given environmental change as a range, rather than a single figure. This issue was analysed by Hanley and Kriström in 2002. They concluded that the use of a payment ladder approach that allows people to reflect their values as a range, instead of a single value, allows respondents to quantify this valuation uncertainty (Hanley and Kriström, 2002). However, these influences upon WTP suggest that simplistic models of preference formation, normally based on individual and immediate influences, are inadequate bases for CV. In principle, all the formats can be followed-up several times, except the payment card and referendum methods (Melichar and Ščasný, 2004).

Furthermore, CVM does not come without its limitations/criticisms. A typical problem found in the benefit estimation literature is related with the benefits procedure employed for aggregating non-use values. In particular, in the estimation of economic benefits derived from an environmental improvement using the CVM, there is a concern with the potential decrease of values with increasing distance from a given valuation site (Hanley, 2001). This is known as distance decay in the available literature. An English valuation study regarding river water quality improvements, carried out by Georgiou et al. (2000), included the calculations of distance decay effects. From the assumption of a linear distance-decay function, the authors derived the distance away from the river at which WTP estimates dropped to 0 for large water quality improvements (distance decay was found at 36 miles away from the site). The authors concluded that in studies where distance decay effects are not taken into account, the aggregation of benefits is often overestimated (Georgiou et al., 2000). This issue is particularly important in relation with the estimation of geographically spread improvements. Distance decay not only affects the aggregate estimation of non-use values for studies that have not calculated it, but also makes it very difficult to transfer benefits from studies that have calculated distance decay to other sites, as it would be necessary to assume the same site characteristics. The results obtained would therefore be less accurate.

- ***Conjoint Analyses (CA)***

These valuation methods do not directly ask people to state their values in monetary terms. Instead, values are inferred from the hypothetical choices or trade-offs that people make. Conjoint analysis is often described as a method where the respondent is asked to state a preference between one group of environmental services or characteristics at a given price or costs and another group of environmental characteristics at a different price or costs. Several approaches of conjoint analysis can be used such as choice experiments (see below), contingent ranking, paired comparison, contingent conjoint ranking or various similar techniques using choices, ranks or matches.

- ***Choice Experiments (CE)***

CE, as other stated preference approaches to valuation, involve eliciting responses from individuals in constructed, hypothetical markets, rather than the study of actual behaviour. The technique is based on random utility theory and the characteristics theory of value (Hanley et al. 2006). Environmental goods are valued in terms of their attributes, by applying probabilistic models to choices between different bundles of attributes. By making one of these attributes a price or cost term, marginal utility estimates can be converted into WTP estimates for changes in attribute levels, and welfare estimates obtained for combinations of attribute changes. CE permits to estimate values for different component parts, or aspects, of environmental quality. These component parts constitute the attributes in the CE design.

2.3 Benefit Transfer

Benefit Transfer (sometimes also called more neutrally *Value Transfer*) is not a specific valuation method which would generate a monetary value itself. Benefits transfer is rather a method that estimates economic values for non-market goods and services by transferring available valuation information from original studies already completed to another – but similar – site (the policy site) where monetary values are required. Benefit Transfer is applied when there are not sufficient resources (time or money) available to carry out primary valuation studies at the policy site. The values estimated for particular ecosystem services on the original study site are applied in the area where there is a need to be informed about the economic value of a certain ecosystem or particular ecosystem components. The transfer of economic values of individual ecosystem services from a particular study site has become a common tool to estimate the value of natural resources.

Another approach of transferring economic values for ecosystem services is called upscaling. In the scaling-up exercise, economic values from a particular study site are transferred to another geographical setting, for instance to a regional, national or global scale. Local values are thus not applied in another local context, but are used to estimate the values of all ecosystems (or ecosystem services) of similar characteristics in a certain region.

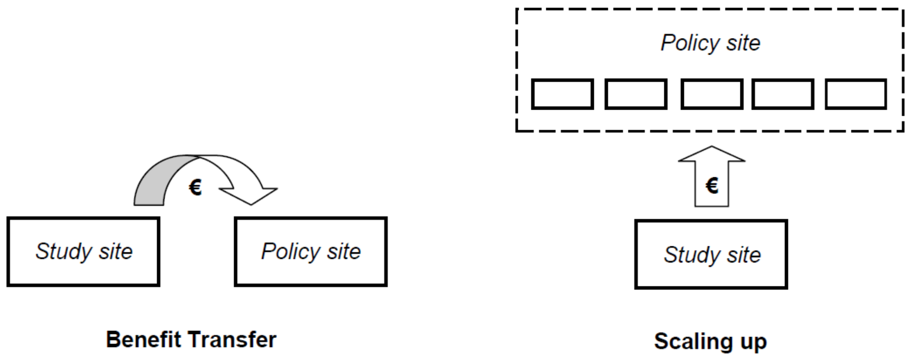


Figure 5 - Benefit Transfer and large-scale value transfer.

Benefit Transfer is usually applied on a case-by-case basis. For instance, a cost-benefit analysis carried out for an individual nature reserve can be transferred to a similar nature reserve. The upscaling of economic values, on the other hand, is usually applied in more strategic policy contexts, for example in the field of policy evaluation, and is mainly used for strategic policy-planning. While the value-transfer exercise is already complex, the scaling-up exercise is accompanied by even more complexity, methodological difficulties and uncertainty. The word “upscaling” already reveals that (spatial) scale is a vital element of the method.

3. Concluding remarks

Valuation methods can be divided into three categories: market valuation approaches, revealed preference approaches, and stated preference approaches. These approaches can be used to analyze conventional, surrogate or hypothetical markets, respectively, for the estimation of use and non-use values.

Stated preference approaches, and particularly the Contingent Valuation Method, are often criticized due to their inherent uncertainties when it comes to the assessment of people's willingness to pay for certain environmental goods or services. Market valuation methods, on the other hand, analyze statistical data (public market prices) and are thus generally accepted among researchers and policy-makers. However, when it comes to the valuation of non-use values (non-priced environmental goods or services), applying stated preference techniques is mostly the only way to elicit people's preferences, i.e. their willingness to pay for improvements in environmental quality or their willingness to accept compensation for reductions in environmental quality.

The integration of natural capital into public decision-making requires the application of both market and non-market valuation methods. As explained in Section III.1, non-use values represent an important component of an ecosystem's Total Economic Value. In order to inform policy-makers on the full implications of their policy decision, as many TEV components as possible should be covered by a valuation study. By ignoring the ecosystem's non-use values, a valuation study would clearly underestimate the TEV of the ecosystem. If incorporated into a cost-benefit analysis of alternative policy options, the valuation results would lead to flawed decisions.

For a policy-maker, it is important that the results of a commissioned valuation study are not vulnerable to criticism and dismissal. The lack of established and consensual methodologies makes valuation studies prone to such criticism and consequent dismissal. They can be easily attacked in view of the methodology used, both with regard to the complexity of the physical processes involved and the methodological limitations implied. In order to ensure credibility and acceptance of the valuation results, it is therefore important to be transparent regarding the assumptions underlying a particular methodology and/or individual study, by laying open study assumptions and clearly stating what is included and what is excluded from the study (Gerdes and Raggamby, 2010).

Thus, the choice of the valuation method is often the determining factor for the success or failure of the valuation results entering the policy process.

Part IV – Evidence on the value of biodiversity and ecosystem services in ORs/OCTs

1. Data sources and uncertainty

Environmental valuations were first carried out in the United States in the 1950s for project evaluations. Since the late 1970s, they were also applied for evaluating new regulations (Navrud and Pruckner, 1997). In the following decades, a growing amount of environmental valuations has been commissioned to inform political decisions. However, for a long time, environmental valuations were mainly applied in Anglo-Saxon countries, where economic assessments and cost-benefit analysis usually play a high role in the political decision-making process. Within the EU, the scientific and political interest in environmental valuations was restricted to the United Kingdom.

While in Europe many environmental assessments used to be carried out in terms of ecological changes and not in monetary terms, a shift towards monetary valuations can be observed from the 1990s onwards. The Millennium Ecosystem Assessment in 2005 and more recently the TEEB initiative have led to a growing amount of monetary valuations being carried out. The literature review was limited to *monetary* valuations of ecosystem services, leaving out quantitative and qualitative assessments, as it was assumed that these would be less suitable for communicating the socio-economic impacts of changes in the state of biodiversity and ecosystem services in the context of the planned stakeholder activities under Task 4.2 of NetBiome-CSA.

The database consists of peer-reviewed and grey literature published between 1990 and 2013; overall, there are 39 studies in the database. In order to broaden the database, valuation studies from the EU Outermost Regions and Overseas Countries and Territories have been complemented with relevant valuation studies from other regions and territories. The data have been collected by means of a literature search in the World Wide Web and by consulting existing databases, such as the TEEB Valuation Database, the AgEcon database, and the Environmental Valuation Reference Inventory (EVRI). In addition, the results of a stakeholder survey carried out under Task 2.1 of NetBiome-CSA have been taken into account.

Various scientific uncertainties enter the equation when valuing non-market goods and services. Particularly, three main sources of uncertainty should be noted. First, the researcher's limited knowledge about the biophysical conditions and ecological functions of the ecosystem under assessment. Second, the researcher's limited knowledge about the welfare implications of biophysical changes. And, third, the limitations of the valuation exercise itself particularly with regard to the valuation methods applied and potential biases on the side of the respondents. These uncertainties should be taken into account when interpreting the individual benefit estimates as presented in the following section and in Annex 1 of this Report.

2. Results of literature review

The results of the literature review on existing valuation studies in the EU Outermost Regions and Overseas Countries and Territories are summarized in the following Table 4. The table includes references to the corresponding papers, covered biomes and ecosystem services, geographic details, valuation methods and monetary estimates of benefit valuations. Monetary values are described in Euros per hectare per year, unless noted otherwise. The complete list of valuations (n = 110) is presented in Annex 1 of this Report.

In order to provide for a meaningful database, the geographic scope of the literature review was extended beyond ORs and OCTs. This means that case studies from other relevant regions and territories have been taken into account. For instance, Mohd-Shahwahid and McNally (2001) carried out a valuation study in Samoa, covering multiple ecosystem services provided by forests. Although not part of the EU ORs/OCTs, its close proximity to New Caledonia (France) makes the valuation results relevant for the *sui generis* group. Similarly, Eade and Moran (1996), Godoy et al. (1993), Hargraeves-Allen (2008) and Trejo (2005) carried out valuation studies in Belize. While the former British colony is not part of the EU ORs/OCTs, the valuation results are relevant for the Dutch, French and British ORs/OCTs located in the Caribbean.

The valuation database will be updated and expanded throughout the NetBiome-CSA project.

Table 5 - Summary of identified literature.

Reference	Ecosystem / Biome	Ecosystem Service(s) covered	Case study location	Valuation Method	Value Range (EUR/ha/year)
Aubanel, 1993	Coral reefs	Extreme events, recreation	French Polynesia	Replacement Cost	5,914.44 - 7,097.33
Bervoets, 2010	Coral reefs	Recreation, fisheries	St. Maarten	Direct market pricing	1,851,377.99 - 57,645,666.39
Beukering et al., 2012	Multiple	TEV	Caribbean Netherlands	Contingent Valuation, Choice Experiment	19,096,200.00
Burke and Maidens, 2004	Coral reefs	TEV, food, recreation, extreme events	Caribbean	Direct market pricing	116.00 - 1,443.63
Burke et al. 2008	Coral reefs	Food, recreation, extreme events	Saint Lucia	Direct market pricing	189.47 - 49,977.90
Charles, 2005	Coral reefs	TEV, multiple ecosystem services	French Polynesia	Direct market pricing	47.32 - 16,184.01
Chong et al., 2003	Coral reefs	Recreation, science / research	Caribbean	Benefit Transfer	35.13 - 34,803.92
Conservation International,	Coral reefs	TEV	Turks and Caicos	no information available	1,041.02

Reference	Ecosystem / Biome	Ecosystem Service(s) covered	Case study location	Valuation Method	Value Range (EUR/ha/year)
2008			Islands		
Cooper et al. 2009	Coral reefs	Recreation, food, extreme events	Belize	Direct market pricing	22.30 - 990.82
Dharmaratne and Strand, 2002	Salt water wetlands	Nursery service	Caribbean	Factor Income / Production Function	378.19
Dixon et al., 1993	Coral reefs	Recreation	Netherlands Antilles	Direct market pricing	1,597.00
Eade and Moran, 1996	Tropical forest	Food, medical, genetic, erosion, extreme events, gene pool	Belize	Benefit Transfer	0.45 – 2,654.73
Echeverria et al., 1995	Tropical forest	Recreation	Costa Rica	Contingent Valuation	330.81
Godoy et al., 1993	Tropical forest	Genetic	Belize	Direct market pricing	146.86
Gren and Soderqvist, 1994	Mangroves	TEV, food, raw material	Puerto Rico	Benefit Transfer	26.99 - 2,991.42
Hamilton and Snedaker, 1984	Mangroves	TEV, food	Fiji Islands	Benefit Transfer	1,427.33 - 3,777.35
Hargraeves-Allen, 2008	Coral reefs	Tourism, fisheries	Belize	Contingent Valuation	3,610,514.02
IFRECOR, 2012	Coral reefs	Extreme events, recreation, bio-prospecting, research/education, fisheries	New Caledonia	no information available	2,652,250.00 - 4,243,600.00
Krutilla, 1991	Tropical forest	Recreation	Costa Rica	Travel Cost	109.10
Lacle et al. 2012	Multiple	Recreation	Bonaire	Choice Experiment	*2,231,526.77 - 3,001,018.75*
Lal, 1990	Mangroves	Nursery	Fiji Islands	Direct market pricing	746.34
Mathieu et al. 2003	Continental Shelf Sea	Recreation	Seychelles	Direct market pricing	24.70
Mohd-Shahwahid and McNally, 2001	Forest	Multiple ecosystem services	Samoa	Benefit Transfer	0.05 - 5.60
Naylor and Drew, 1998	Mangroves	TEV, food, raw material, gene pool, extreme events	Micronesia	Contingent Valuation	134.39 - 1,972.47
Pagiola et al., 2004	Tropical forest	Gene pool	Costa Rica	PES	43.86
Pendleton, 1995	Coral reefs	Recreation	Netherlands Antilles	Travel Cost	7,065.27
Raboteur and Rhodes, 2006	Coral reefs	Genepool	Guadeloupe	Contingent Valuation	70.98

Reference	Ecosystem / Biome	Ecosystem Service(s) covered	Case study location	Valuation Method	Value Range (EUR/ha/year)
Rausser and Small, 2000	Tropical forest	Medical	New Caledonia	Factor Income / Production Function	1,040.78
Ricketts et al, 2004	Tropical forest	Pollination	Costa Rica	Direct market pricing	129.09
Schep et al., 2012	Coral Reefs	Fisheries	Bonaire	Direct market pricing	*307,796.80 - 538,644.39*
Schep et al., 2013	Multiple	Recreation	Bonaire	Choice Experiment	*37,353,980.00*
Shultz et al., 1998	Tropical forest	Recreation	Costa Rica	Contingent Valuation	1,785.55
Thur, 2010	Coral reefs	Recreation	Bonaire	Contingent Valuation	*63.08 – 138.57*
Tobias and Mendelsohn, 1991	Tropical forest	Recreation	Costa Rica	Travel Cost	78.73
Trejo, 2005	Marine	Recreation	Belize	Contingent Valuation	10.18
Uyarra, 2010	Coral reefs	Recreation	Bonaire	Contingent Valuation	29.01
Zanten and van Beukering, 2012	Coral reefs	Extreme events	Bonaire	Damage cost approach	*25,720.27 - 53,951.39*

3. Analysis of results

With regard to the biomes and ecosystems covered by the valuation studies, coral reefs, tropical forests and mangroves dominate in the literature (Figure 6). The focus on marine ecosystems is not surprising, as all ORs/OCTs are insular states. With regard to the non-OR/OCT literature included in the review, the countries covered are likewise dominated by coastal or marine ecosystems. With almost 60% of the literature covering marine or coastal ecosystems, one can conclude that those ecosystems are perceived relatively important in terms of the ecological functions they provide. At the same time, the focus on coastal or marine ecosystems in the identified literature may also hint at the fact that they are particularly threatened by natural and/or human-induced impacts, and may therefore have been selected for an economic assessment.

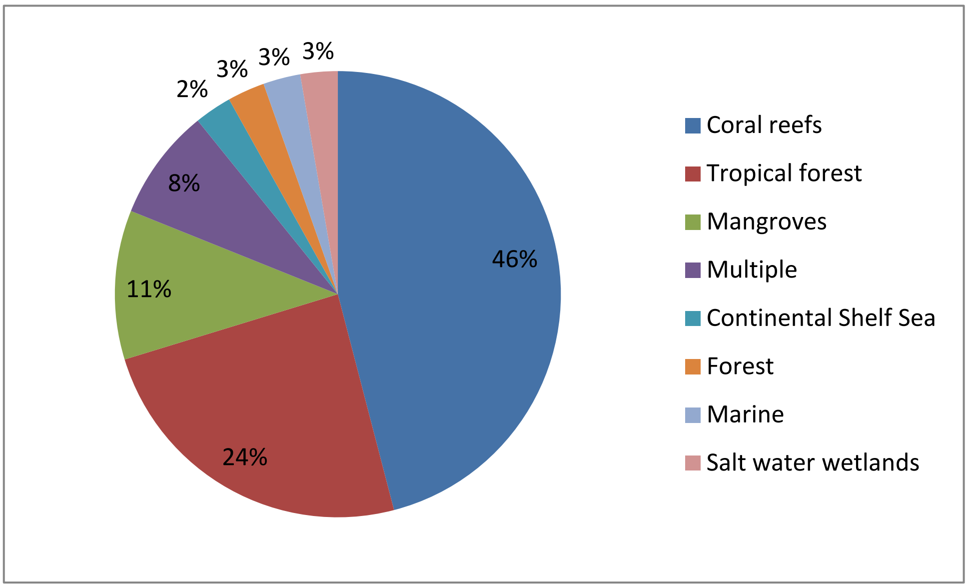


Figure 6 - Distribution of ecosystems among the reviewed literature.

The focus of the identified valuation literature on coastal and marine ecosystems is also reflected in the distribution of individual ecosystem services covered. Among the 105 benefit estimates included in the database (see Annex 1), 22 cover recreational ecosystem services (tourism), 15 are related to the provision of food (mainly fish), and 11 cover extreme events (e.g. flood protection). 10 benefit estimates for the provision of genetic resources have been identified, which may link to the high level of biodiversity in the respective case study locations. Seven valuations did not investigate individual ecosystem services, but estimated the Total Economic Value of the respective ecosystem.

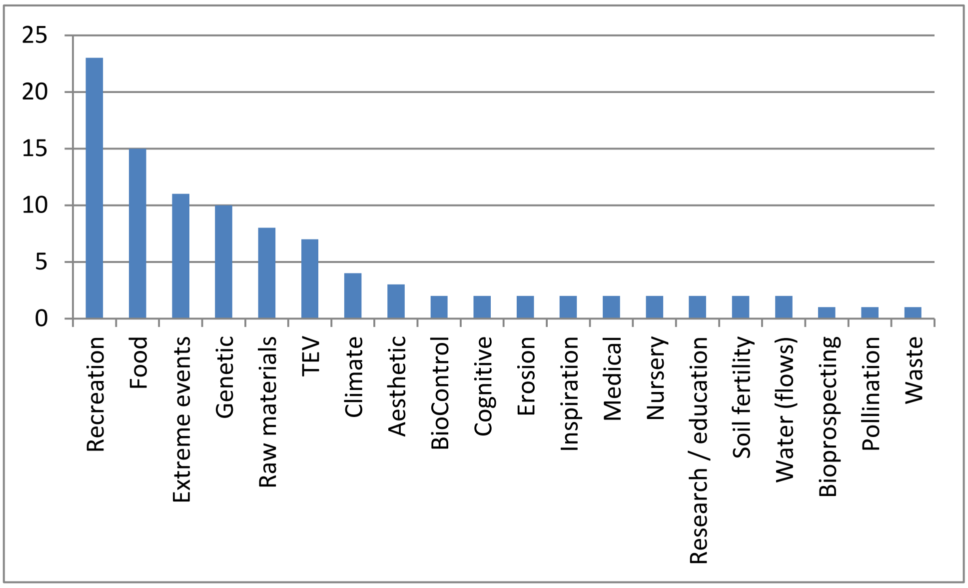


Figure 7 - Overview of ecosystem services covered in the reviewed literature.

With regard to the valuation methods applied in the identified literature, direct market pricing dominates, followed by the Contingent Valuation Method and Benefit Transfer (Figure 8). This reflects the range of ecosystem services covered by the valuation studies: the benefits of food provision and coastal protection services result in direct and indirect use values, respectively, which can be determined by direct market pricing. Recreational benefits are classified as direct use values which can be determined by applying the Contingent Valuation Method or Choice Experiments (cf. Chapter III.2). Five valuation studies are not based on primary valuations; instead, Benefit Transfer techniques have been applied, which made use of primary valuations that had been previously carried out in locations which are similar to the valued ecosystem.

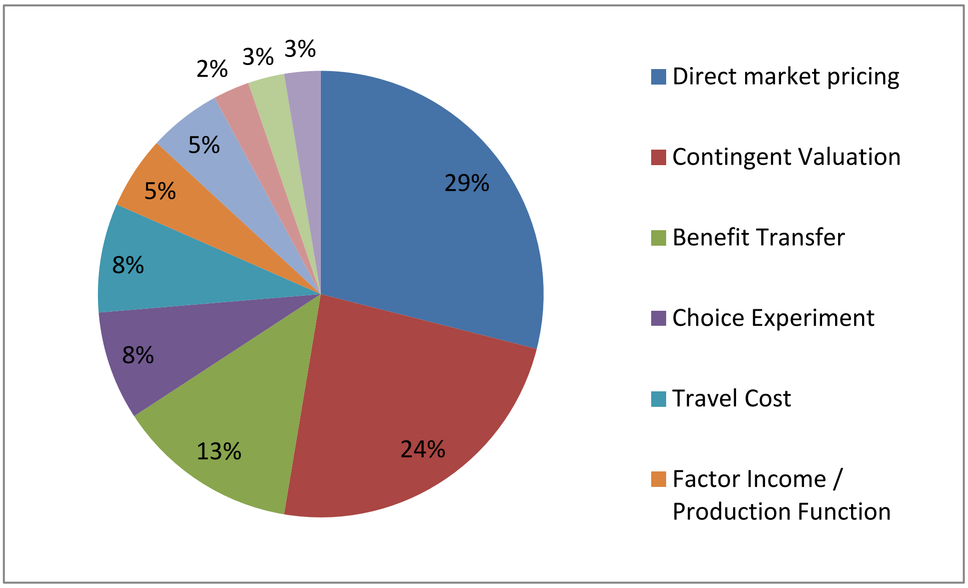


Figure 8 - Distribution of valuation methods applied among the reviewed literature.

Reported in Figure 9 are average benefits per hectare and year that include all comparable observations for a selected ecosystem service. The values have been converted to EUR and inflation adjusted by the Consumer Price Index (CPI) for 2014. The findings are discussed below. The figure shows the minimum and the maximum (whiskers) of specific ecosystem service benefits. The box bottom shows the first quartile, the band shows the median and the top of the box shows the third quartile of the selected values of ecosystem services. According to the very broad variety of the values, for less than 1 Cent up to 50,000 Euro (adjusted 2014/ha/year), the axis have logarithmic values. The first box plot includes all specific ecosystem services of different ecosystems, and the other three box plots include the ecosystem benefits of Tourism/Recreation, Fisheries, Genetic resources and Raw materials of different ecosystems.

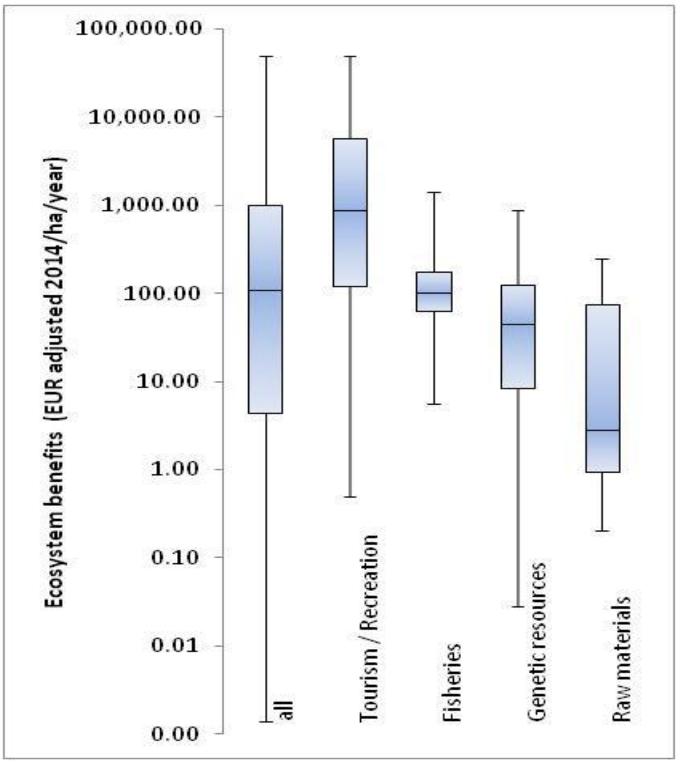


Figure 9 - Value ranges for selected ecosystem services (n = 92)

All included ecosystem services (n=92) have a wide value range between a minimum of less than 0.01 EUR up to a maximum of 49,978 adjusted EUR/ha/year. One quarter of the estimated values are below 5 EUR (1st quartile), the median value is 110 EUR and one quarter of the values are above 1,003 EUR (3rd quartile). The average estimated value is 2,001 EUR.

The observed ecosystem services of **tourism and recreation** (n=18) have the highest values compared to the other services including the maximum of 49,978 adjusted EUR/ha/year across all case study locations. There are two outliers (the two estimates from Samoa with 49,978 and 34,804 Euro) followed by values below 7,065 EUR. The 1st quartile is 123 EUR, the median 889 EUR and the 3rd quartile 5,745 EUR.

For the ecosystem service **fisheries** (n=10) the estimated range of the values of ecosystem benefits is between 6 EUR and 1,427 adjusted EUR per hectare and year. The second highest value (below the maximum of 1,427 EUR) is 202 EUR per hectare and year. One quarter of the estimated values are below 63 EUR (1st quartile), the median value is 101 EUR and one quarter of the values are above 176 EUR (3rd quartile). The average estimated value is 232 EUR.

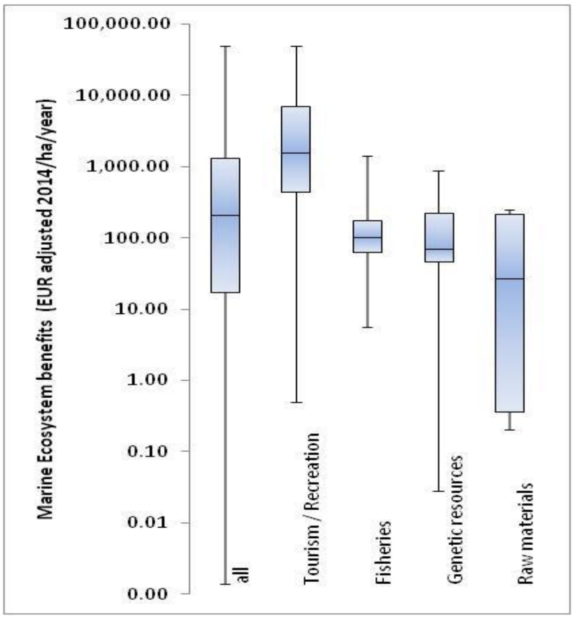
The benefits of **Genetic resources** (n=10) have a range between a minimum of 0.03 EUR and a maximum of 893 adjusted EUR/ha/year. The 1st quartile is 9 EUR, the median 46 EUR and the 3rd quartile is 128 EUR. The average of genetic resources is at 145 EUR.

The ecosystem services of **raw materials** (n=8) is lower and spreads between 0.20 EUR and 252 adjusted EUR/ha/year. The 1st quartile is below 1 EUR, the median only 3 EUR. The 3rd quartile is 75 EUR and the average is at 63 EUR.

All services also represented in the first column in the figure above. Not separately visualised, but like the other ecosystem services in the box plot (all) are e.g. food provision (n=3) and flood protection (n=3). For food provision, the estimates values were 0.42 and 0.45 EUR (both Benefit Transfer) and one other study estimated 435 EUR (direct market pricing) per hectare and year in three different case study locations. The ecosystem service **flood protection** (extreme events) is valued in three studies with 30 EUR; 551 EUR and 1,079 EUR per hectare and year across different case study locations.

Among the identified valuation literature, a clear focus on marine and coastal ecosystems and their main ecosystem services is evident. This can be attributed to the fact that all ORs/OCTs are insular states, implying that marine and coastal ecosystems play an important role in the perception of the local population. This is also reflected in the fact that the dominant ecosystem services covered in the literature are marine and coastal ecosystem services, followed by ecosystem services provided by tropical forests.

The figures below are showing the benefits of marine (n=66) and forest (n=26) Ecosystem services. The axes have logarithmic values and according to the lower values of the forest ecosystem benefits the values of the axis of the two figures have different scales. The box plots represent all ecosystem services and the range of tourism/recreation, fisheries, genetic resources and raw materials of the marine ecosystem (left side of the next figure). All benefits of the forest ecosystem (right side of the next figure) are shown in the first column. In addition tourism/recreation, genetic resources and raw materials are shown on the other columns.



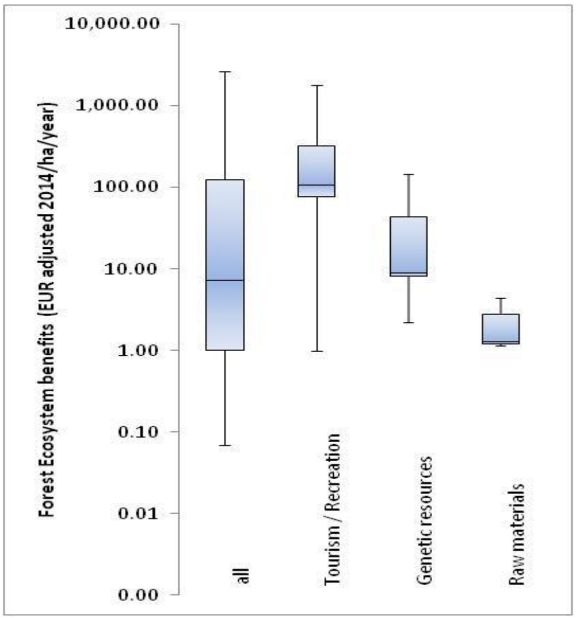


Figure 10 - Value ranges for Marine and Forest ecosystem services (Marine n= 66, Forest n = 26).

Comparing the benefits of the **marine (all)** and the **forest (all)** ecosystem services the valuation of the marine (all) ecosystem benefits (EUR adjusted 2014/ha/year) are much higher than the benefits of the forest (all) ecosystem benefits. The valuation of the 1st quartile (17 EUR marine and 1 EUR forest) , the median (211 EUR marine and 7 EUR forest) and the 3rd quartile (1,340 EUR marine and 124 EUR forest) are substantial different. The averages (2,658 EUR marine and 331 EUR forest) are also very far away from each other.

The ecosystem services **tourism and recreation** are contributing with high values to both biomes marine and forest. There is a tendency in the observed the studies to estimate **genetic resources** and **raw materials** lower than tourism and recreation (marine and forest biomes). The benefits of **fisheries** are estimated below the average of the other marine ecosystem services (all). These effects can also be seen in the next figures.

The following figures are showing the marine and forest ecosystem benefits on an individual marker basis. Every ecosystem service is represented by one marker. The axes have logarithmic values and the same axis for marine and forest. The ecosystem services tourism and recreation; fisheries; genetic recourses and raw materials are displayed separately. The other ecosystem benefits are shown in the last column.

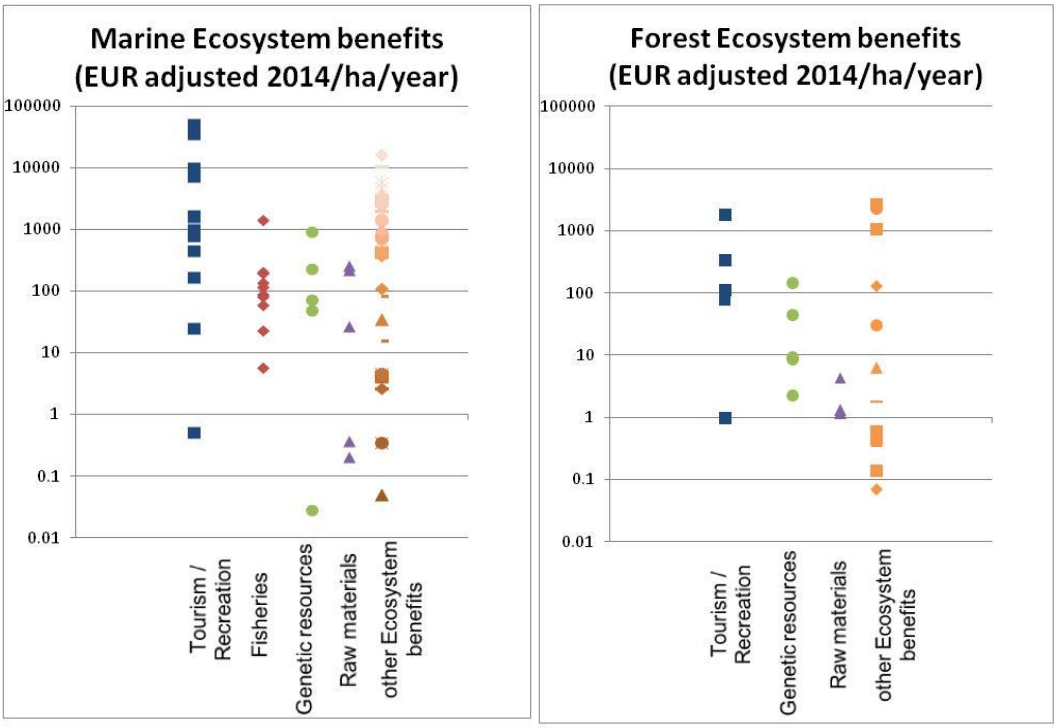


Figure 11 - Individual values for Marine and Forest ecosystem services (Marine n= 66, Forest n = 26).

These examples show that, depending on a variety of factors, ecosystem services may be valued differently in different locations. People's preferences vary according to location, their socio-economic context and the abundance or scarcity of ecosystem services in their location. The uncertainties underlying the methods for the economic valuation of natural resources may be another reason for large estimate ranges. Particularly stated preference techniques such CVM are often criticized for their shortcomings, which may influence the estimated values: Depending on the design of the questionnaire used in the survey, the interviewer might not be able to eliminate potential errors such as strategic behaviour of the respondents, protest answers, response bias (where the respondents would provide the interviewer with the "morally right" answer), and respondents ignoring income constraints. These and other shortcomings of certain valuation methods need to be taken into account when interpreting the benefit estimates. The factors which enable or hinder the uptake of environmental valuations in the policy process will be further discussed in Chapter V.

In general, however, the results of the literature review reveal awareness for environmental valuations in ORs/OCTs. 35 individual valuation studies were identified in ORs/OCTs and related regions and territories (e.g. Samoa and Belize), containing 105 individual valuations of ecosystem services. It turned out that certain locations, biomes and ecosystem services are stronger represented than others, implying that there is room for future valuation efforts. While the design and implementation of primary valuation studies (particularly non-market valuations) is usually dependent on abundant financial resources, the collected evidence could potentially provide the basis for the transfer of values by means of Benefit Transfer to other ecosystems at local, regional or even national scale. This could be an option for ORs/OCTs which are currently underrepresented in the existing valuation literature.

Part V – Good-practice examples

1. The policy impact of environmental valuations

While authors of valuation studies usually stress the relevance of their results for policy-making, evidence on the successful uptake of environmental valuations in policy processes remains limited. To some extent, the reason may be found in issues that are connected to the science-policy interface. Gupta (2005) highlights two key theories regarding science and its relation to the policy process. Firstly, the scientific culture is regarded very different from that of policy culture and the two often encounter difficulties communicating effectively with each other. Secondly, science is selectively used by policy-makers – it is only used if it is consistent with the policy-makers' expectations, understanding, and interests.

These theories have their roots in traditional research practices, which have generated a disciplinary structure of science. This form of knowledge production is characterized by "cognitive and social norms, which must be followed in the production, legitimization and diffusion of knowledge" (Gibbons 2001). However, one can observe a blurring of the distinction between science and policy. A movement towards multi-disciplinary, inter-disciplinary, and trans-disciplinary science is taking place where knowledge is produced in the context of its application. This new mode of research has the following characteristics: (i) it is problem-focused rather than based on the development of theory, (ii) trans-disciplinary rather than based on a single discipline, (iii) more accountable and subject to quality control, and (iv) more likely to be undertaken in a wide variety of organizations (Scott 2000).

On the side of the policy-maker, the uptake of valuation results is primarily hampered by mistrust in the valuation procedures. According to Barde and Pearce (1991), reasons for mistrust include (i) a lack of familiarity with valuation procedures, (ii) the belief that valuation techniques do not give rise to "real" values (compared to direct monetary impacts), and (iii) a lack of belief in the underlying paradigm.

Moreover, decision-makers' disbelief in the underlying valuation methodologies may be accompanied by a perceived unimportance of the problem at stake. In this context, Cash and Clark (2001) identify three elements which determine the uptake of assessments by policy-makers: (i) the perceived relevance or value of the assessment to particular groups who might employ it to promote any of the policy changes (saliency), (ii) the perceived authoritativeness or believability of the technical dimensions of the assessment process to particular constituencies (credibility), and (iii) the perceived fairness of the assessment process to particular constituencies (legitimacy).

When applying these theories to the EU Outermost Regions and Overseas Countries and Territories, one can certainly identify some aspects which may prevent the results from environmental valuations to enter the policy process. This concerns the general knowledge about local biodiversity and its functions among policy-makers and stakeholders, the lack of experience with monetary valuation studies or economic assessments (such as cost-benefit analysis), and the strong impact of sectoral interests (e.g. tourism, fisheries, agriculture) on local and regional policy-making. Nevertheless, Waite et al. (2014) identified a number of cases in the Caribbean where the results of environmental valuations have influenced policy decisions. Table 5 provides an overview of the case study characteristics and the way the valuation results have entered the policy process.

Table 6 - Examples of uses of tropical coastal and marine ecosystem valuations in decision-making (Waite et al., 2014)

Reference	Location, biome, ecosystem services covered	Valuation method applied & reported impact on regional/local policy-making
Hargreaves-Allen (2010)	**Bahamas** Coral reefs, beaches, wetlands, forest, mangroves Use & non-use	Valuation method: Benefit Transfer "Justified the protection of the west side of Andros Island. The Bahamas Science and Technology Commission are also using the results to inform coral reef damage estimates; furthermore, valuation results are being used to raise awareness of the economic benefits of conservation to decision makers and the general public."
Cooper et al. (2009)	**Belize** Coral reefs / mangroves Tourism, fisheries, shoreline protection	Valuation method: Market prices, damage costs "Supported action on multiple fronts, including a landmark Supreme Court ruling to fine a ship owner an unprecedented and significant sum for a grounding on the Mesoamerican Reef; the government's decision to enact a host of new fisheries regulations (a ban on bottom trawling, the full protection of parrotfish, and the protection of grouper spawning sites); and a successful civil society campaign against offshore oil drilling."
Trejo (2005)	**Belize** Coral reefs Tourism	Valuation method: Contingent valuation "Justified the Hol Chan Marine Park increase in user fees, making it one of the few self-financed marine parks in the Caribbean."
Hargreaves-Allen (2008)	**Belize** Coral reefs Tourism / fisheries	Valuation method: Contingent valuation "Justified funding requests for ongoing planning and management of the Gladden Spit Marine Reserve, resulting in increased donations; additionally, valuation results helped the Gladden Spit Marine Reserve facilitate a historically strained dialogue with fishers and tour operators."
Clarke et al. (2013)	**Belize** Coral reefs, mangroves, seagrasses	Valuation method: Market price, production function, damage costs "Played a key role in the development of Belize national Integrated Coastal Zone Management Plan (currently in draft

Reference	Location, biome, ecosystem services covered	Valuation method applied & reported impact on regional/local policy-making
	Fisheries, tourism, shoreline protection	form) by ecosystem services provision and value in three coastal zoning scenarios: conservation, development, and informed management."
Figueredo Martín et al. (2009)	**Cuba** Coral reefs, mangroves, seagrasses Use & non-use	Valuation method: Contingent valuation, travel cost, benefits transfer, market price "Helped to justify the establishment of the Jardines de la Reina National Park, which includes the largest marine reserve (no-take zone) in the Caribbean region."
Wielgus et al. (2010)	**Dominican Republic** Coral reefs Dive tourism	Valuation method: Hedonic price, market price, contingent valuation, travel cost "Findings used to justify significant increase in user fees. Additional revenue has been used to help establish an aquatic center, a conservation fund to support park management, and a community fund to support local development projects."
Dixon et al. (1993) Uyarra (2002) Uyarra et al. (2010) Thur (2010)	**Bonaire** Coral reefs Dive tourism	Valuation method: Contingent valuation "Justified the Bonaire Marine Park adoption, and later increase, of user fees, making it one of the few self-financed marine parks in the Caribbean."
Bervoets (2010), WRI (2008a), WRI (2008b)	**St. Maarten** Coral reefs Tourism / fisheries	Valuation method: Market price, contingent valuation "Used by the government of St. Maarten to establish the Man of War Shoal Marine Park – the country's first national park; furthermore, the valuation results are currently being used to sue for damages caused by the sinking of a boat inside the Man of War Shoal Marine Reserve."

While not all of the discussed examples are from ORs/OCTs, this overview shows that some policy makers are open to environmental valuations and willing to base their decisions on the communicated results. While direct market pricing has been applied in the majority of the listed valuation studies, the overview shows that in certain cases policy-makers even accept valuation results which are based on stated and revealed preference techniques, e.g. the Contingent Valuation Method or Hedonic Pricing. The next section presents three best-practice examples in which the results of environmental valuations have successfully entered the local or regional policy process.

2. Case Studies

The following section will describe in detail three case studies which serve as best-practice examples for the use of environmental valuations in local and regional decision making of selected ORs/OCTs and related territories. Each case study will provide information on the socio-economic context, the ecological challenges addressed, the applied valuation methods, the estimated magnitude and potential transferability of the economic benefits, and the impact of the valuation study on local and/or regional policy making. Geographical distribution, ecosystem characteristic, provision of ecosystem services, applied valuation methods and data availability for each case study is included as well. Notably, the relevant valuation literature is biased towards coral reefs or marine ecosystems in general (cf. Chapter IV.2 and Annex 1), as reflected in the final selection of case studies since the three cover marine ecosystems. Moreover, previous research on the uptake of environmental valuation has focused on one geographical region, namely the Caribbean.

2.1 The Economic Value of Bonaire's ecosystems and National Marine Park

Shoreline on Bonaire. Source: Janderk / Wikipedia Commons.

As part of the Caribbean Netherlands, Bonaire represents one of the EU Overseas Countries and Territories. Together with Aruba and Curaçao, it is located off the north coast of South America. Bonaire used to be part of the Netherlands Antilles until 2010, when it became a special municipality within the Netherlands country. Nature and fisheries regulations have largely been taken over from the former Netherlands Antilles (Ministry of Economic Affairs, 2013). The following chapter covers two case studies; the first examines the TEV of Bonaire ecosystems, while in the second the TEV of the National Marine Park on the island is analyzed. Despite dating back only one year and containing no information about the influence on local/regional authorities, the first case study is nevertheless a very good example, since stakeholders from the local government were intensively involved from the beginning of the project. Generally, the research conducted within the two case studies gives many opportunities for decision makers to improve and prevent the ecosystems on the island.

Socio-Economic Situation

The economy of Bonaire is mainly based on tourism, while salt mining is also a significant industry with a long tradition in the island. Salt pans cover 10% of Bonaire surface, and the island produces approximately 441,000 tons (400,000 metric tons) per year.[2] Due to the climate and the geography , farming does not play an important role for the economy of Bonaire, providing food only for local consumption. Aloe is the only export crop and it generates some income for the local farmers. Still, more than half of the country GDP is derived from tourism, particularly dive tourism. Fishery is also playing a significant role for the economy of the island. Because of this, the economy of the island relies on the quality of the island's ecosystems.

Ecosystem Characteristics

Bonaire is a very flat island; in the northern part of the island the higher elevation is reaching a maximum of 238 m. According to Köppel's classification, the climate is arid-tropical with high temperature and low rainfall, and almost constant easterly trade winds (Strahler, 2002). Despite lying outside the hurricane belt, tropical storms and hurricanes passing north of Bonaire may cause extensive damage to the reefs and coastal zone of the leeward shore. Damaging wind reversals were recorded in 1976, 1981, 1985, 1990 and 1996 (de Meyer, 1997). A distinctive feature of Bonaire is the coral reef which surrounds the island; more than 55 species of coral can be found on the reefs inside the Bonaire National Marine Park (BNMP), and the marine environment has a generally rich biodiversity with more than 340 known species of fish. This park was established in 1979 and covers an area of 27 km² with different types of ecosystems, such as sea grass beds, beach areas, mangroves, lagoon areas, karstic systems and bacterial mats. On land, Bonaire is characterized by dry forests (van der Lely, 2013).

Bonaire has a long tradition in nature and particularly marine protection. In 1969, nearly 20% of the total land area of Bonaire was designated as a national park. 10 years later, the waters around Bonaire from the high water mark to the depth of 60 meters (200ft) had been designated a marine park and protected by law. Activities within the marine park are restricted in order to ensure the continued sustainability of the coral reef, sea grass and mangrove systems. Since then, Stichting Nationale Parken Bonaire (STINAPA Bonaire) has been managing the protected areas of Bonaire and caring for the preservation and conservation of nature on the island in general. On its website, STINAPA[3] describes its mission as "[..]dedicated to the conservation of Bonaire's natural and historical heritage through the sustainable use of its resources".

Most of the funding available for nature management in Bonaire is generated by user fees. According to the Ministry of Economic Affairs, the national parks in Bonaire generate 85% of their total budget for nature management, by means of the said fees. Part of the exploitation costs for the designated protected areas is covered by subsidies from the islands governing bodies (Ministry of Economic Affairs, 2013).

Social-Ecological Challenges

According to the park management, the main challenges for Bonaire's marine ecosystem include overfishing, nutrient enrichment, development and conversion of land use, poaching, heavy recreational use (snorkeling/diving), sedimentation, terrestrial run off, illegal sand mining and artificial beach creation. On the land side, environmental problems are related to the unregulated and illegal dumping of raw sewage and chemical pollutants, which leach through the permeable limestone of the island and threaten the quality of groundwater. Destruction of wildlife habitat for commercial development threatens endangered species, and it is also highlighted as a problem.

The sixth report of "Status and Trends Bonaire's Reefs in 2013" noted that 10% of Bonaire's reef corals bleached and died due to unusually warm sea temperatures during November 2010; this led to a sharp increase in reef damaging seaweeds. The decline in herbivorous parrotfish during the last 8-10 years has had a negative influence on the coral reef as well, since they graze upon the seaweed (Steneck et al., 2013).

Eutrophication is a serious problem affecting the coral reefs in Bonaire. "The reef of Bonaire faces nutrient inputs by various sources, of which enriched groundwater outflow from land is considered to be a substantial one. It is assumed that groundwater is enriched with nutrients e.g. due to leaking septic tanks" (Slijkerman et al., 2013).

Currently, Bonaire's government is poised to eliminate the legal protection against commercial construction in the Bonaire National Marine Park waters. According to experts, permitting urban development in the Bonaire National Marine Park could cause irreversible damage to coral reef ecosystems. Since the tourism in Bonaire is largely based on these coral reefs, this damage could have a negative impact on the Bonaire economy. Several initiatives against this decision have already been in place in order to protect the marine park from commercial construction.

Economic Valuation of Bonaire Natural Capital

[3] http://www.stinapa.org/index.html

The total economic valuation of Bonaire was estimated in 2012 to be within the project framework funded by the Netherlands Ministry of Economic Affairs[4]. The TEV of the 10 ecosystem services considered by the project team annually amounted to more than US$100 million. These values vary among the different ecosystem services as follows: tourism ($50 million), local recreation and cultural values ($3.9 million), support to fisheries ($1.1 million), research and educational services ($1.4 million), coastal protection ($0.1 million), and (most importantly) the non-use values enjoyed by people in the Netherlands ($15.5 million per month). If no action is taken to prevent the ecosystems, the TEV of Bonairean nature will decrease from today's $105 million to around $60 million in ten years' time, and to less than $40 million in 30 years (van der Lely 2013). The outcome of this project concluded that it is more efficient to prevent extensive environmental damage than attempting to revitalize the environment. A scenario that was aimed at the abatement of invasive species also proved very cost effective. For example, by removing the threat of goatfish and lionfish, the environment has had the possibility to regenerate. This demonstrates that interventions and policies must be aimed at preventing damage to Bonaire's nature.

Thur (2010) estimates the willingness of recreational scuba divers to pay for access to the Bonaire National Marine Park, by using a contingent valuation survey administered to 211 American scuba divers who had previously visited Bonaire. The results of the survey suggest that divers are willing to pay significantly more than the existing US $10 annual user fee for access to the park. 94% of respondents were willing to pay at least US $20, over 75% were willing to pay at least US $30, and more than 50% were willing to pay at least US $50. Annual mean WTP is estimated to be US $61, which is described by the author as a conservative estimate, based on higher estimates produced by other elicitation formats and the potential for strategic bias by park users.

Taking in to account the findings of Planter and Piña (2006) who estimated that "a revenue maximizing fee (US $50 for BNMP) results in a substantial decrease in total number of visitors (44%) and, as such, the adverse economic impacts of decreased tourism that such a fee would produce make it an undesirable policy option", it can be acknowledged that Thur (2010) shows "that nominal increases in fees can produce substantial increases in revenues without significantly decreasing overall tourism demand. For example, the US $20 fee for the BNMP dive tag, which was acceptable to 94% of divers, would generate over US $500,000 in revenues - nearly double the budget of the park" (IMS/REMP, 2012).

Influence on Local/Regional Policy-Making

The valuation studies commissioned by the Ministry of Economic Affairs of the Netherlands were successful in two respects. First, they raised awareness of the value of the island's coral reef and led to a decision to construct a water purification plant, in order to reduce the negative impacts of waste water on

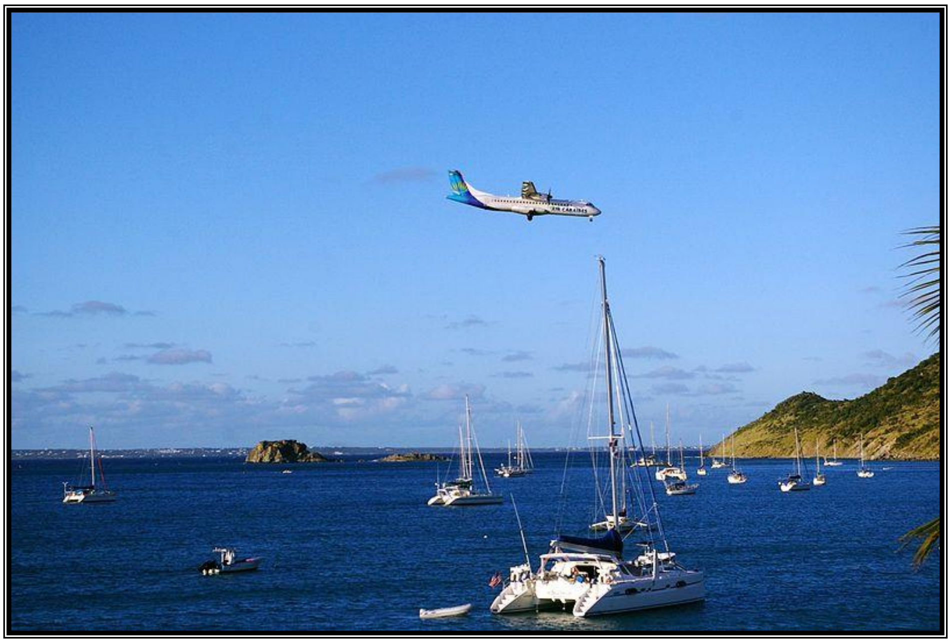
[4] http://www.ivm.vu.nl/en/projects/Projects/economics/Bonaire/index.asp, accessed on 19 March.

the coral reef. Second, based on the outcome of the valuation studies, the Ministry decided to provide additional funding – an extra 7.5 million Euros – available for nature protection in the Dutch Caribbean. The uptake of the valuation results by policy-makers and stakeholders was, to a large extent, based on the participatory approach that was applied by the researchers in designing and implementing the studies. This included workshops, training sessions and public debates, which served as a means to raise awareness about the economic valuations and their potentials in the design of sustainable management strategies.

The valuation study justified the Bonaire Marine Park adoption and later increase of user fees. The price of a dive tag was increased to US $25 in 2005. This made BNMP one of the few self-financed marine parks in the Caribbean.

This study on Bonaire National Marine Park shows how a relatively small investment in social science research can provide significant results supporting natural resource management. Before the beginning of the research, there was some concern among the local stakeholders that an increase in the user fee could lead to a decrease in the number of tourists. “The results of this study, coupled with the decision to charge other types of marine park users, enabled the management authority to convince the dive operators that little or no adverse effects would be caused by an increase in the dive tag price” (Thur, 2009). Moreover, this case study is a fantastic best-practice example, which could be applied to countries/regions with similar ecosystems and where tourism has a crucial role for the economy.

2.2 The Economic Value of St. Maarten's Coral Reef Resources



Grand Case Bay, St. Maarten. Source: Florian-Zet / Wikipedia Commons.

Like Bonaire, St. Maarten was previously part of the Netherlands Antilles until 2010, when it became a constituent country of the Netherlands Kingdom. It encompasses the southern half of the Caribbean island of Saint Martin, while the northern half of the island constitutes the French overseas collectivity of Saint-Martin. The ditch part of the island is the more densely populated of the two sides and due to this fact, it faces more environmental challenges. Saint Maarten's has a population of 39,689 (estimated for 2013). Since the country covers a total area of 34 km^2, it is not one of the most highly populated countries of the world. Due to the high number of unregistered illegal immigrants, the population is assumed to be much higher than the number provided by the official sources.

Socio-Economic Situation

St. Maarten is the second largest economy of the Netherlands Antilles, with a 20% share of total Gross Domestic Product and the highest per capita income among the five islands that formerly comprised the Netherlands Antilles. Since the end of the economic recession in 2001, the economy of St. Maarten has grown by an annual average rate of 3.8% in real terms, as compared to 1.8% for Bonaire. According to World Bank statistics, St. Maarten falls under the category of non-OECD high-income countries with a GDP of $400 million in 2003. The GDP for 2010 is estimated in $798.3 million. Only 10% of the territory is covered by arable land, and the agriculture accounts only about 0.4% of the GDP. Because of this, agriculture does not play a significant role for the island and almost all food must be imported, as well as energy resources and manufactured goods. St. Maarten economy is mostly based on tourism, either from stay-over tourists vacationing on the island or day tourists from the many cruise lines that come to the island. In contrast to

Bonaire, stay-over tourism is less important than cruise tourism. The number of stay-over visitors is approaching the 500,000 limit, while the number of cruise passengers has exceeded 1.4 million. In 2007, St. Maarten accounted for 56% of stay-over tourism and 76% of cruise tourism in the Netherlands Antilles. More than 80% of the labour forces are engaged in the tourism sector.

Ecosystem Characteristics

Vegetation type differs on the island per location, with evergreen seasonal forests found at higher elevations in the central hills, and drought deciduous and mixed evergreen deciduous thorn woodlands abundant in the lower plains. The island has numerous bays, rocky shores and white sandy beaches with coastal vegetation and succulent evergreen shrub land. Mangroves lines brackish ponds and parts of the Simpson Bay Lagoon. The (newly established) St. Maarten Marine Park covers an area of approximately 5,000 hectares and features coral reef, mangrove wetland, and sea grass bed ecosystems.

The vegetation is seasonal in evergreen forests, drought-deciduous and mixed evergreen deciduous thorn woodlands, and succulent evergreen shrub land. The mangrove forests are vital breeding grounds for reef fish and other marine life.

Social-Ecological Challenges

From the original 19 mangrove ponds found in St. Maarten only four remain, and they are threatened by development pressures and pollution. The sea grass beds suffer damages from boat anchors, pollution and dredging. Increasing tourism affects the marine environment as well, calling for the protection of habitats and species.

Coastal runoff has contributed to the coral reef degradation, and the increasing population and unbridled development are also important factors altering the ecosystems (Holian, 2012).

Economic valuation of St. Maarten's natural capital

Bervoets (2010) carried out a valuation of St. Maarten coral reef resources. The author has shown that St. Maarten coral reef resources provide important goods and services to the economy of the island, and estimates that the revenue that the resource is able to generate by means of coral reef associated tourism and fishery is approximately US $57.6 million per year. Bervoets (2010) applied direct market pricing in order to estimate the value and the economic impact of coral-reef related recreational activities and fisheries. The details are presented in Figure 3 below.

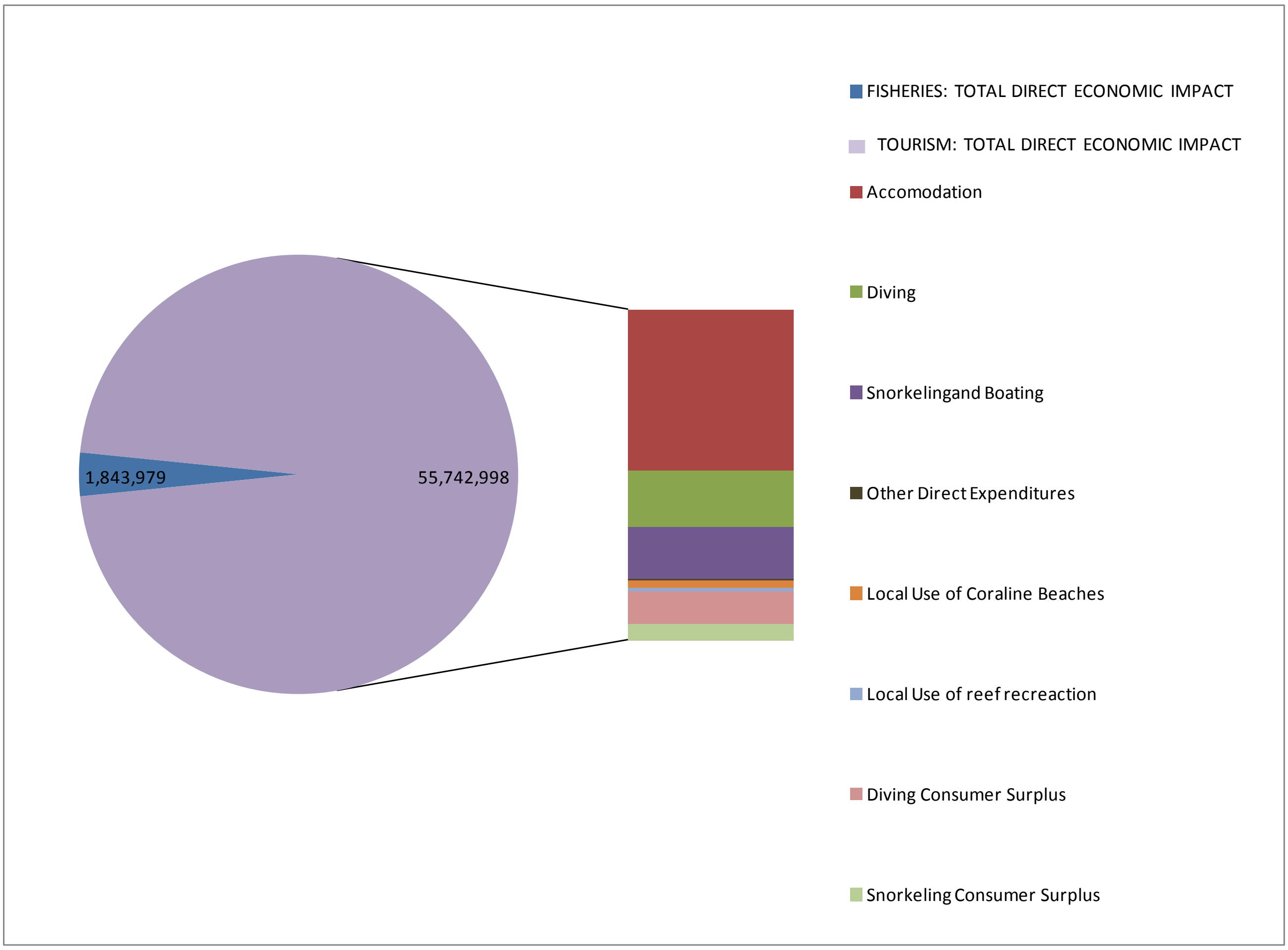


Figure 12 - Valuation of Coral Reef Resource (acccrding to Bervoets, 2010)

The author also develops specific policy recommendations. He suggests to a) establish a marine protected area, b) incorporate economic valuation into environmental impact assessments, c) include economic impacts in assessing fines for damages to coral reefs from activities such as anchoring in the reserves, oil spills, etc., d) weigh revenues from a growing tourism industry against long-term economic losses from environmental impacts, e) evaluate distributional effects of proposed coastal development projects, f) invest in scientific research, and g) increase support from the private and public sector in the proposed Marine Park Management Authority.

Influence on Local/Regional Policy-Making

Waite et al. (2014) found that the valuation results were "used by the government of St. Maarten to establish the Man of War Shoal Marine Park – the country's first national park; furthermore, the valuation results are currently being used to sue for damages caused by the sinking of a boat inside the Man of War Shoal Marine Reserve". In this context, Kushner et al. (2012) claim that "the government's recognition of the economic importance of coastal ecosystems, and the establishment of the country's first national park, are milestones for conservation and sustainable development"; and that "this recent success sets a precedent for other countries for how to make the political case for protecting ecosystems for the sake of people and the planet”.

2.3 The Economic Value of Belize’s Coral Reefs and Mangroves

Great Blue Hole, Belize. Source: U.S. Geological Survey / Wikipedia Commons.

Belize is not part of the EU Outermost Regions and Overseas Countries and Territories, but was originally part of the British Empire and became an independent member of the Commonwealth in 1981. Being part of the Caribbean, the country is situated in close proximity to the Cayman Islands (UK), Aruba, Curacao and Bonaire (Netherlands), with comparable ecosystems and similar climate. Although Belize is classified as upper middle income country faced with a more varied socio-economic situation, where ecosystem preservation is not a high priority of the government, this case study has the potential to transfer economic benefits. Thus, evidence from the country on the use of environmental valuations in local and regional decision-making can be regarded relevant for the EU ORs/OCTs. The literature review of valuation studies identified a total of 19 valuations that have been carried out on the Belize territory (see Annex 1 for details)

Socio-Economic Situation

According to the World Bank, Belize belongs to the upper middle income level countries. The GDP for 2013 is estimated on $3.083 billion. Belize has a small, mostly privatized enterprise economy, which is primarily based on the export of petroleum and crude oil, agriculture, agro-based industry and merchandising, while tourism and construction are recently assuming a greater importance. Belize is an oil-exporting country and a producer of industrial minerals. Sugar is the chief crop and accounts for nearly half of exports, while the banana industry is the country's largest employer. Other important agricultural products are cocoa, citrus, fish, cultured shrimp and lumber. However, and according to Cooper (2008), "tourism is a vitally important industry in Belize, contributing almost a quarter of GDP. Over 250,000 overnight tourists visited it in 2007, coming to see spectacular attractions both in land and on the coast."

Poverty is still a very large issue for the country. According to the last Country Assessment Report from 2009[5], 41.3 % of the population lives in poverty. The poverty rate estimated for 2013 remains unchanged. Poverty and ecosystem degradation are strongly linked since the poorest people and communities are predominantly rural in Belize, with their livelihoods largely depending on access to natural resources. Besides this, heavy foreign debt burden, high unemployment and growing involvement in the Mexican and South American drug trade are worsening the situation.

Ecosystem Characteristics

The total national territory of Belize covers 22,960 km², which includes 22,806 km² of terrestrial land and 160 km² of water (the coastline itself is 386 km long). Despite its small territory, Belize features an abundance of terrestrial and marine species, and a high diversity of ecosystems supporting the development of tourism, fishery, agriculture and forestry. Belize's rich biodiversity includes more than 150 species of mammals, 540 species of birds, 151 species of amphibians and reptiles, and nearly 600 species of freshwater and marine fishes. One can count thousands of plant species, including 200 different orchids and 500 species of trees. While inland vegetation is dominated by tropical rainforest, the coast is dominated by mangroves.

A total of 22.6 % of the whole national territory has been designated as protected areas (UNDP 2013, p. 80). The most important nature reserve is the Blue Hole National Park, which is home to a large number of rare plants and animals. An outstanding feature is the barrier reef, which extends almost 300 km along the coast of Belize, and includes other ecosystem types such as mangrove forests, sea grass beds, estuaries and numerous small islands or cays (Cooper et al., 2009).

According to Köppen's classification, Belize has a tropical, non-arid climate (Strahler, 2002); this climate is characterized by very hot and humid weather. The tropical ecosystems are often significant areas of biodiversity and species. Because Belize lies within the hurricane belt, the experts' expectancy is that the frequency and intensity of the hurricanes will increase due to climate change; this remains a real threat to Belize's ecosystems and to the people living there (Young, 2008).

[5] http://www.caribank.org/uploads/2012/12/Belize-2009-Report-Vol1.pdf, accessed on 18. March 2014.

Social-Ecological Challenges

The country of Belize is facing several environment-related problems. It is affected by climate change and vulnerability, not the least because it is a low-laying small island. Poverty is worsening the situation significantly. In the Sixth Annual Climate Change Vulnerability Index Report, published by the Maplecroft risk consultancy firm, Belize ranked 30 out of the 170 covered countries. According to the report, the most important climate change effects influencing Belize are:

- Frequent and severe weather events
- Increased incidents of tropical cyclones
- A mean sea level rise of 1 meter by 2050
- A mean sea surface temperature increase of 2 degrees Celsius by 2050
- Shorter wet periods with an increase in the amount of precipitation leading to increased incidents of flooding.

In particular, Belize's coastal zone is under increasing pressure, due to the rapid development of tourism and the expansion of aquaculture, primarily shrimp and tilapia farming (Cooper, 2009; Young, 2009). Coastal urban development leads also to the increase in solid waste and pollution of coastal waters. A major problem, resulting from the increasing number of tourists, is the improper waste management and the high water nutrient content. This has a negative impact on coastal lagoons, sea grasses, and reefs.

Mangroves play a crucial role in coastal tropical biodiversity by acting as a nursery for many species that live in and around coral reefs, providing multiple niches for great numbers of fish, crustaceans, and other species; their disappearance due to coastal development poses a serious threat to both mangrove and reef diversity in Belize. Cooper et al. (2009) report that over-fishing, pollution, deforestation, sedimentation, population growth and extreme weather events are the main threats to the mangroves and coral reefs along the coast of Belize. Coral reefs are in a status of severe degradation caused by the previously mentioned threats. McField and Bood (2007) found that corals have declined by more than 50% between the mid-1990’s and 2006. Coastal development and increasing tourism (e.g. snorkeling) in combination with overfishing, have worsened the situation.

“Terrestrial and marine protected areas are faced with trans-boundary incursions and illegal access that are beyond the scope of the domestic protected areas co-management agencies to deal with” (Young 2009). There is very limited capacity within the government to sustainably enforce protection regulations. Generally, the weak enforcement of the legislation -due to lack of financial and human resources- stand out as one of the most prominent challenges.

Economic Valuation of Belize’s Natural Capital

Cooper et al. (2009) carried out a valuation of Belize's coral reefs and mangroves. They estimated that the total value of fisheries, tourism, and coastal protection services provided by reefs and mangroves is between US $395 and US $559 million per year, compared to a national GDP of approximately US $1.5 billion (2011). The authors evaluated the annual benefits by applying direct market pricing and a damage cost approach. The details of the applied valuation methods and the valuation results are presented in Figure 2 below.

Box 4 - Coral Reef and Mangrove Valuations (Cooper et al., 2009)

"Coral reefs and mangroves are highly interconnected habitats, physically supporting each other and providing habitat for fish species. For example, mangroves filter sediment and pollutants from coastal runoff, supporting the clean water favoured by corals. Many species important to fisheries and tourism rely upon both mangrove habitat and coral reefs for a portion of their life-cycle.

This study did not directly evaluate the independent contributions of mangrove and coral reef habitats to the fisheries and tourism sectors, but assessed their collective value. To produce a rough estimate of the breakdown by habitat, we examined the proximity of mangroves to coral reefs across Belize to estimate the portions of fisheries and tourism values that a) rely exclusively on coral reefs, b) rely exclusively on mangroves, and c) depend upon both.

We estimate that US$60 - 78 million of Belize's annual tourism revenue stems from the presence of healthy mangroves. Approximately US$15 to $19 million of that total comes from tourist spending associated with mangrove-dependent activities, such as manatee tours and sport fishing, while the remaining $45 - $59 million is attributed to supporting services (including nursery habitat) provided by mangroves that grow in close proximity to reefs. Coral reef-associated tourism not supported by nearby mangroves earns an estimated US$90 - $117 million per year; in addition, reef-based tourism contributes $45 - $59 million that is supported by mangroves (mentioned above). The combined value of reef- and mangrove associated tourism is approximately US$150 – 196 million per year (see the Results section, below); revenues are only counted once, regardless of whether they are associated with one or both habitats.

For the fisheries sector, we assessed which fish species were primarily reef dependent, mangrove dependent, or depended heavily on both habitats, and allocated fisheries revenues accordingly. We estimate that mangroves contribute approximately US$3 to $4 million in fisheries value per year, while reef-based fisheries provide US$13 to $14 million per year.

The combined value of reef and mangrove-associated fisheries is US$14 - 16 million. Again, revenues are counted only once, even if a fish relies upon both habitats during its life cycle.

WRI's shoreline protection analysis differentiates between the protection provided by mangroves and reefs from the outset. Mangroves play an especially important role in buffering against storm surge and reducing erosion. We estimate that Belize's mangroves contribute US$111 - $167 million in avoided damages per year. Coral reefs provide $120 - $180 million in protection. These values are independent of one another, and can be added together for a combined value of US$231 - $347 million per year."

Estimated Coral Reef and Mangrove contributions to the economy (million USD)*

	Coral Reefs	Mangroves	Combined Contribution
Tourism	135 - 176	60 - 78	150 - 196
Fisheries	13 – 14	3 – 4	14 – 16
Shoreline Protection	120 - 180	111 - 167	231 - 347

*Mangrove & reef fisheries and tourism values are **not** additive, as they include revenues that rely on both habitats.

Source: Cooper et al. (2009)

Based on their results, the authors concluded that a number of actions were needed in order to preserve the local ecosystems and ensure the country's long-term economic interest. Firstly, they called for increased investments in management, monitoring and compliance, as well as tightening and enforcing fishing regulations. Secondly, they recommended planning and implementing coastal and marine development sensibly, which includes the enforcement of land-use regulations in the coastal zone and the design of sustainable tourism concepts. Thirdly, it was also suggested increasing the support to the country's system of marine protected areas, including the improvement of fee collection.

Influence on Local/Regional Policy-Making

According to an evaluation made by Waite et al. (2014), the valuation study carried out by Cooper et al. (2009) "supported action on multiple fronts, including a landmark Supreme Court ruling to fine a ship owner with an unprecedented and significant sum for a grounding on the Mesoamerican Reef; the government's decision to enact a host of new fisheries regulations (a ban on bottom trawling, the full protection of parrotfish, and the protection of grouper spawning sites); and a successful civil society campaign against offshore oil drilling". Kushner et al. (2012) hold that the valuation results had a high impact on the marine conservation agenda in Belize, and "were used in both policy making and advocacy, with more emphasis on the latter." With regard to the acceptance of the valuation results among stakeholders, they state that "respondents thought that the accuracy of results was important, although they noted that more flexibility may be warranted in certain situations (e.g., a less precise valuation could be fine for educational purposes, but not for legislation). Respondents felt that values should be updated regularly (e.g. every 5 years); they also thought that time series data with future predictions would be helpful, but were concerned about potential inaccuracies. Respondents would also like additional education and awareness about economic valuation, including targeted outreach to key decision makers (tourism, government, and development agencies)".

In order to promote sustainable livelihoods for fishermen and to support bio-ecological stocks and ecosystems, the Belize Fisheries Advisory Board and the Cabinet of Ministers authorized the use of Managed Access at two pilot sites on the coast in 2011. Under this program, fishermen are expected to personally benefit from an increase in their individual production, which should provide the incentive for their commitment to comply and conform to the fisheries management measures (UNDP 2013).

3. Analysis of Case Studies

The above three case studies presented best-practice examples for the use of economic valuation studies in local and/or regional policy-making. As it turned out, the available evidence on the use of valuation results by policy-makers or managers concentrated heavily on a few marine ecosystem services, namely recreational activities and those for the provision of seafood (mainly fish). In the case of Belize, the valuation study also covered coastal protection services.

The case studies reveal that under certain conditions, environmental valuations may provide valid arguments for policy decisions. In the case of Belize and St. Maarten, direct market pricing methods were applied in order to estimate the economic value of the provided ecosystem services. In both cases, the income generated in the tourism and fisheries sector was calculated based on statistical evidence. In addition for Belize, the value of the coastal protections service was estimated by applying a damage cost approach. Both valuation methods (direct market pricing and damage cost) rely on statistical (economical) data and do not make use of so-called stated-preference techniques. As they are based on relatively simple statistical analyses, results of market-based valuations are often easier to communicate to policymakers and relevant stakeholders than non-market valuations. In the case of Belize and St. Maarten, the application of comprehensible and reliable valuation methods might have positively influenced the uptake of the results by policymakers.

In the case of Bonaire, the Contingent Valuation Method has been applied to estimate the willingness to pay for access to the Bonaire National Marine Park (BNMP). As explained in Chapter III, Section 2, CVM introduces hypothetical situations to a (representative) sample of a population, situations which are often presented in a questionnaire in order to elicit willingness to pay or to accept compensation for a contingent product. CVM suffers from a range of methodological uncertainties, such as distance decay effects. However, it is widely applied to estimate the value of goods and services for which no market exists. The Bonaire case study shows that non-market valuations can actually provide the basis for informed decisions – in this case, the increase of access fees to the national park.

The perceived robustness and relevance of the valuation methods, and thus of the valuation results, are certainly a determining factor for the uptake of the research results by policy-makers. However, there are other factors which also play a role, including the political agenda and will over time, the existence and type of link between political decision makers and the source of the study, the perceived lack of results relevance for the policy process, the level of knowledge and experience on environmental valuations by different groups, and the vulnerability of research results to criticism and dismissal (Gerdes et al. 2010). These factors might explain why there is relatively little available evidence on the successful uptake of valuation results in policy processes.

Part VI – Conclusions

The EU overseas entities are well-known hotspots of terrestrial, freshwater, and marine biodiversity. They support unique ecosystems, which are home to an estimated one third of the globally threatened species, including many endemic species. However, their ecosystems are threatened by invasive species, climate change, and habitat loss – the latter of which is often induced by human activities. In addition, biodiversity conservation in ORs/OCTs turns out to be challenging due to the complex jurisdictional matters among the EU, Member States, and overseas territories levels.

With this knowledge, innovative approaches for biodiversity conservation in ORs/OCTs are needed, which incorporate a broad range of actors at local and regional levels. The economic assessment of the environmental change consequences can be a potentially a useful tool to inform environmental policymaking. As this report has shown, there is evidence for the successful uptake of environmental valuations in local and regional policy decisions on ORs/OCTs, and in further relevant countries and territories, such as Belize and Samoa.

The widespread integration of environmental valuations into decision making tools, such as cost-benefit analysis, would be a major step towards better conservation of (sub)tropical biodiversity and ecosystem services. This, however, will not happen overnight and it will require the stakeholders' acceptance of the underlying concepts and methods. The integration of economic arguments into environmental awareness raising campaigns could be a first step on the way to the full acknowledgement of the nature value in political decision making.

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Annex 1 – Overview of Identified Benefit Estimates

Author	Year	Location	Ecosystem	Ecosystem Service	Valuation Method	EUR (2014)	Unit
Aubanel	1993	French Polynesia	Coral reefs	Extreme event prevention	Replacement Cost	5,914.44	ha/yr
Aubanel	1993	French Polynesia	Coral reefs	Tourism / Recreation	Direct market pricing	7,097.33	ha/yr
Bervoets	2010	St. Maarten	Coral reefs	Tourism / Recreation	Direct market pricing	57,645,666.39	year
Bervoets	2010	St. Maarten	Coral reefs	Fisheries	Direct market pricing	1,851,377.99	year
Beukering et al.	2012	Carribean Netherlands	Multiple	TEV	Contingent Valuation, Choice Experiment	19,096,200.00	year
Burke and Maidens	2004	Caribbean	Coral reefs	Fisheries	Direct market pricing	116.00	ha/yr
Burke and Maidens	2004	Caribbean	Coral reefs	Flood prevention	Direct market pricing	551.23	ha/yr
Burke and Maidens	2004	Caribbean	Coral reefs	Tourism / Recreation	Direct market pricing	787.61	ha/yr
Burke and Maidens	2004	Caribbean	Coral reefs	TEV	Direct market pricing	1,443.63	ha/yr
Burke et al.	2008	Saint Lucia	Coral reefs	Fisheries	Direct market pricing	189.47	ha/yr
Burke et al.	2008	Saint Lucia	Coral reefs	Storm protection	Avoided Cost	10,542.43	ha/yr
Burke et al.	2008	Saint Lucia	Coral reefs	Tourism / Recreation	Direct market pricing	49,977.90	ha/yr
Charles	2005	French Polynesia	Coral reefs	Genetic resources	Contingent Valuation	47.32	ha/yr
Charles	2005	French Polynesia	Coral reefs	Fisheries	Direct market pricing	57.73	ha/yr
Charles	2005	French Polynesia	Coral reefs	Fisheries	Direct market pricing	79.50	ha/yr
Charles	2005	French Polynesia	Coral reefs	Climate regulation	Direct market pricing	85.17	ha/yr
Charles	2005	French Polynesia	Coral reefs	Research & Education	no information	110.73	ha/yr
Charles	2005	French Polynesia	Coral reefs	Genetic resources	Benefit Transfer	227.13	ha/yr

Charles	2005	French Polynesia	Coral reefs	Raw materials	Direct market pricing	251.74	ha/yr
Charles	2005	French Polynesia	Coral reefs	Flood prevention	Replacement Cost	1,078.87	ha/yr
Charles	2005	French Polynesia	Coral reefs	Landscape aesthetics	Contingent Valuation	4,731.89	ha/yr
Charles	2005	French Polynesia	Coral reefs	Tourism / Recreation	Direct market pricing	9,766.62	ha/yr
Charles	2005	French Polynesia	Coral reefs	TEV	no information	16,184.01	ha/yr
Chong and Balasubramanian	2003	Caribbean	Coral reefs	Research & Education	Benefit Transfer	35.13	ha/yr
Chong and Balasubramanian		Caribbean	Coral reefs	Tourism / Recreation	Benefit Transfer	1,660.64	ha/yr
Chong and Balasubramanian	2003	Caribbean	Coral reefs	Tourism / Recreation	Benefit Transfer	34,803.92	ha/yr
Conservation International	2008	Turks and Caicos Islands	Coral reefs	TEV	no information	1,041.02	ha/yr
Cooper et al.	2009	Belize	Mangroves	Fisheries	Direct market pricing	22.30	ha/yr
Cooper et al.	2009	Belize	Coral reefs	Fisheries	Direct market pricing	86.02	ha/yr
Cooper et al.	2009	Belize	Mangroves	Tourism / Recreation	Direct market pricing	439.65	ha/yr
Cooper et al.	2009	Belize	Mangroves	Storm protection	Avoided Cost	885.68	ha/yr
Cooper et al.	2009	Belize	Coral reefs	Storm protection	Avoided Cost	955.77	ha/yr
Cooper et al.	2009	Belize	Coral reefs	Tourism / Recreation	Direct market pricing	990.82	ha/yr
Dharmaratne and Strand	2002	Caribbean	Coastal	Nursery service	Factor Income / Production Function	378.19	ha/yr
Dixon et al.	1993	Bonaire	Coral reefs	Tourism / Recreation	Direct market pricing	1,597.00	ha/yr
Eade and Moran	1996	Belize	Tropical forest	Bioprospecting	Benefit Transfer	2,654.73	ha/yr
Eade and Moran	1996	Belize	Tropical forest	Erosion prevention	Benefit Transfer	2,225.70	ha/yr
Eade and Moran	1996	Belize	Tropical forest	Flood prevention	Benefit Transfer	30.13	ha/yr
Eade and Moran	1996	Belize	Tropical forest	Food provision	Benefit Transfer	0.45	ha/yr

Eade and Moran	1996	Belize	Tropical forest	Genetic resources	Benefit Transfer	8.38	ha/yr
Eade and Moran	1996	Belize	Tropical forest	Genetic resources	Benefit Transfer	9.17	ha/yr
Echeverria et al.	1995	Costa Rica	Tropical forest	Tourism / Recreation	Contingent Valuation	330.81	ha/yr
Godoy et al.	1993	Belize	Tropical forest	Genetic resources	Direct market pricing	146.86	ha/yr
Gren and Soderqvist	1994	Fiji Islands	Mangroves	Raw materials	Benefit Transfer	26.99	ha/yr
Gren and Soderqvist	1994	Fiji Islands	Mangroves	Fisheries	Benefit Transfer	202.40	ha/yr
Gren and Soderqvist.		Puerto Rico	Mangroves	TEV	Benefit Transfer	2,991.42	ha/yr
Hamilton and Snedaker	1984	Fiji Islands	Mangroves	Fisheries	Benefit Transfer	1,427.33	ha/yr
Hamilton and Snedaker	1984	Fiji Islands	Mangroves	TEV	Benefit Transfer	2,453.22	ha/yr
Hamilton and Snedaker	1984	Puerto Rico	Mangroves	TEV	Benefit Transfer	3,777.35	ha/yr
Hargreaves-Allen	2008	Belize	Coral reefs	TEV	Contingent Valuation	3,610,514.02	year
Krutilla	1991	Costa Rica	Tropical forest	Tourism / Recreation	Travel Cost	109.10	ha/yr
Lacle et al.	2012	Bonaire	Multiple	Tourism / Recreation	Choice Experiment	2,231,526.77 - 3,001,018.75	all households/year
Lal	1990	Fiji Islands	Mangroves	Nursery service	Direct market pricing	746.34	ha/yr
Marre and Pascal	2012	New Caledonia	Coral reefs	Multiple	Choice Experiment	2,652,250.00 - 4,243,600.00	all households/year
Mathieu et al.	2003	Seychelles	Coastal	Tourism / Recreation	Direct market pricing	24.70	ha/yr
Mohd-Shahwahid and McNally	2001	Samoa	Coral reefs	Cultural values	Benefit Transfer	0.00	ha/yr
Mohd-Shahwahid and McNally	2001	Samoa	Coral reefs	Genetic resources	Benefit Transfer	0.03	ha/yr
Mohd-Shahwahid and McNally	2001	Samoa	Coral reefs	Research & Education	Benefit Transfer	0.05	ha/yr
Mohd-Shahwahid	2001	Samoa	Coral reefs	Raw materials	Benefit Transfer	0.20	ha/yr

and McNally							
Mohd-Shahwahid and McNally	2001	Samoa	Coral reefs	Biological control	Benefit Transfer	0.35	ha/yr
Mohd-Shahwahid and McNally	2001	Samoa	Marine	Biological control	Benefit Transfer	0.35	ha/yr
Mohd-Shahwahid and McNally	2001	Samoa	Coral reefs	Raw materials	Benefit Transfer	0.37	ha/yr
Mohd-Shahwahid and McNally	2001	Samoa	Temperate / boreal forest	Climate regulation	Benefit Transfer	6.19	ha/yr
Mohd-Shahwahid and McNally	2001	Samoa	Temperate / boreal forest	Cultural values	Benefit Transfer	0.07	ha/yr
Mohd-Shahwahid and McNally	2001	Samoa	Marine	Tourism / Recreation	Contingent Valuation	0.50	ha/yr
Mohd-Shahwahid and McNally	2001	Samoa	Temperate / boreal forest	Erosion prevention	Benefit Transfer	0.58	ha/yr
Mohd-Shahwahid and McNally	2001	Samoa	Temperate / boreal forest	Extreme event prevention	Benefit Transfer	0.14	ha/yr
Mohd-Shahwahid and McNally	2001	Samoa	Temperate / boreal forest	Food provision	Benefit Transfer	0.42	ha/yr
Mohd-Shahwahid and McNally	2001	Samoa	Temperate / boreal forest	Genetic resources	Benefit Transfer	2.25	ha/yr
Mohd-Shahwahid and McNally	2001	Samoa	Coral reefs	Climate regulation	Benefit Transfer	2.70	ha/yr
Mohd-Shahwahid and McNally	2001	Samoa	Marine	Climate regulation	Benefit Transfer	2.70	ha/yr
Mohd-Shahwahid and McNally	2001	Samoa	Coral reefs	Waste treatment	Benefit Transfer	4.08	ha/yr
Mohd-Shahwahid and McNally	2001	Samoa	Coral reefs	Nutrient cycling	Benefit Transfer	4.37	ha/yr
Mohd-Shahwahid and McNally	2001	Samoa	Marine	Nutrient cycling	Benefit Transfer	4.37	ha/yr
Mohd-Shahwahid and McNally	2001	Samoa	Coral reefs	Landscape aesthetics	Benefit Transfer	4.57	ha/yr
Mohd-Shahwahid and McNally	2001	Samoa	Marine	Landscape aesthetics	Benefit Transfer	4.57	ha/yr

Mohd-Shahwahid and McNally	2001	Samoa	Marine	Fisheries	Direct market pricing	5.60	ha/yr
Mohd-Shahwahid and McNally	2001	Samoa	Coral reefs	Extreme event prevention	Benefit Transfer	15.85	ha/yr
Mohd-Shahwahid and McNally	2001	Samoa	Temperate / boreal forest	Raw materials	Direct market pricing	1.16	ha/yr
Mohd-Shahwahid and McNally	2001	Samoa	Temperate / boreal forest	Raw materials	Direct market pricing	1.30	ha/yr
Mohd-Shahwahid and McNally	2001	Samoa	Temperate / boreal forest	Raw materials	Benefit Transfer	4.36	ha/yr
Mohd-Shahwahid and McNally	2001	Samoa	Temperate / boreal forest	Tourism / Recreation	Contingent Valuation	0.99	ha/yr
Mohd	2001	Samoa	Temperate / boreal forest	Water provision	Benefit Transfer	0.78	ha/yr
Mohd	2001	Samoa	Temperate / boreal forest	Water regulation	Benefit Transfer	1.76	ha/yr
Naylor and Drew	1998	Micronesia	Mangroves	Fisheries	Benefit Transfer	134.39	ha/yr
Naylor and Drew	1998	Micronesia	Mangroves	Raw materials	Benefit Transfer	220.16	ha/yr
Naylor and Drew	1998	Micronesia	Mangroves	Food provision	Direct market pricing	434.79	ha/yr
Naylor and Drew	1998	Micronesia	Mangroves	TEV	no information	658.51	ha/yr
Naylor and Drew	1998	Micronesia	Mangroves	Genetic resources	Contingent Valuation	893.30	ha/yr
Naylor and Drew	1998	Micronesia	Mangroves	Storm protection	Contingent Valuation	1,972.47	ha/yr
Pagiola et al.	2004	Costa Rica	Tropical forest	Genetic resources	no information	43.86	ha/yr
Pascal	2010	New Caledonia	Coral reefs	Fisheries	Factor Income / Production Function	58,288,300.46 - 81,141,464.21	year
Pascal	2010	New Caledonia	Coral reefs	Tourism / Recreation	Factor Income / Production Function	26,739,050.45 - 31,851,066.45	year
Pascal	2010	New Caledonia	Coral reefs	Flood prevention	Factor Income / Production Function	129,479,253.83 - 246,980,167.59	year
Pascal	2010	New Caledonia	Coral reefs	Bioprospecting	Factor Income / Production Function	660,223.47 - 4,913,948.95	year
Pascal	2010	New Caledonia	Coral reefs	Research & Education	Factor Income / Production Function	3,197,367.94 - 3,914,181.99	year
Pendleton	1995	Carribean	Coral reefs	Tourism /	Travel Cost	7,065.27	ha/yr

		Netherlands		Recreation			
Raboteur and Rhodes	2006	Guadeloupe	Coral reefs	Genetic resources	Contingent Valuation	70.98	ha/yr
Rausser and Small	2000	New Caledonia	Tropical forest	Bioprospecting	Factor Income / Production Function	1,040.78	ha/yr
Ricketts et al.	2004	Costa Rica	Tropical forest	Pollination	Direct market pricing	129.09	ha/yr
Schep et al.	2013	Bonaire	Multiple	Tourism / Recreation	Direct market pricing, Choice Experiment	37,353,980.00	year
Schep et al.		Bonaire	Coral reefs	Fisheries	Direct market pricing	307,796.80	year
Schep et al.	2012	Bonaire	Coral reefs	Tourism / Recreation	Direct market pricing, Choice Experiment	538,644.39	year
Shultz et al.	1998	Costa Rica	Tropical forest	Tourism / Recreation	Contingent Valuation	1,785.55	ha/yr
Spash	2000	Curacao	Coral reefs	Tourism / Recreation	Contingent Valuation	60.34	year
Thur	2010	Bonaire	Coral reefs	Tourism / Recreation	Contingent Valuation	63.08 - 138.57	year
Tobias and Mendelsohn	1991	Costa Rica	Tropical forest	Tourism / Recreation	Travel Cost	78.73	ha/yr
Trejo	2005	Belize	Marine	Tourism / Recreation	Contingent Valuation	10.18	visit
Uyarra et al.	2010	Bonaire	Coral reefs	Tourism / Recreation	Contingent Valuation	29.01	year
Zanten and van Beukering	2012	Bonaire	Coral reefs	Flood prevention	Damage cost approach	25,720.27 - 53,951.39	year