VALUING NET BENEFITS OF BIODIVERSITY CONSERVATION IN WEST AFRICAN MARINE PROTECTED AREAS

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The thesis is submitted in partial fulfilment of the requirements for the award of the degree of Doctor of Philosophy (PhD) of the University of Portsmouth.

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Abstract

Momentum behind the economic valuation of ecosystems, after a decade of hopeful support from researchers and policymakers, is currently petering out and decision-makers still do not consider biodiversity conservation to be a political priority. Surprisingly, the economic benefits provided by the conservation of ecosystems have been poorly investigated, unlike the ecosystems themselves. Furthermore, is the valuation of conservation (the valuation of the “interest rate” made on the natural capital saved, instead of the valuation of the natural capital itself) an efficient means to better serve decision-making? The research presented here addresses this question, in proposing a more effective approach to the valuation of conservation. It also investigates how such economic valuation exercises could best serve the decision-making process.

The research method for measuring conservation value relies on a comparison of Total Economic Values for analogous ecosystems both within a protected area and in outside adjacent areas. This methodology is tested in a sample of five marine protected areas in West Africa. For the estimation of the Total Economic Values in these sites, the research has applied most of the available valuation tools and includes all values for which data are available, including non-use values.

The results indicate a predominance of benefits linked to indirect use values over direct use values and non-use values. The marine protected areas display substantial benefits when compared to unprotected sites. These benefits are thought to derive primarily from the better marine health status associated with protected areas, and subsequent higher indirect use values which compensate for the decrease in direct use values caused by the conservation policy and the subsequent limitations imposed as a result. The ‘paper areas’ (i.e. those protected areas with no management plan) show, however, a deficit even when compared to unprotected sites.

The research discusses and highlights the shortcomings of such an approach within the West African context (data-poor situation, non-monetised economies, value transfer to developing countries, difficulties in communicating non-use values of biodiversity) and associated time and space considerations. It also underlines the importance of considering the socio-cultural context in any economic valuation, which provides key information for valuation interpretation.
Furthering the approach within the ‘economics of protection’ stream (after the ‘economics of degradation’ and the ‘economics of welfare’), this research delivers a new approach for valuing biodiversity conservation. The extensions of this research for policy purposes may include management support (comparison of conservation benefits with costs of management, increased consideration of indirect use values), advocacy information (through the calculation of the costs of policy inaction), and mechanisms for sustainable financing (through the development of payment for ecosystem services).
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<tr>
<td>BT</td>
<td>Benefit transfer</td>
</tr>
<tr>
<td>CA</td>
<td>Comparison Areas</td>
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<tr>
<td>CBA</td>
<td>Cost-Benefit Analysis</td>
</tr>
<tr>
<td>CEA</td>
<td>Cost-Effectiveness Analysis</td>
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<tr>
<td>CEMARE</td>
<td>Centre for the Economics and Management of Aquatic Resources</td>
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<tr>
<td>CFA Franc</td>
<td>Franc des Colonies Françaises d’Afrique</td>
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<tr>
<td>CO₂</td>
<td>Carbon Dioxide</td>
</tr>
<tr>
<td>COPI</td>
<td>Costs of Policy Inaction</td>
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<tr>
<td>CVE</td>
<td>Cape Verdean Escudo</td>
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<tr>
<td>CMRP</td>
<td>Coastal and Marine Regional Programme</td>
</tr>
<tr>
<td>DC</td>
<td>Discounting Coefficient</td>
</tr>
<tr>
<td>EC</td>
<td>European Commission</td>
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<tr>
<td>EU</td>
<td>European Union</td>
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<tr>
<td>GB</td>
<td>Guinea-Bissau</td>
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<tr>
<td>GDP</td>
<td>Gross Domestic Product</td>
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<tr>
<td>GIS</td>
<td>Geographical Information System</td>
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<tr>
<td>GVA</td>
<td>Growth Added-Value</td>
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<tr>
<td>HPM</td>
<td>Hedonic Price Method</td>
</tr>
<tr>
<td>IBAP</td>
<td>Biodiversity and Protected Areas Institute</td>
</tr>
<tr>
<td>IC</td>
<td>Intermediary Consumption Costs</td>
</tr>
<tr>
<td>INDP</td>
<td>National Institute for the Development of Fisheries (in Cape Verde)</td>
</tr>
<tr>
<td>IPBES</td>
<td>Intergovernmental Policy-Science Platform for Biodiversity and Ecosystem Services</td>
</tr>
<tr>
<td>IPCC</td>
<td>Intergovernmental Panel on Climate Change</td>
</tr>
<tr>
<td>IUCN</td>
<td>International Union for the Conservation of Nature</td>
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<tr>
<td>MCA</td>
<td>Multi-Criteria Analysis</td>
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<tr>
<td>MCE</td>
<td>Marine and Coastal Ecosystems</td>
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<tr>
<td>MPA</td>
<td>Marine Protected Areas</td>
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<td>MSY</td>
<td>Maximum Sustainable Yield</td>
</tr>
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<td>NGO</td>
<td>Non-Governmental Organisation</td>
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<tr>
<td>NPV</td>
<td>Net Present Value</td>
</tr>
<tr>
<td>OECD</td>
<td>Organisation for Economic Co-operation and Development</td>
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<tr>
<td>PES</td>
<td>Payments for Ecosystem Services</td>
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<tr>
<td>REDD+</td>
<td>Reducing Emissions from Deforestation and Forest Degradation</td>
</tr>
<tr>
<td>TCM</td>
<td>Travel Cost Method</td>
</tr>
<tr>
<td>TEEB</td>
<td>The Economics of Ecosystems and Biodiversity</td>
</tr>
<tr>
<td>TEV</td>
<td>Total Economic Value</td>
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<tr>
<td>UNEP</td>
<td>United Nations Environment Programme</td>
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<tr>
<td>UNEP-WCMC</td>
<td>UNEP’s World Conservation Monitoring Centre</td>
</tr>
<tr>
<td>WTP</td>
<td>Willingness-To-Pay</td>
</tr>
<tr>
<td>WTR</td>
<td>Willingness-To-Receive</td>
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Declaration

Whilst registered as a candidate for the above degree, I have not been registered for any other research award. The results and conclusions embodied in this thesis are the work of the named candidate and have not been submitted for any other academic award.

The word count of the document is 69,547 words excluding ancillary data such as footnotes, bibliographies, references and appendixes.

Thomas Binet, candidate
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1 Chapter 1: introduction

1.1 Context
Public and economic policy makers have long considered nature as *res nullius*, something that has no owner. The failure to safeguard natural ecosystems is caused by this non-recognition of the value of nature (Pagiola et al., 2004). The valuation of ecosystems aims to counterbalance this and assign a monetary value to nature in general and ecosystem services more particularly. It rests on a double weakness in current policy-making, which neither gives such services their full economic weight nor accounts sufficiently for environmental damage caused by human activity. Setting monetary values for ecosystem services and for anthropogenic degradation of the environment helps create market-based mechanisms to pay for such services (or to compensate for such damages). Environmental economists currently believe this approach, which situates biodiversity within mainstream economics and public policy making to enable more efficient spending, represents the only way to curb biodiversity loss (Binet et al., 2011).

The first marine and coastal economic valuation took place in 1926, when a specialist in fisheries biology, Percy Viosca, estimated the conservation value of Louisiana’s coastal wetlands (Viosca, 1928). Recently, accidental marine pollution incidents have increased the need for such valuation: following the 1989 Exxon Valdez oil tanker spill in Alaska, the American Supreme Court fined Exxon over $1 billion in its final court judgment in 2008 for ecological losses and compensatory damages. Ecosystem valuations are currently being used to estimate the 2010 Deepwater Horizon oil spill impacts on coastal ecosystems in the Gulf of Mexico.

Hence, through the valuation of the socioeconomic utility of ecosystems, valuation exercises have aimed to uncover the “value of Earth”, the natural capital (Costanza et al., 1997) or the total annual value of flows from an ecosystem, when considered over one year only (Pagiola

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1Ecosystem services are defined as those functions of ecosystems that (directly or indirectly) support human welfare. They occur at multiple scales, from climate regulation and carbon sequestration at the global scale, to flood protection, soil formation, and nutrient cycling at the local and regional scales (Boumans and Costanza, 2007). They are divided into four categories: provisioning, regulating, supporting and cultural (see MEA, 2005 for more details)

2The loss estimates were determined through an economic analysis, with economic losses directly due to the accident surpassed by non-market losses. The less well-known 1978 Amoco Cadiz case established the need to measure the cost of ecological damage, but demands for indemnities based on economic valuations were abandoned during litigation.
et al., 2004). During the 1990s, such valuations estimated large figures using a broad scale approach, when a team of researchers led by Robert Costanza estimated the economic value of the entire world’s ecosystem services. They calculated that ecosystem services contributed $21 trillion dollars annually to human well-being; most (60%) of these services were concentrated along coastlines that make up only 9% of the world’s surface area (Costanza 1999). These coastal and marine areas – including coastal wetlands and mangroves – represent 77% of the world’s total ecosystem services value (Martinez et al. 2007). Following this approach, a lot of studies have valued marine ecosystem services in various regions of the World. In the Mediterranean, for instance, marine ecosystems are estimated to be worth nearly 26 billion euros annually, with cultural and leisure services providing two-thirds of that total (Mangos et al. 2010). In the United Kingdom, provisioning services are estimated to be worth 713 million euros, cultural services 15 billion euros, regulating services between 840 million euros to 10 billion euros, while supporting services exceed 1 trillion euros in value (Beaumont et al. 2008). In each of these valuations, the estimated worth of “commercial” goods and services is less than that of cultural, supporting and regulating services.

1.2 Research questions and objectives of the research
Although economic valuation of ecosystems has been increasingly undertaken worldwide, it has been shown to be of limited use for decision-making. Laurans et al. (2013) have estimated that only 2% of the published papers on ecosystem services valuation have clearly influenced the outcome of a policy decision. This observation was backed by a World Resource Institute report on the use of coral reef economic valuations in the Caribbean: out of 200 valuation exercises undertaken on coral reef ecosystems in the Caribbean, only 13 of them have actually influenced marine and coastal management policies (Kushner et al., 2012).

What are the reasons for such poor use of economic valuation of ecosystems within decision-making? These may include the fact that the figures produced are often considered to be too large to be realistic, the values provided are intangible when compared to real money born by other sectors, or the biases related to the methods used (Binet et al., 2011). But it may also be the object of the valuation that is not relevant to decision-making needs. Hence, as Pagiola et al. (2004) noted (p. 18): "estimates of the total annual flow of benefits from an ecosystem have frequently been used to justify spending to address threats or to
improve its condition. But using such value estimates in this way would be a mistake. To examine the consequences of ecosystem degradation, or to assess the benefits of a conservation intervention, it is not enough to know the total flow of benefits. Rather, what is needed is information on how that flow of benefits would change.”

Following that route of questioning the object of valuation, it is surprising to note that the economic benefits of protection per se (i.e. the actual benefits brought by the action of conservation) have been poorly investigated. It is even truer in the marine environment, where comprehensive examples of economic valuation of benefits provided by marine protected areas (MPA) are scarce. When the object of a valuation is a specific area’s protection along with its associated benefits, most studies confine the valuation to an estimate of the willingness-to-pay (through contingent valuation method) to ensure that some elements of the protected area are conserved or managed: for example, conservation of endangered species (Bandara and Tisdell, 2004), recovery of fish stocks (Ojea and Loureiro, 2010), improvement of the quality of ecosystem (Bhat, 2003), and protection of recreational assets (Ahmed et al., 2007).

Building on these observations, one can wonder if extending the valuation exercises to conservation benefits more broadly would not be more useful to decision-makers than limiting the valuation to individual ecosystem services. Also, how is it possible to assess all benefits associated with the protection, and not only specific benefits of the action of conservation? As a consequence, the research questions that channel my study are twofold: how is it possible to measure the total value of marine conservation? How can this help decision-making in conservation?

In line with these questions, the objectives of this thesis are to develop a method based on the total flows of values from ecosystems to estimate the total benefits of conservation and provide results that are able to feed into decision-making in a more efficient way. For this, the concept of the Total Economic Value (TEV) is applied. This concept enables not only the direct use values (associated to merchant uses) to be considered, but also the indirect use values (related to ecological functions such as coastal protection, water treatment) and the non-use values (related to the attachment to ecosystems, independently of their uses, namely the existence and bequest values). As a consequence of this conceptual choice of using the TEV, the specific objectives include:
- The valuation of direct use values in MPAs;
- The valuation of indirect use values in MPAs;
- The valuation of non-use values in MPAs;
- The valuation of the Total Economic Value and subsequent benefits of conservation, through the comparison between protected and unprotected sites; and
- The development of potential uses of these valuation exercises for decision-making.

1.3 Content

This thesis presents the methods, tools and results of a study developed in order to address these questions in the specific case of marine conservation through the implementation of MPAs.

The methodological chapter provides a literature review relating to the need for sound coastal management and protection of marine biodiversity, the method and tools employed in this research. The review first introduces the context of sustainable management of marine resources and the protection of coastal and marine biodiversity, with an emphasis on Marine Protected Areas. It then provides a review on the concept of value of nature within the history of economics, the concept of ecosystem services, and recent history of environmental economics in relation to the economic valuation of ecosystems. It provides a background to the conceptual framework of value and especially the Total Economic Value (TEV), and then presents the various tools used for environmental value measurement. Finally, this chapter examines the various methods adopted in the literature to value conservation and the protection of ecosystems.

The third chapter introduces our case study. It presents the ecosystems studied in West Africa as part of this research, their current health status, and the ecological functions they provide or contribute to. It then presents the MPAs that will be scrutinized. For each MPA, details about the ecosystems found in these MPAs are provided, along with all relevant information on these MPAs: details about their creation, their institutional status and management authorities, the categories of populations (autochtons and allochtons\(^3\)), the main economic activities and their surface area. It also exposes the survey methods: the

\(^3\)
categories of populations interviewed (both users and non-users), the sampling plan for the survey and the questionnaire used and the socioeconomic profiles of respondents.

The fourth chapter presents the specific method developed and the results of the valuation exercise for the direct use values for the sample of MPAs and their related areas of comparison. It also provides a critique of the findings and proposes some improvements to the methodology.

The fifth chapter presents the specific method developed and the results of the valuation exercise for the indirect use values for the sample of MPAs and their related comparison areas. It also provides a critique of the research and discusses the shortcomings of the approach adopted as evidenced via the research.

The sixth chapter presents the specific method developed and the results of the valuation exercise for the non-use use values for the sample of MPAs and their related comparison areas. It also provides a critique of the findings.

The seventh chapter presents the specific method developed and the results of the valuation of the TEV and the estimates of conservation benefits brought by the MPAs of the study. This chapter also presents a discussion section with the critique of the findings on the TEV and conservation benefits, along with a broader discussion on the importance of the prevalent socio-cultural context. It focuses on the perceptions held of marine ecosystems by the local populations, as well as the willingness of inhabitants to see ecosystem management improved and space and time considerations as part of the economic valuation. This discussion section also discusses time and geographical aspects of the valuation exercise.

The results presented through the four chapters hence revolve around the components of the Total Economic Value, as shown in Figure 1-1.
The last chapter develops some key policy considerations with regards to the economic valuation of conservation benefits. It proposes some applications of the economic valuation for public decision-making: i) by comparing the economic value with management costs; ii) by focusing on indirect use values for management; iii) by highlighting the costs of policy inaction; or iv) by developing innovative financing mechanisms for biodiversity conservation.
2  Chapter 2: a framework for the economic valuation of marine protected ecosystems (literature review)

The objectives of this chapter are to: present the pressures on the coastal resources and the need for their sustainable management; provide an understanding of the economic valuation of ecosystems services; identify methods that are employed for such valuation; and establish the approach to be adopted for the valuation of West African Marine Protected Areas (MPAs). To this end, the chapter first provides a brief history of the economic valuation of ecosystem services and the development of this approach. Second, it explains the concepts and methods used in the economic valuation of ecosystems. Third, it offers an overview of the three main approaches to the economic valuation of ecosystems and, finally, provides more details on one of these approaches: the economics of protection.

2.1  Marine and coastal conservation context

The need for tools to promote conservation of marine and coastal ecosystems has materialized with the increasing pressure born on these ecosystems and the exceptionally high rate of loss of marine and coastal biodiversity. This section presents an overview of the threats to marine and coastal ecosystems and how this has led to the development of management approaches such as marine protected areas and integrated coastal zone management. It also provides an overview of recent progress on these new approaches to marine and coastal biodiversity conservation.

2.1.1  Threats to coastal and marine ecosystems

The threats to marine and coastal ecosystems have both natural and human origins. Natural threats include erosion by waves, extreme events and naturally-occurring climate change. Human threats are more diverse. They include coastal building, agricultural development, fisheries and aquaculture, tourism and recreational activities, development of industries and ports, marine transport, and anthropogenic carbon emissions (Cummins et al., 2004). These threats have led to a number of factors impacting the coastal and marine ecosystems, including habitat conversion, exploitation of natural resources, pollution, invasive species development, sedimentation, sea level rise and ocean acidification. First, habitat conversion occurs on coastal ecosystems as a result of building or development projects. This includes conversion of wetlands and salt marshes for construction, mangrove deforestation for
shrimp farming development or dredging of waterways (Woodard, 2000; WRI, 2001). These threats account for the majority of losses of ecosystem surface area.

Second, resource extraction may remove only a small proportion of the habitat but change the whole ecosystem dynamics. If these resources contribute to the physical structure, the habitat may lose its ability to support ecosystem services such as the provision of nursery habitat (de Groot, 1992). Mangrove wood-cutting is an example of such resource extraction that jointly affects the ecosystem and the habitat structure. Some destructive fishing practices (e.g. blast fishing, bottom trawling) may also cause irreversible habitat loss (Agardy, 1997a; Chambers, 1991; Dayton et al., 2000). Mining and dredging may also affect the habitat structure. More generally, however, resource extraction can impact on the ecosystem balance through its effects on food webs, including the notable "fishing down marine food webs" effect (Pauly et al., 1998), and other cascading effects by keystone species removal (Myers and Worm, 2003). This can indirectly lead to ecosystem loss, if, for instance, unsustainable fishing practices deplete certain fish stocks.

Third, pollution mostly originates from industry, agriculture or domestic sources on land. Pollution can have a serious impact when releases of sewage and waste into coastal ecosystems directly increase microbial activity as a result of increased levels of organic matter in the environment. This in turn depletes oxygen in the water column and can lead to the development of 'dead zones' in coastal waters. In other places, this artificial enrichment of coastal waters (i.e. eutrophication) causes outbreaks of harmful algal blooms, which have harmful consequences for all marine life.

Fourth, the introduction of alien invasive species is usually caused by human activity: for example, water taken onboard a ship as ballast and dumped in another region, intentional or accidental releases of alien species by aquaculture or by aquariums or even individuals, or overexploitation of one species which can create an opportunity for the invasion of an ecosystem by another species (Molnar et al., 2008). The spread of such species can have severe impacts on the ecosystems and native species.

Fifth, sedimentation is another form of pollution caused by run-off from land. Sedimentation can greatly alter coastal ecosystems by increasing turbidity, decreasing light penetration and causing the death of filter-feeding organisms (Burke et al., 2002).
Sixth, anthropogenic carbon emissions caused by the burning of fossil fuels, cement production and deforestation are having a major impact on the coasts and oceans (Bijma et al., 2013). The subsequent increase of CO₂ concentrations in the atmosphere causes a warming of the ocean temperature (Rayner et al., 2003; IPCC, 2007; Belkin, 2009; Sherman et al., 2009; Reid and Beaugrand, 2012). Another direct impact of raised atmospheric CO₂ levels is ocean acidification, through its entry into marine surface waters from the atmosphere and its chemical reaction with water to form carbonic acid (Caldeira and Wickett, 2003; Caldeira, 2007; Cao and Caldeira, 2008). Along with these direct impacts, there are many indirect and cascading impacts of anthropogenic climate change on coasts and oceans, including (Bijma et al., 2013): sea level rise, the increase of surface ocean stratification, changes to wind and currents, decrease of surface oxygen concentrations, changes to thermohaline circulation. These oceanographic impacts will necessarily have biological impacts (e.g. changes in primary productivity, range shifts and species invasion, redistribution of commercial fish stocks).

2.1.2 The efforts for enhanced coastal and ocean protection
Concerns about the degradation of marine and coastal resources have been expressed for a long time (Garcia and Boncoeur, 2004): overfishing has been recognized and described for centuries (Tiphaigne de la Roche, 1760; Pauly and Chua, 1988) and since Warming (1911), and Graham (1935), it has been clear that fishery resources could be depleted by human activities (Garcia and Boncoeur, 2004). Until three decades ago, the main challenge for coastal ecosystem protection efforts were primarily concerned with the overexploitation of fish stocks, where no management measures had been implemented. Along with overfishing, consciousness about the degradation of marine habitats only became apparent following major oil spills.

It is only at the end of the 20th century, with the accelerating degradation of marine and coastal ecosystems, that threats to these ecosystems began to be seriously considered. This happened thanks to international events (in the first instance the UN Conference on Environment and Development – known as Earth Summit – in Rio in 1992, and taken forward in Johannesburg in 2002 with specific targets and timetables – embodied within the Johannesburg Plan of Implementation -JPOI). In particular, the Rio summit enabled an agreement to be reached on the Convention for Biological Diversity and Agenda 21; which highlighted the need for coastal and marine ecosystems protection.
Efforts for the protection of coasts and oceans since 1992 have focused primarily on fisheries management. The research developed ecosystem approaches to fisheries management, which considered not only targeted species but the whole ecosystem to which these species belonged.

Efforts have also been put into the development of Integrated Coastal Zone Management (ICZM), which was first introduced on the international policy agenda at the Rio World Summit in 1992 (Chouinard et al., 2011). Historically, the concept of coastal zone management first originated in response to numerous threats from human activities and conflicts between different uses that appeared as a consequence of these threats. The concept was first introduced formally in the 1970s when coastal zone management was identified as a priority (Chouinard et al., 2011). The word ‘integrated’ was added in the 1980s when it became evident that sectoral and individual disciplinary approaches to coastal zone management (fisheries management in the first instance) would not solve the environmental challenges that the coastal ecosystems faced. Multi and cross-disciplinary approaches were then perceived as the only way forward (Blanchard and Vanderlinden, 2010). Another rationale for the adoption of ICZM was the need to reconcile environmental approaches with social and economic ones. ICZM rapidly gained interest after the Rio Conference. In 2002, 700 ICZM projects were recorded all over the World (Belfiore, 2003).

Efforts also converged towards the development of Marine Protected Areas (MPAs). They were seen as a powerful tool for fish stocks recovery, along with being a necessary instrument to halt marine and coastal biodiversity loss and habitat destruction. For this reason, interest in MPAs dramatically increased after 1992 (Hoagland et al., 1995; Conover et al., 2000; Alban et al., 2008). Until then, MPAs were very few: De Silva listed a total of 430 marine protected areas created by 1985, most of them covering a small area (De Silva et al., 1986). However, the rate of MPA designation quickly increased after 1992, and at the end of the century most countries had implemented some form of MPA (Kelleher et al. 1995).

The emergence of new tools for marine conservation and coastal management (such as MPAs and ICZM approaches) occurred in line with improvements in data collection and knowledge about coastal and marine processes (Sala and Knowlton, 2006; Cummins et al., 2004; Chouinard et al., 2011). Global assessments were undertaken, foremost amongst these being the “Assessment of Assessments (AoA),” a scientific evaluation process for the oceans, similar to the Regular Process for climate reporting through the Intergovernmental
Panel on Climate Change (IPCC) and the Census of Marine Life that recognized implicitly that no country in the world could meet its obligations to catalogue marine species under the Convention on Biological Diversity (CBD) (National Research Council, 1995).

2.1.3 The urgent need for sustainable management of marine and coastal ecosystems

In spite of these increased efforts to promote conservation and gain knowledge about coastal and marine ecosystems, it is recognized that international policy objectives have not succeeded in reaching their targets (Cicin-Sain et al., 2011). The tools and actions to scale up coastal and marine ecosystem protection have been poorly implemented and as a result these ecosystems have continued to degrade, at an increasing rate (Klinger, 2004).

Hence, for the past 10 years, fisheries worldwide have been generally reported as being in an extremely poor state, with almost no improvement in sight (e.g., Pitcher, 2001; Pauly et al., 2002). Marine fisheries catches have not increased since the 1980s (FAO, 2012) and there is evidence of episodes of serial depletions by location and depth (e.g., Morato et al., 2006; Swartz et al., 2010).

Marine and coastal ecosystems collapse as a result of past and current stressors (i.e. pollutants, run-off, sediment loads, and overexploitation of natural resources). Yet, these stressors are likely to have ever greater amplitude (Rogers and Laffoley, 2011): potential increase of harmful algal blooms in recent decades (Van Dolah, 2000; Landsberg, 2002; Wang and Wu, 2009), ‘dead zones’ (Rabalais et al., 2002; Diaz and Rosenberg, 2008), changes of ecosystems toward the benefits of planktonic organisms – including jellyfish (Mills, 2001; Boero et al., 2008), dramatic changes in microbial communities with substantial impacts at the ecosystem scale (Jackson, 2010), and the impact of emerging contaminants on ecosystems (La Farré et al., 2008). About invasive species, only 16% of marine ecoregions have no reported marine invasions, although the true figure should surely be inflated due to under-reporting (Molnar et al., 2008). Roger and Laffoley recently concluded that the observed impairment damages or eliminates the ability of ecosystem to support humans (Rogers and Laffoley, 2011). This observation is incompatible with the fact

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4 The “Oceans at Rio+20” Report (Cicin-Sain et al., 2011) provides more details about the assessment of action towards the achievement of objectives set in Rio
that population density on the coast has rocketed and in 2015, it is thought that more than 60% of the World population will live within 30km of a coast (Rosenthal et al., 2011).

Although the human-induced pressures of overexploitation and habitat destruction are the main causes of recently observed extinctions (Dulvy et al., 2009; Bijma et al., 2013) climate change is increasingly adding to this. In addition, and as a reciprocal principle, these threats impede the capacity of ecosystems to adapt to climate change.

To face these increasing concerns about marine and coastal ecosystems sustainability, international networks in favour of greater ocean protection have called for strengthened efforts towards increased conservation and management efforts, which include (Rogers and Laffoley, 2011): immediate reduction in CO₂ emissions, urgent actions to restore the structure and function of marine ecosystems, proper and universal implementation of the precautionary principle and urgent introduction by the UN of effective governance of the high seas.

In parallel, researchers have aimed to raise concerns among policymakers and the public about the consequences of coastal and marine ecosystems destruction. In doing so, economists have aimed to put a value of those ecosystems and estimate the costs associated to their losses. The following provides some insight into this approach.

2.1.4 MPA as a powerful management tool to curb biodiversity loss
As noted above, the development of the MPA approach has thrived since the 1990s. However, the objectives of associated with the designation of protected areas worldwide are still far from being reached. Also, a substantial proportion of MPAs created still lack effective management and enforcement of measures, while other MPAs have only been declared officially without any practical consequences – they remain “paper MPAs”.

According to Agardy (1997b), effective marine biodiversity conservation should implement the three following measures:

- Maintain ecological processes and protect species, populations and threatened habitats;
- Set sustainable exploitation levels in order to control and reduce catches if appropriate; and
- Ensure a fair and effective distribution of protection benefits.
MPAs have the potential to be a key tool to implement these three measures. First, they have proved effective on fish abundance, size and biomass (Harmelin-Vivien et al., 2008; Halpern, 2003). A literature review by Halpern and Warner (2002) concludes that marine reserves lead to a significant and sustainable increase in species density, biomass and diversity over a period of one to three years. This is however not verified for long-living species and in some sites where illegal fishing occurs (Ibid).

Fish recruitment is also known to increase within an MPA (Robert and Hawkins, 2000). Specific diversity also increases with the existence of an MPA, especially in highly exploited areas (NRC, 2001). Halpern (2003) has found that 59% of the reserves studied have seen an increase in their species diversity of about one third. MPAs also have an effect on threatened habitats, through the ban on destructive fishing practices for instance.

MPAs have also been recognized as an effective tool to maintain localized fish populations. The general concept is to create an area where the local stock of fish can thrive and create a localized surplus population. When this surplus occurs, the extra fish expands into the surrounding areas. This expansion is known as the “spillover effect”. This positively impacts commercial fishermen in surrounding areas. Several studies have shown that fish migrate outside the MPAs borders (Forcada et al., 2008; Guidetti, 2007, Abesamis and Russ, 2005; Cole et al., 2000; Tupper, 2007) and an increase of fishing catch per unit of effort outside an MPA (McClanahan and Mangi, 2000). Roberts et al. (2001) demonstrated in Saint Lucia that five years after the creation of a network of small reserves, the adjacent catches of small-scale fishers had risen from 46% to 90%, depending on the fishing equipment considered.

As a second measure to effective marine biodiversity conservation cited by Agardy (1997b), MPAs are also an important tool for fisheries management (through the setting and respect of sustainable exploitation levels). This is particularly true for sedentary reef species that are mostly overexploited and for which the fishing pressure is difficult to control (NRC, 2001).

Furthermore, it can promote better collaboration and participation of local communities in management, and ensure that local users share the benefits of such biodiversity protection. This is particularly the case in MPAs that are managed by the local populations through community-based management. The local users hence take part in the implementation of the management plan and the enforcement of such measures is then easier.
As of 2010, the world hosted around 6,000 MPAs, encompassing more than 1% of the world's oceans, according to the World Database on Protected Areas (WDPA). According to the IUCN and United Nations Environment Programme's World Conservation Monitoring Centre (UNEP-WCMC), the total number of MPAs is around 5,000 for a total coverage of 2.85 million km² - 0.8% of the World's oceans and 2% of the waters under national jurisdiction. There are, however, additional efforts required to establish more MPAs worldwide, along with a better enforcement of the management measures in these MPAs. This would however require significant political willingness from local, regional and national decision-makers, and possibly better communication on these benefits by researchers and MPA managers.

2.2 A history of the economics of ecosystems

2.2.1 The economic value of ecosystem goods and services (1500-1990)

2.2.1.1 General context: about value and utility

The term “value” that interests us can have several meanings according to the fields of philosophy, economics and sociology. The values which citizens or decision-makers refer to when they make their choices may be justified in various ways. Back in the 16th century, Galiani defined value as the subjective relationship of equivalence between goods (Galiani, 1787). He notes that this depends on utility and rareness. From the end of the 18th century, the “classical” economists began to emphasize labour as the major force driving the production of wealth. Value was then a facet of labour. The current meaning of economic value dates back to the end of the 18th century and stems from the utilitarian philosophy of J. Bentham. Bentham proposed assessing individual and public behaviour on the basis of their contribution to the achievement of “greatest happiness to the greatest number”, that is to say, their social utility.

5 See the online interface of the WDPA on http://www.protectedplanet.net/
7 Several authors have proposed clarification on the issue of justification in the field of environment; see for instance Godard (2004).
8 The starting point of Benthamite theory is that "ethical good" is a observable and demonstrable reality which can be defined based on the basic motivations of human nature: the "natural" trend to pursue happiness, that is to say, the maximum of happiness for a minimum of sorrow (Bentham, 1787).
Utilitarianism as introduced by Bentham can be characterized by a set of principles:

- the "good" is defined as the well-being;
- actions are judged according to their consequences, and not upon the willingness of agents (consequentialism);
- the value of an action is the net welfare sum regardless of its distribution; and
- individuals are substitutable (impartiality and universalism).

One important feature of utilitarianism is therefore its rationality. The value of an act is not determined by the principles of intrinsic value anymore. Instead, pleasure is the sum of the effects of an action on the well-being of all. It assumes the capacity of economists to measure these effects and to evaluate their impact on the well-being of individuals. John Stuart Mill later introduced the concept of indirect utilitarianism to economic analysis, for which pleasure is a means of achieving welfare for the greatest number. After Mill, for economists, welfare (and value associated) would not differ from social utility.

At the end of the 19th century the "neoclassicals" (Jevons, Menger, Walras, & Marshall) transformed the utility maximization approach with their marginal approach. It is utility provided by the unit gained (or lost) that guides choice (and sets prices). However, the main barrier to this approach is that economists find it difficult to define an objective scale to measure utility. At the beginning of the 20th century, the concept of New Welfare Economics clearly distinguished the issues related to the effective distribution of income and considers them separately. Utility is considered as an ordinal measure that does not allow for direct comparisons between individuals; and issues related to effectiveness are assessed against the Pareto criteria and the Hicks-Kaldor compensation test (Chevassus-au-Louis et al., 2009). In other words, the question of value becomes less a measure than a comparison. This approach therefore is more general than utilitarianism, for which social welfare is the simple sum of individual utilities. It leads to the core of neoclassical welfare economics, which sees useful goods as having value (that contribute to well-being) and are rare from an economic perspective (goods for which demand exceeds supply).

Later, alternative theories developed, mostly as critiques of the neoclassical welfare approach. They introduced the concept of freedom of choice. R. Nozick therefore considered that an outcome was fair if the process that led to this outcome was judged as fair (Nozick, 1974). This put the emphasis on the freedom of choice criteria as fundamental
to understanding utility or social well-being. Subsequently, A. Sen proposed a dualist vision of the individual as simultaneously consumer (aiming for satisfaction of his preferences) and citizen (considering objectives which surpass individual interests) (Sen, 1977, 1987). His theory of “capabilities” introduced the notion that agents can freely make their choices.

Another heterodox approach\(^9\) revolves around the legitimacy of determining the social value of goods and services based on the sole preferences of agents. Works by economic psychologists, for example, have highlighted the numerous biases in choice making (Tversky and Kahneman, 1981; Kahneman and Tversky, 1982).

### 2.2.1.2 The value of nature

Natural capital, in the form of land, has played an important role in classical economics. However, the recognition of the value of nature by classical economists takes a number of forms (Gomez-Baggethun et al., 2009). For Smith, it is the amount of labour associated with nature (the rent derived from its appropriation) that made value out of nature (Smith (1976) 1909). For Say, nature's services are costless, free gifts to human beings (Say, 1829). Ricardo denies that nature’s services contribute to the creation of exchange values (Ricardo (1817) 2001) (Figure 2-1).

It is Marx who first recognised the economic value of nature. For him, value emerged from the combination of labour and nature: “Labour is not the source of all wealth. Nature is just as much the source of use values (...) as labour, which itself is only the manifestation of a force of nature” (Marx (1891) 1970; Gomez-Baggethun et al., 2009). However, like others before him, he did not recognize nature as contributing to the creation of exchange values.

After Marx, nature was overlooked by economists during the 19\(^\text{th}\) century industrial revolution. Economists concentrated on labour and capital, rather than land and labour (e.g. Schumpeter, 1954). This period also saw an important move from use values to exchange values (Naredo, 2003; Gomez-Baggethun et al., 2009), which opened the door to theorization by neoclassical economics on the substitutability of natural resources and human-made capital, hence starting what Mayumi called the “temporary emancipation from land” (Mayumi, 1991; Gomez-Baggethun et al., 2009). For Solow, land had been removed from the production function following recognition that the inputs from nature could be...

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\(^9\)Heterodox economics refers to methodologies or schools that are considered outside of mainstream (namely neoclassical) economics.
substituted by manufactured capital (Solow, 1973). The issue of resource exhaustibility could therefore be solved by rising prices and producer choice switching towards the production of alternative consumption goods.

![Figure 2-1: Landmarks in the evolving conception of nature by economists](image)

Source: Gomez-Baggethun et al., 2009

The second half of the 20th century saw the development of standard economics to analyse environmental problems. The first academic group that took this direction was the Society of Environmental and Resource Economics in the early 1960s (Turner et al., 1994; Gomez-Baggethun et al., 2009). By this time, environmental and resource economics (‘environmental economics’) had extended beyond the scope of neoclassical economics, developing methods to value and internalizing the economic impacts on the environment into decision-making (notably through cost-benefit analysis). Environmental economics recognizes the neoclassical economic approach. However, it criticizes the fact that the contribution of nature is limited to those services that bear a price, which thus leads to a lack of consideration of non-marketed natural goods and services in decision-making.
processes (Costanza et al., 1997). In order to counterbalance this, environmental economists have developed methods to value non-marketed environmental costs and benefits (Gomez-Baggettun et al., 2009). This has led to the distinction of use and non-use values by Krutilla (Krutilla, 1967) and, subsequently the aggregation of all these values within the concept of Total Economic Value (TEV) (e.g. Heal et al., 2005). To ascertain such values, techniques were developed and refined over time (see section 2.3.2 for details about valuation techniques).

2.2.1.3 Environmental and ecological economics: Two different views on the value of nature (1980-1997)

In the 1980s, divergences in the views of environmental economists emerged. These stemmed from the fact that ecologists and economists use the term ‘value’ in discussions of ecosystems and their services in two different ways (Freeman, 2003)\(^\text{10}\). These two different uses of the word correspond to a difference between the *intrinsc* value (valuable in, and for itself, independent of any use or function – Callicott, 1989) and *instrumental* value (related to a means to some other end or purpose, something that contributes to some other goal–Costanza and Folke, 1997).

This opposition to the use of individual preferences as a sound indicator of value is the core argument that led to the differentiation of environmental economics (that accords to individual preferences) and ecological economics (that rejects it, and has developed alternative measurements and indicators) (Martinez-Alier et al., 1998). The Box 1 provides an overview of the various approaches to the value of nature.

\(^{10}\) Ecologists define value as that "which is desirable or worthy of esteem for its own sake, thing or quality having intrinsic worth" (Webster’s New Dictionary, 1988). Economists use the term value as "a fair or proper equivalent in money, commodities" (Ibid) where, as Freeman states, the "equivalent in money" represents the sum of money that would have an equivalent effect on the welfare or utilities of individuals.
### Box 1: A general value typology

<table>
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<tr>
<th>1. Anthropocentric Instrumental Value</th>
<th>This is equivalent to &quot;Total economic value&quot;. The non-use category is bound by the existence value concept which has itself been the subject of much debate. Existence value may therefore encompass some or all of the following motivations:</th>
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<td></td>
<td>i. intragenerational altruism: resource conservation to ensure availability for others;</td>
</tr>
<tr>
<td></td>
<td>ii. intergenerational altruism (bequest motivation and value): resource conservation to ensure availability for future generations;</td>
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<tr>
<td></td>
<td>iii. stewardship motivation: human responsibilities for resource conservation on behalf of all nature. This motivation may be based on the belief that non-human resources have rights and/or interests and as far as possible should be left undisturbed (if existence value is defined to include stewardship then it will overlap with the next value category outlined below).</td>
</tr>
<tr>
<td>2. Anthropocentric Intrinsic Value</td>
<td>This value category is linked to stewardship in a subjectivist sense of the term value. It could be culturally dependent. The value attribution is to entities which have ‘goods of their own’, and instrumentally use other parts of nature for their own intrinsic ends. It remains an anthropocentrically related concept because it is still a human evaluator that is ascribing intrinsic value to non-human nature.</td>
</tr>
<tr>
<td>3. Non-Anthropocentric Instrumental Value</td>
<td>In this value category entities are assumed to have goods of their own, independent of human interests. It also encompasses the good of collective entities (e.g. ecosystems) in a way that is not irreducible to that of its members (this category may not demand moral considerability as far as humans are concerned).</td>
</tr>
<tr>
<td>4. Non-Anthropocentric Intrinsic Values</td>
<td>This value category is viewed in an objective sense (i.e. &quot;inherent worth&quot;); the value that an object possesses independently of the valuation of evaluators. It is a meta-ethical claim, and usually involves the search for constitute rules or trump cards with which to constrain anthropocentric instrumental values and policy.</td>
</tr>
</tbody>
</table>

Source: adapted from Hargrove, 1992; Turner and Postle, 1994
This divergence over the value of nature has led to the creation of a new approach to the economics of nature. This was influenced by work on human-nature interactions (von Bertalanffy, 1968; Georgescu-Roegen, 1971; Odum, 1971; Daly, 1977; Kapp, 1983), and helped create what later became modern ecological economics. Gomez-Baggethun has reviewed the main differences between environmental and ecological economics (Gomez-Baggethun et al., 2009). Basically, the two overlap in that they use specific techniques to measure sustainability, to evaluate policies and to assist in decision-making (and in practice many scholars working in ecological economics also employ the tools of neoclassical microeconomics). Both approaches, however, differ significantly in the qualitative framework within which they operate (Costanza, 1991; Ozkaynak et al., 2002; Gowdy and Erickson, 2005). Hence, one is operating within the neoclassical economics framework (theory of consumer choice, perfect information, and marginal productivity theory of distribution), while the other challenges this framework and views the economic system as an open subsystem of the ecosphere exchanging energy, materials and waste flows with the social and ecological systems with which it co-evolves (Daly, 1977; Noorgard, 1994; Gomez-Baggethun et al., 2009).

As a direct consequence of the controversy between environmental and ecological economics, the valuation of natural goods and services has differed in views and methods. Ecological economists maintain that incommensurability11 is a key obstacle to the valuation of natural goods in monetary terms (Martinez-Alier, 2002). In contrast, environmental economists use neoclassical economics to value ecosystem goods and services. Nevertheless, ecological economics also has a valuation dimension: researchers have developed methods to account for the physical and social costs involved in economic performance, using monetary, biophysical accounts and other non-monetary valuation languages (Martinez-Alier, 2002). They prefer to use deliberative and multi-criteria based decision processes, rather than extended cost-benefit analysis (Martinez-Alier, 1987; Munda, 2004; Spash, 2008).

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11 Incommensurability reflects the idea that different types of value may not be expressed in a common measurement unit (Kapp, 1965, 1983; O’Neill, 1993).
2.2.2 Media interest in the economic valuation of ecosystems (1997-2000)

One article has had a radical impact on the history of environmental economics. Published in 1997 in *Nature*, the monetary valuation of ecosystems of the world carried out by Costanza and his colleagues (Costanza et al., 1997) has had a major impact on the subject. For the first time it made the economic valuation of ecosystems understandable to a large public audience thanks to extensive media coverage. It also triggered major controversy on the limits and uses of economic valuation of ecosystems.

Costanza started his work from the premise that there was no global estimation of the value of ecosystems goods and services. Instead, values most often originated from specific valuation methods or specific situations (e.g. valuation of degradation, or cost-benefit analysis of a conservation project). Costanza therefore proposed a synthesis of existing data in the form of a meta-analysis. The method used was simple: Costanza first estimated the value per biome, and then multiplied this unitary value by the global surface of this biome. Values were then aggregated in order to get to the total economic value of global ecosystems: $33 \times 10^{12}$ USD. The authors suggested two uses for such an exercise: taking the values into account in national accounting on the one hand and in project evaluation on the other hand.

The publication of this article has not only had an impact among the public, it has also raised a huge controversy among academics. Some have offered positive opinions about the work (Herrendeen, 1998; Daly, 1998), while some others, such as Norgaard in his article “Next, the value of God...” were very critical (Norgaard et al., 1998). From the debate around this article, three major discussion topics emerged:

i) the methodological aspects of valuation techniques: the use of marginal measurement and the aggregation of unitary values to get a global asset value, the ignorance of ecological complexity, and the importance of thresholds in ecosystem services provision (Daly, 1998; Turner et al., 1998);

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12 This title and content of this section is inspired by P. Meral’s work on the history of ecosystem services (Meral, 2010)

13 As Costanza himself stated: “the paper received broad media coverage, including stories in the NY Times, Newsweek, Science, Science News and US News and World Report and reports on US National Public Radio and the BBC. It was also included as one of Discover magazine’s top 100 science stories for 1997.”
ii) the utilitarian approach to monetary valuation of ecosystem services: many heterodox economists see this article as a step backwards in ecological economics, limiting the role and value of nature to some very limited services (Norgaard et al., 1998); and

iii) the role of such valuation for decision-making: despite the decision-making objectives pursued by Costanza’s paper, many economists wondered about the usefulness of such valuation (Toman, 1998), and some even find it counterproductive (Turner et al., 1998), asserting that the paper does not bring any revolutionary material to the debate and it is little different from a Meadows-like report\textsuperscript{14}.

In spite of its critics, however, Costanza’s work has contributed to bringing ecological economics to the forefront of the scene.

2.2.3 The road to science and policy reconnection (2000-2013)

Following Costanza’s work, researchers in the 2000s have sought to better connect to decision-making. This was marked not only by the Millennium Ecosystem Assessment (MEA), but also by research on valuation techniques and policy-oriented initiatives on the economics of ecosystems (‘The Economics of Ecosystems and Biodiversity’ - TEEB initiative - in the first instance) in an attempt to operationalize the valuation of ecosystem services.

2.2.3.1 Millennium Ecosystem Assessment

As Perrings states: “the Millennium Ecosystem Assessment has changed the way that we think about the interaction between social and ecological systems. By connecting ecological functioning, ecosystem processes, ecosystem services and the production of marketed goods and services it has identified ecological change as an economic problem. It has also drawn attention to a new dimension of the environmental sustainability of economic development” (Perrings, 2006). But above all, the MEA has enabled the adoption of a commonly agreed classification for ecosystem goods and services that could be understood by a large audience.

\textsuperscript{14}The Limits of Growth was commissioned by the Club of Rome and published in 1972 with Donella and Dennis Meadows as first authors (Meadows et al., 1972). The aim of the publication was not to make specific predictions, but to explore how exponential growth interacts with finite resources. In this regard Costanza’s paper could be viewed as a set of large figures generated in order to highlight the importance of nature to the World.
The MEA was launched by scientists and decision-making experts in a bid to apply the work carried out on climate change (through the International Panel on Climate Change – IPCC - set up in 1988) to biodiversity and ecosystems. The scientific aim is also to prepare an international evaluation of ecosystems based on a commonly agreed methodology.

The MEA distinguishes 4 categories of services (MEA, 2005):

- **Provisioning services**: these are goods and services from ecosystems for food (crops, livestock, fisheries, aquaculture, wild food, and freshwater), energy fuel (firewood, ethanol production from cereals), manufacturing materials (timber, fibre), products for genetics and pharmacology.

- **Regulating services**: these are the functions of process control exercised by natural ecosystems that benefit humans. They include services as diverse as climate regulation, water cycling, air quality control, erosion control, pest control, prevention/mitigation of natural hazards, pollination and treatment of organic waste and pollutants.

- **Supporting services**: these services do not directly benefit populations but they condition the functioning of ecosystems. These services may include: nutrient cycling, soil formation and primary production.

- **Cultural services**: these include all benefits of a recreational, aesthetic, existential, spiritual, scientific, educational, and heritage nature originating from ecosystems.

Ecosystem services contribute to human well-being through different channels, and also to various components of well-being, as detailed below (Figure 2-2).
One of the major outcomes from the MEA has been the setting up of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) in April 2012 (following years of negotiations). This platform (an equivalent structure to the International Panel for Climate Change – IPCC - for biodiversity) creates the conditions for international cooperation in the science-policy arena. It is the main tool for biodiversity research to influence international policy-making. However, the task appears to be much more difficult than for climate change, for which only one major indicator (CO₂ level) was referred to. In the case of biodiversity, estimating the level of biodiversity loss and degradation of ecosystems in order to take action against these is very complex, since decisions have to be based on sets of indicators that are very difficult to document.

### 2.2.3.2 The TEEB initiative and recent developments

Following the MEA, The Economics of Ecosystems and Biodiversity (TEEB) initiative has aimed to promote a better understanding of the value of ecosystem services, and to propose
economic tools that take into account this value (TEEB, 2008). TEEB has addressed several aspects of environmental economics (e.g. rethink subsidies, repay non-marketed benefits, extend the polluter-pay principle, develop new markets for biodiversity and share biodiversity conservation benefits). TEEB has presented its results to various targeted audiences (such as TEEB for Business, TEEB for citizens, and TEEB for national and international policy-making). As part of its research TEEB has proposed a renewed pathway from ecosystems and biodiversity to human well-being, building on the MEA. It goes beyond the MEA in that it introduces the notion of value as part of well-being (Figure 2-3).

![Figure 2-3: From Ecosystems Functions to Value, TEEB Overview Diagram](source: adapted from Haines-Young and Potschin, 2009)

TEEB has set the background for ecosystem valuation research worldwide and lists the various tools available in order to influence business and decision-making. However, it failed to address precise concerns about ecosystem and biodiversity economics as it kept the research at a rather theoretical level of thinking.

TEEB studies are currently being developed at national and regional scales in order to provide more applied results. These initiatives are concomitant with the creation of the
“Ecosystem Services” research journal, in partnership with the Ecosystem Services Partnership (ESP)\textsuperscript{15}.

In particular, compulsory national studies have been developed as part of the National Ecosystem Assessments in the European Union, under the 2020 Biodiversity Strategy (EC, 2011): “\textit{Member States, with the assistance of the Commission, will map and assess the state of ecosystems and their services in their national territory by 2014, assess the economic value of such services, and promote the integration of these values into accounting and reporting systems at EU and national level by 2020}”.

In parallel to TEEB, studies of marine and coastal ecosystem services valuation have multiplied: all underscore the importance of marine areas in providing goods and services. In the Mediterranean, they are worth nearly 26 billion euros annually, with cultural and leisure services providing two-thirds of that total (Mangos et al. 2010). In the United Kingdom, provisioning services are worth 713 million euros, cultural services 15 billion euros, regulating services between 840 million euros to 10 billion euros, while supporting services exceed 1 trillion euros in value (Beaumont et al. 2008). In these valuations, the estimated worth of “commercial” goods and services is less than that of cultural, supporting and regulating services.

In spite of these initiatives, and the great advances made in ecological economics to characterize ecosystem services provisioning in economic terms and to monetize biodiversity losses, the valuation of ecosystem services remains a real research challenge. It is however very promising with regards to the opportunities created for natural assets to be accounted for in national accounts, or damage to ecosystems compensated. This approach has also opened the door for an increased use of market-based instruments for biodiversity conservation, such as the payments for ecosystem services approach.

\subsection*{2.3 Valuation of ecosystems: concept and methods}

The list of ecosystem services as defined by the MEA (MEA, 2005) provides a clear and comprehensive approach to the benefits gained from ecosystems. Nevertheless, it must be

\textsuperscript{15}As part of this partnership, the author has contributed to the TEEB final report (Armstrong et al., 2010). Since then, he has been highly involved in this partnership and became part of the biome groups “coastal systems”, “coral reefs”, “coastal wetlands”, “polar regions and high mountains” and co-led the Caribbean regional chapter.
adapted in order to be used for translating these benefits into economic value (that is to say, avoiding double counting and facilitating aggregation of values). While criticism over the concept of services has increased in recent years, no new concept has really emerged that provides a better framework for economic valuation.

Ecosystem goods and services are by definition, benefits that contribute to human well-being. If we consider that there is a correlation between biodiversity and ecosystem functioning on the one hand, and the quantity and quality (or stability) of goods and services provided on the other hand, then valuing the economic value of goods and ecosystem services is a reasonable approximation of the "economic value of ecosystems and biodiversity" for human well-being (see above Figure 2-3).

This understanding leads to the notion that conservation efforts regarding ecosystems and biodiversity are guarantees of the value of services provided. Conversely, the erosion of biodiversity necessarily implies a loss of the ecosystem goods and services which biodiversity supports. Depending on whether one adopts a positive or negative approach, the notion of "economic value of biodiversity and ecosystems" can estimate the value of services provided or the cost of the services lost.

2.3.1 Framework for economic valuation of ecosystem services

Valuation exercises, according to the TEEB report, should ideally: i) acknowledge the existence of alternative, often conflicting, valuation paradigms; and ii) be explicit about the valuation paradigm that is being used and its assumptions (TEEB, 2010). A review of various approaches to valuation distinguishes two well-differentiated ones: biophysical methods; and preference-based methods. Biophysical valuation uses a 'cost of production' perspective that derives values from the measurement of the physical costs (e.g. in terms of labour, surface requirements, energy or material inputs) of producing a given good or service (TEEB 2010). In terms of ecosystem services valuation, this approach values the costs associated with the maintenance of a given ecological state.

In contrast, the preference-based approach relies on models of human behaviour and rests on the assumption that values arise from the subjective preferences of individuals. This approach assumes the commensurability of values of ecosystem goods and services. It also assumes that monetary measures offer a way of establishing the trade-offs involved in the
various uses of ecosystems. The resulting value of this approach for ecosystem goods and services is the ‘output value’ or, for most authors, the Total Economic Value (TEV).

The TEV is not a means to provide an absolute value of ecosystems, but it rather enables the consideration of the multiple economic uses that underlie the values of ecosystems (Balmford et al., 2002). The advantage of such a framework is that all the ecosystem values can be compared and aggregated (thanks to commensurability of values). Also, thanks to its extensive use in ecosystem valuation literature, the TEV approach enables comparison and transfer of value from one site to another (this method is called benefit transfer). Economic valuations of mangroves and coral reefs ecosystems have thus extensively relied on TEV for the valuation of their services.

TEV can be divided into two broad categories: use values and non-use values (also called passive use values) (see Figure 2-4 for a breakdown of the TEV). Use values are associated with the direct use of ecosystems. They can be divided into direct active use values, induced use values and indirect use values. Direct active use values are the most common uses of biodiversity, namely the extractive uses (e.g. fisheries, exploitation of raw materials, mining, dredging) and the non-extractive uses (tourism, recreational uses, research and education). Induced active use values cover services provided by ecosystems as an input for marketed services (such as aquaculture). In this case, the use is examined as a means to support the marketed activity (e.g. the nutrients that flow through farmed fish cages). The indirect use values consist of the regulating and support services of the ecosystems, and are therefore associated with the ‘ecological functions’ of ecosystems.
The non-use values are related to the satisfaction of knowing that a species or ecosystem exists (existence value) (Krutilla, 1967) or knowing that future generations (bequest value) (Pearce and Moran, 1994) or other people (altruist value) (Christie, 2004) will have access to such benefits provided these benefits are managed in a sustainable way. Non-use values therefore relate to current and future values. They rely on the permanence of their existence and the maintenance of ecosystems, independently of the uses of these ecosystems. They have also been called preservation values (Greenley et al., 1981, Walsh et al., 1984; Point, 1998), passive use values (Carson et al., 1992; Carson et al., 1999) or patrimonial values (Rambonilaza, 2010). Bequest value in some regions, as is the case with marine and coastal ecosystems (these are abbreviated to MCE below) in West Africa, may have a greater importance because of the traditions associated with the ecosystems for local populations, and the willingness of these populations to see their children maintain these traditions after them.

Interestingly, TEEB has recently produced a framework for promoting improved articulation between ecosystem services and TEV components. Until then, the linkages...
between services and TEV components were hardly explained in the literature, and most studies assumed the linkage was evident (although it does not always appear to be so). The TEEB study has therefore provided a conversion table to help in valuing ecosystem services through the TEV framework (Table 2-1).

**Table 2-1: Valuing ecosystem services through the TEV framework**

<table>
<thead>
<tr>
<th>Group of services</th>
<th>Direct use value</th>
<th>Indirect use value</th>
<th>Option value</th>
<th>Non-use value</th>
</tr>
</thead>
<tbody>
<tr>
<td>Provisioning</td>
<td>Yes</td>
<td>N/A</td>
<td>Yes</td>
<td>N/A</td>
</tr>
<tr>
<td>Regulating</td>
<td>N/A</td>
<td>Yes</td>
<td>Yes</td>
<td>N/A</td>
</tr>
<tr>
<td>Cultural</td>
<td>Yes</td>
<td>N/A</td>
<td>Yes</td>
<td>Yes</td>
</tr>
<tr>
<td>Support</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

*Source: (TEEB, 2010)*

2.3.2 Review of environmental goods valuation techniques

When it comes to valuation techniques, most references point to the utilitarian model, which is the one best able to value one ecosystem good or service in monetary terms. Through this approach, the economic value of ecosystem services is measured by the willingness of a person to acquire this good (less its cost of production). So when nature provides a service, it is the willingness to pay of individuals that is likely to reflect the value of the resource providing the service in question (whether there is an effective transaction or not). In other words, the monetary value of the MCE can be evaluated by estimating their contribution to market activities (that save costs and benefits) and non-market activities (which only records profits).

This has led to a common categorization of the techniques available to value ecosystem services, which are: i) direct market valuation approaches; ii) revealed preference approaches, and iii) stated preference approaches. In addition, one other method should be mentioned, the indirect valuation technique: the transfer of benefits (or value transfer). The following sub-sections provide a description of the four different approaches and the

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16 See the earlier debate between environmental economists and ecological economists. The latter have also developed valuation techniques; however it was decided not to detail these further in this thesis given we focus on economic valuation in monetary terms (rather than describing value in qualitative equivalent, or through multi-criteria analysis).

17 This technique is not considered as a valuation technique *per se*. However, it enables one to adapt one value (valued using one of the other methods) to the studied site or ecosystem.
tools associated for valuation. A table at the end presents each approach’s strengths and weaknesses and the conditions for application.

### 2.3.2.1 Direct market valuation methods

These approaches are divided into three methods: 1) the market price-based method; 2) the cost-based method; and 3) the method based on production functions. The principal advantage of these methods is that they stem from actual markets for which data is available. Another advantage is that they directly reflect the preferences of consumers (through prices) or costs (through expenditures).

#### 2.3.2.1.1 Market price-based method

The market price-based method is a common method used for the valuation of provisioning services for which the commodity considered in the service is marketed. The market price method estimates the economic value of ecosystem products or services that are bought and sold in markets. For those resources for which markets exist, economists determine individuals’ values by observing their preferences and willingness to pay for the goods and services at the prices offered in the market.

The standard method for measuring the use value of resources traded in the marketplace is the estimation of consumer surplus and producer surplus using market price and quantity data. The total net economic benefit, or economic surplus, is the sum of consumer surplus and producer surplus (King and Mazzotta, n.d.). It uses standard economic techniques for measuring the economic benefits from marketed goods, based on the quantity people purchase at different prices, and the quantity supplied at different prices.

#### 2.3.2.1.2 Cost-based methods

Cost-based methods are based on the estimation of the costs associated with the existence or absence of one ecosystem service. There are three main methods used in this approach. The first is the avoided cost method, which relates to the costs that would have been incurred in the absence of ecosystem services. This method is used mostly for services provided for protection/regulation of damages. For instance, the avoided cost method can be used for the valuation of protection against storms by mangroves, or flood regulation by wetlands.

The second is the replacement cost method, which estimates the costs incurred by replacing the ecosystem service with an artificial mechanism or technology that ensures the same
output. For instance, the value represented by the filtering service provided by wetlands is equal to the costs incurred by the setting up of one water treatment plant that would replace this service in the absence of wetlands.

The third method is the mitigation or restoration cost method, which relates to the cost of mitigating or restoring the effects of the loss of one ecosystem service. This method is more difficult to apply because there are less monetary data available for restoration or mitigation costs, than there are for replacement by some form of technology. Also, it tends to be more site-specific too. For that reason, this method is used in very few studies.

Though commonly used, these cost-based methods have the disadvantage of conflating value with cost. For example, the use of the cost of a water treatment plant enables an estimate of the costs of water filtering service, but not the value of this service (which is derived from the willingness of beneficiaries to pay for pure water).

2.3.2.1.3 Production function-based approach

The production-function method measures the contribution of an ecosystem service to the delivery of a service or commodity which is traded on an existing market. The contribution is estimated by reference to the enhancement of productivity or income created by the service (Mäler et al., 1994; Pattanyak and Kramer, 2001; TEEB, 2010). The method consists of first getting a good knowledge of the changes created by a change of one environmental quality variable on the delivery of one ecosystem service (such as air quality, soil erosion, or water temperature). Then, the impacts of the changes are valued in terms of the corresponding changes in marketed output of the traded activity\(^\text{18}\).

This approach supposes that the evaluation benefits from a good set of data are required to enable the determination of the production function. Otherwise, the use of such a method is not possible. In any case, this method is difficult and costly to implement.

2.3.2.2 Revealed preferences methods

Revealed preference methods are based on the observation of choices by individuals in markets that relate to the ecosystem service considered. Observations are carried out by survey where individuals reveal their choice. There are two main methods that use revealed preferences: the travel costs method (TCM) and the hedonic pricing method (HPM).

\(^{18}\)For further details on the production function-based approach, see Freeman (2003).
The travel costs method (TCM) is used for estimating the value of service that involves travelling by the agent. This is the case with tourist activities for instance. The TCM enables the valuation of the recreational service of the ecosystem by estimating the costs incurred by tourists to come and enjoy recreational activities in the considered ecosystem (this includes direct expenses such as travel to the site and entrance fees, as well as opportunity costs on time spent to travel). This method is very useful in the case of recreational activities for which access is typically not allocated through markets (Freeman, 2003). With TCM, the value of the recreational service of one ecosystem is the sum of the individual value calculated upon the costs incurred for travel to the studied site.

The hedonic pricing method (HPM) is based on the implicit demand for an environmental attribute of marketed commodities. It can be used to estimate economic benefits or costs associated with environmental amenities, such as aesthetic views or proximity to recreational sites. The basic premise of the HPM is that the price of a marketed good is related to its characteristics, or the services it provides. The HPM is most often used to value environmental amenities that affect the price of residential properties. In this case, the value of a change in amenity will be reflected by a change in the value of the property. By estimating a demand function for property, the analyst can infer the value of a change in the non-marketed environmental benefits generated by the environmental good.

2.3.2.3 Stated preference methods

The stated preference methods use theoretical markets used to estimate the value of an ecosystem service. The simulation of markets is created using a survey on hypothetical changes in the provision of the ecosystem services. The main difference with the revealed preference method is that it draws its data from people’s responses to hypothetical questions rather than from observations of real-world choices. These methods are used when no direct market valuation is possible and when no surrogate market exists from which the value of the ecosystem can be calculated (as is the case with the two previous approaches). The three main stated preference methods include the: i) contingent valuation method; ii) choice experiment; and iii) group valuation.

2.3.2.3.1 Contingent valuation

The contingent valuation method uses questionnaires to ask people how much they would be willing to pay (WTP) to increase or enhance (or maintain) the provision of ecosystem service they enjoy. Alternatively, this method can also be used to value the willingness to
accept a loss in the provision of ecosystem services. The contingent valuation method is the most common method in the literature to estimate non-use values (see for instance Hundloe 1984, Spash et al., 1998; Ayob et al., 2001; Seenprachawong, 2003; Subade, 2005). Non-use values are estimated by the WTP for the preservation or restoration of ecosystems (usually via the financing of an MPA) independently of the use made to this ecosystem by the surveyed person. However, it has many biases (see for instance Arrow, 1993).

2.3.2.3.2 Choice experiment

The choice experiment method estimates the WTP of an individual based on the choice made among various scenarios. Surveyed people are given a number of alternatives which offers the same attributes at different levels of realization. The choices made can then be analysed to determine the marginal rate of substitution between any characteristics, and the level of realization of the attribute considered. One of these attributes is money, which enables the computation of the respondent’s WTP in monetary terms for the considered service on the basis of the choice between alternatives. Alternative methods have been developed based on choice experiments. For instance, the conjoint analysis asks respondents to rate a set of attributes on a specified scale (Freeman, 2003).

Advantages of the choice experiment method include:

- It is (among the stated preference methods available) the most successful in accounting for social attributes, cultural and ethical natural assets (Dachary-Bernard, 2004). It is therefore the best in guaranteeing an accurate reflection of the WTP of individuals.
- It allows better control of the experience, especially the hypothetical scenarios (Kjaer, 2005). In this way, individuals agree on a common vision of the ecosystem and make their choice based on this vision. This guarantees that individuals homogenize their perceptions, which is very difficult in the case of non-use values that integrate both individual and collective representation.
- It minimizes the risk of strategic behaviour by the interviewee by incorporating the cost as one component among others (Bennett, 1996). This is important with regards to the non-market and non-tangible values of non-uses, since it removes monetization from the core of the survey.
- It has the advantage of characterizing an environmental good by its various facets (attributes). This feature allows for a better understanding of the true complexity of
an ecosystem and a fortiori associated ecosystems: the individuals interviewed can find among the different attributes of non-use values those that best reflect their perceptions of the ecosystem's non-use values.

In practice, the method consists of providing individuals with several scenarios. Each scenario is a unique combination of attribute levels. Each scenario represents a possible evolution of the ecosystem, usually resulting from a change in policy. Often, it offers the individual a choice between three thresholds of realization for the attributes, two thresholds that mean change for the attribute and one baseline level (commonly referred to as the "status quo"). The presence of the status quo enables respondents the option to reject the two thresholds that would mean change. It also ensures that the answers are all relative to a baseline level and are therefore comparable (Rolfe et al., 2000).

2.3.2.3 Group valuation

The third stated preference method for valuation as identified in TEEB is the group valuation method. This method uses stated preference techniques, with the valuation carried out as a deliberative process during meetings with groups of the surveyed population. This method is used as a way to take into account value pluralism, incommensurability, non-human values or social justice (Spash, 2008; TEEB, 2010).

2.3.2.4 Transfer of benefits (value transfer)

The transfer of benefits (or value transfer) method consists of estimating economic values for ecosystem services by transferring available information from studies already completed in another location and/or context. The transfer method uses existing information on the value of an ecosystem service (average or marginal), economic parameters (e.g. opportunity cost of time), or economic function (estimated at one place and time) to make inferences about the same service at another place and time. Benefit transfer is often used when it is too expensive and/or there is too little time available to conduct an original valuation study. It is important to note that benefit transfers can only be as accurate as the initial study.

In practice, estimates are either transferred as value units (e.g. means or medians) or as value functions (conditioned on explanatory variables) that define the attributes of an ecological and economic choice setting (Wilson and Hoehn, 2006). Of these two approaches function transfer is regarded as more robust because it uses a set of explanatory variables.
upon which values are deemed to depend, while unit transfer tends to be a more simple adoption of monetary numbers from one context into another. However, when data is lacking (as is the case in my case studies), transfer of units can be a good alternative.

The benefit transfer method is most reliable: i) when the original site and the study site are very similar in terms of factors such as quality, location, and population characteristics; ii) when the environmental change is very similar for the two sites; and iii) when the original valuation study was carefully conducted and used sound valuation techniques.

Even though the number of valuation studies has increased recently, valuations carried out in Africa (and moreover in MCEs in West Africa) are scarce. This makes the value transfer method more problematic and requires careful attention to the source study characteristics. Furthermore, specific work across the region would be required to validate the values transferred in the region, including measuring the ability of MCEs to restrict the power of the waves, (depending on the frequency and magnitude of storms) and flooding and reduce erosion due to currents in the estuaries and mangroves. Unfortunately, this information is absent from the studies carried out in the region and it was not possible to undertake such important ecological research during the fieldwork carried out in this study. Investigative work on the ground is therefore needed to distinguish the ecosystem areas that play an effective role in providing the service considered from those that play a lesser role in providing this service.

2.3.2.5 Review of advantages and constraints of method

The Table 2-2 provides a review of various valuation techniques as detailed above and gives the main advantages and constraints of each of these techniques.
<table>
<thead>
<tr>
<th>Valuation approach</th>
<th>Method</th>
<th>Sub-method</th>
<th>Principal application field and advantages</th>
<th>Constraints</th>
</tr>
</thead>
<tbody>
<tr>
<td>Direct market valuation</td>
<td>Market price-based</td>
<td>Principal application in valuation of direct use values - advantages in accuracy of the valuation (based on market data) - production data easier to obtain and at lower costs</td>
<td>- Market data may not reflect the value of all productive uses of a resource - in some cases, markets are distorted (by subsidy or lack of competition for instance), which causes prices not to be a good reflection of marginal preferences. - seasonal variations and other effects on price must be considered</td>
<td></td>
</tr>
<tr>
<td>Cost-based approach</td>
<td>Avoided cost method</td>
<td>Efficient method for indirect use values related to protection services against extreme events (storm protection, flood regulation) - less data and resource intensive methods - methods provide surrogate measures of value that are as consistent as possible with the economic concept of use value</td>
<td>- Should be used with caution in presence of uncertainty - assumes that expenditures to repair damages or to replace ecosystem services are valid measures of the benefits provided - does not consider social preferences for ecosystem services, or individuals’ behaviour in the absence of those services</td>
<td></td>
</tr>
<tr>
<td>Replacement cost method</td>
<td>Good method for valuation of regulating services that have an artificial equivalent (e.g. water purification, waste treatment) - less data and resource intensive methods - methods provide surrogate measures of value that are as consistent as possible with the economic concept of use value</td>
<td>- Should be used with caution in presence of uncertainty - tends to conflate cost and value - requires information on the degree of substitution between the market good and the natural resource. Few environmental resources have such direct or indirect substitutes. Substitute goods are unlikely to provide the same types of benefits as the natural resource (e.g., stocked salmon may not be valued as highly by anglers as wild salmon) - services being replaced probably represent only a portion of the full range of services provided by the natural resource</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Restoration cost method</td>
<td>Based on the costs of mitigating the effects of loss of ecosystem service, or costs of getting services restored</td>
<td>- Restoration activities still not common and processes costs unknown - not suitable for all ecosystems: some ecosystems cannot be restored if degraded (seagrass meadows</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Valuation approach</td>
<td>Method</td>
<td>Sub-method</td>
<td>Principal application field and advantages</td>
<td>Constraints</td>
</tr>
<tr>
<td>--------------------</td>
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<td>---------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------</td>
<td>----------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------</td>
</tr>
</tbody>
</table>
|                    |                                |            | - Can be used for valuation of regulating services (water treatment by wetlands for instance)  
- less data and resource intensive methods  
- methods provide surrogate measures of value that are as consistent as possible with the economic concept of use value  
- for instance are difficult to restore                                                                                                    |                                                                                                                                                                                                           |
|                    |                                |            | Production function-based method  
Method used when there is sufficient data on cause-effect of ecosystem service delivery on the output level of marketed commodities  
Accurate valuation and takes into account variations in production (not the case with other methods)                                                                 | - In most cases, not enough data and knowledge about cause-effects linkages  
- high tendency for double-counting                                                                                                           |
| Revealed preference approach | Travel cost method |            | Mostly used for recreational activities (direct use values)  
- Based on actual behaviour (not contingent) which causes less biases to the valuation  
- inexpensive  
- relatively easy to interpret and explain                                                                                              | - Market imperfections and policy failures can distort results; good quality data (large data sets and accuracy) required; involved statistical treatment  
- expensive and time-consuming  
- inappropriate for non-use values  
- technique assumes that travel is motivated by one single purpose  
- question of opportunity cost of time can be problematic                                                                                     |
|                    | Hedonic price method |            | Mostly used for non-extractive use values (aesthetic, cultural values), through estimation of land price for instance  
- it can be used to estimate values based on actual choices  
- property records are typically very reliable and easy to get  
- the method is versatile, and can be adapted to consider several possible interactions between market goods and their direct consequences  | - Market imperfections and policy failures can distort results; good quality data (large data sets and accuracy) required; involved statistical treatment  
- can be expensive and time-consuming  
- scope of environmental benefits that can be measured is limited to things that are related to housing prices  
- the method will only capture people’s WTP for perceived differences in environmental attributes (and their direct consequences) |
<table>
<thead>
<tr>
<th>Valuation approach</th>
<th>Method</th>
<th>Sub-method</th>
<th>Principal application field and advantages</th>
<th>Constraints</th>
</tr>
</thead>
</table>
|                    |        |            | environmental quality.                      | - results depend on model specification  
|                    |        |            |                                              | - limited scope of application         |
| Stated preference approach | Contingent valuation method |            | Can be used for values for which no market data is available or in data-poor situations (non-use values, indirect use values) | - bias due to strategic responses by respondents  
|                    |        |            |                                              | - expensive and time-consuming         |
|                    |        |            |                                              | - ‘insensitivity to scope’ problem (same WTP for protection of one hectare or 1000 hectares)  
|                    |        |            |                                              | - issue of commensurability of non-use values  
|                    |        |            |                                              | - problems of valuation in non-monetised economies |
| Choice experiment |        | Best for non-use values | - it enables better control of experience  
|                    |        |            | - it diminishes biases due to strategic behaviour of surveyed persons  
|                    |        |            | - it describes nature by various facets (attributes) | - use of fictional scenarios  
|                    |        |            |                                              | - presumes trust of respondents in the realization of the scenario  
|                    |        |            |                                              | - market imperfections and policy failures can distort results; good quality data (large data sets and accuracy) required; involved statistical treatment  
|                    |        |            |                                              | - expensive and time-consuming         |
| Deliberative approach (group valuation) |        | Can be used in contexts where deliberative valuation is favoured | - valuation techniques not really adapted to group valuation |
| Based on previous results | Transfer of benefits (value transfer) | Can be used for all services | - may not be accurate  
|                    |        |            |                                              | - good studies relating to the policy or issue in question may not be available.  
|                    |        |            |                                              | - reporting of existing studies may be inadequate to make the needed adjustments.  
|                    |        |            |                                              | - adequacy of existing studies may be difficult to assess.  
|                    |        |            |                                              | - extrapolation beyond the range of characteristics of the initial study is not recommended.  
|                    |        |            |                                              | - benefit transfers can only be as accurate as the initial value estimate.  
|                    |        |            |                                              | - unit value estimates can quickly become dated. |

Source: adapted from: Barbier, 2007; Freeman, 2003; TEEB, 2010; King and Mazzotta, n.d.
2.4 Methods for valuing conservation benefits

This sub-section first explains cost-benefit analysis, the most common method used for assessing the net benefits of management measures or policies. It then presents other methods used, namely cost-effectiveness analysis and multi-criteria analysis. Many other methods exist that have not been presented here: environmental impact assessment, strategic environmental assessment, risks-benefits analysis, as they were inappropriate for the proposed study. However, these are detailed in an OECD publication (Pearce et al., 2006).

2.4.1 Cost-benefit analysis

Cost-Benefit Analysis (CBA) is a widely used financial and economic appraisal tool for projects. It is particularly useful when a choice has to be made between several projects (selection), and when the project involves a stream of benefits and costs over time, usually covering more than one year (from several to dozens of years). The basic theory of CBA can be defined as a comparison between the decrease and the increase of human well-being (or utility) associated with one project or a policy development. The aggregation of benefits from various social groups can lead to the summation of WTP or Willingness to Receive (WTR) without acknowledging the specific situations of losers and winners or, alternatively, applying coefficients to less favoured groups.

The costs and benefits within CBA are considered within a time horizon. To account for time in CBA, a discount rate is used. The TEEB report provides a good definition of discount rate (p. 260):

“an investor has a choice between letting a valuable tree grow at a rate of 5 per cent per year, or cutting the tree down, selling it and putting the money in the bank. Which decision is best depends on the rate of interest the bank pays. If the bank pays 6 per cent and the price of timber is constant, the investor will earn more money by cutting the tree down and selling it, that is, by converting natural capital into financial capital. (...) Suppose [now that] the tree was not growing at all and the rate of interest on money was 6 per cent. By not cutting down the tree and putting the money earned from selling it in the bank, the investor would be losing – per cent per year. This would be the discount rate on the tree in the world of financial investment” (TEEB, 2010).
Costs and benefits are applied as a coefficient which reflects the discounting of research and development projects that can be expressed as:

\[ DC_t = \frac{1}{(1 + s)^t} \]

Eqn 2-1: Discount coefficient at the time \( t \) (\( s \) being the discount rate)

The general formula of CBA is presented below. This formula should be positive if the project or policy is considered acceptable. Alternatively, the outcome of applying this formula for policy A should be superior to the result for policy B, for policy A to be preferred to B (i.e. value for policy A/value for policy B is superior to 1).

\[
\sum_{t,i}^{T,N} w_i \hat{B}_{i,t} \cdot (1 + e \cdot y_i)^t - w_i \hat{C}_{i,t} \over (1 + s(t))^t
\]

Eqn 2-2: General formula for the cost-benefit analysis

\( W \) is a coefficient reflecting the social value attached to benefits (and costs) of various group of individuals. \( S(t) \) is the discount rate expressed according to time (t). Benefits are likely to increase with time at the rate \( e \cdot y \). This rate considers the increase of revenue per inhabitant (increase rate = \( y \)) and the positive elasticity of WTP compared to revenue (e). In this case, \( e \cdot y \) is different from inflation. It applies to estimation of future benefits. The circumflex on \( B \) and \( C \) indicates that these are expected values for benefits and costs (or values adjusted according to risks). \( T \) is the time horizon, \( i \) the \( i^{th} \) individual considered and \( N \) the total number of individuals considered.

In most cases, the coefficient is equal to 1 and \( s \) is a constant not variable with time. The CBA formula can therefore be simplified and expressed as:

\[
\sum_{t,i}^{T,N} \frac{\hat{B}_{i,t} - \hat{C}_{i,t}}{(1 + s)^t}
\]

Eqn 2-3: Simplified equation of the cost-benefit analysis
As Pearce noted in his book, CBA has attracted a lot of critics, including questions about its theoretical basis. In particular with respect to the Kaldor-Hicks compensation principle within welfare economics\(^\text{19}\), and the fact that the “social welfare function” used for CBA is only one function among many others that could have been used for decision-making (Pearce et al., 2006).

Furthermore, the issue of discounting in CBA offers many challenges. Among them, the choice of the discount rate may not reflect reality, and moreover it is not determined by any specific economic guidelines. High discount rates could lead to the long-term degradation of biodiversity and ecosystems. A 5 per cent discount rate implies that biodiversity loss 50 years from now will be valued at only 1/7 of the same amount of biodiversity loss today (TEEB, 2010). In sum, decisions about discount rate levels may greatly change the results of the study.

### 2.4.2 Cost-effectiveness analysis

Cost-effectiveness analysis (CEA) is used when there is only one management option available. CEA compares the effectiveness (E) with the costs (C) involved, through an E/C ratio. With CEA, it is not the option that is examined. Rather, it is how the option is implemented that is measured. CEA does not inform decision-makers on whether the ecosystem is worth being conserved or protected, except in instances when C and E are both expressed in the same monetary unit (Pearce et al., 2006).

### 2.4.3 Multi-criteria analysis

Multi-criteria analysis (MCA) is a tool that values benefits provided by ecosystems through using economic, social and environmental indicators, and not through monetary units alone. Recognizing the limits of CBA and accepting that incommensurability should be considered for decision-making issues, MCA was identified as an alternative method. MCA is in a sense, very close to CEA but involves multiple indicators of various units (Kapp, 1970; O'Neil, 1997; Foster, 1997; Martinez-Alier et al., 1998). It has been designed to encompass multiple dimensions of decision-making as part of the sustainability principle (Shmelev, 2010).

---

\(^{19}\) The Kaldor-Hicks criterion is a measure of economic efficiency that captures some of the aspects of Pareto efficiency but also considers situations where there can also be losers. While Pareto recognizes only “win-win” scenarios, the envisaged policy can be considered as good using the Kaldor-Hicks criterion as long as the winners can compensate the losers while still enjoying net benefits (Pearce et al., 2006).
In practice, MCA involves: i) identifying the decision criteria through stakeholders’ analysis; ii) identifying different attributes to be compared; and iii) integrating all relevant decision criteria. The indicators are measured by ranking of stakeholders. The ranking enables a prioritization of the sites to be protected.

2.4.4 Synthesis
The methods presented above differ in that they employ different ways to integrate the costs and benefits of conservation. CBA and MCA are the two that are most comprehensive. While MCA could even have a higher degree of comprehensiveness with regards to questions of efficiency and equity (which are barely considered in CBA), the CBA approach seems more useful for the valuation of conservation benefits for two main reasons:

- The valuation is most often carried out after the protected site has been created; there are therefore no alternative scenarios to be considered on these sites (except options for reinforcing/relaxing the protected site regulatory framework); and
- The arguments for decision-making are likely to be more powerful if presented in monetary terms, rather than multi-dimensional indicators; local managers and national decision-makers are likely to better use arguments for conservation if these are expressed in monetary units (as they are then able to compare these to other sectoral policies).

Hence, valuing the net benefits of existing protected sites through a CBA approach involves a comparison of the values of one area before and after the establishment of the protection. Figure 2-5 below proposes a representation of both unprotected (left) and protected (right) sites. For the same unit of surface area of ecosystem the figure shows:

- The diminished extractive use value that accrues due to no-take policy and/or limitations on industrial and commercial uses;
- The more important indirect and distant\(^\text{20}\) use values in the MPA thanks to more pristine ecosystems (and thus better ecosystem functions delivery); and
- The increased non-use value in the MPA thanks to local populations’ willingness to maintain the area in good health for future generations.

\(^{20}\)Distant use values are the indirect use values that arise from the site to the neighbouring areas (such as downstream water services values).
The difference between the TEV of the unprotected and the protected site is the benefits of conservation. The costs of conservation within the protected site are pictured below the X axis. These costs should be compared as a part of the cost-benefit analysis. In this case costs seem equal to benefits, which produce no net benefits from conservation.

**Figure 2-5: Calculation of the net benefits of a protected site for one ecosystem**

Source: adapted from Pagiola, 2004

The comparison between unprotected and protected sites enables the calculation of the increase or decrease of each of the TEV components. These are pictured on the Figure 2-6. Pagiola et al. consider that the reduction of extractive uses in a protected site can be considered as the opportunity costs of foregone benefits.
These figures depict an example of a protected site that has enforced limitations on extractive uses and, by effective management of ecosystems, has led to a better health status of the ecosystem (i.e. increased indirect use values) and an improved perception by populations of their ecosystems (i.e. increased non-use value). It is to be noted that opportunity costs associated with the implementation of protection policy (decreased extractive uses) are very difficult to measure, especially in data-poor situations.

2.5 Conclusion of the chapter
This chapter has provided an overview of the literature on the need to protect marine and coastal biodiversity, including through the development of MPAs. It has also presented a historical and epistemological context relating to the economics of natural capital over time, from classical economics to the controversy between environmental economists and ecological economists, and the recent development of the economics of ecosystems and biodiversity. It has also reviewed the various methods used for the valuation of conservation benefits on two levels:
- At the valuation technique level, it has reviewed and identified the method to be used to value each of the components of the TEV
- At the valuation of conservation level, it has reviewed and provided a synthesis on the method to assess the net benefits of conservation.
3 Chapter 3: case study on a sample of West African MPAs

This chapter sets the context for the study sites. It first presents the three ecoregions of West Africa. It then details the seven ecosystems found within these ecoregions, their geographical coverage along the coast of West Africa, associated ecological functions and health statuses. The chapter also introduces the five MPAs of the sample and the comparison areas for each MPA, which are used for the valuation of conservation benefits. It then provides the ecosystem surface areas calculated for each studied site. It also gives further details about the calculation methods applied to this case study. It then details the socioeconomic activities that prevail along the coast of West African and in the studied MPAs and CAs in particular. Finally, it presents the design and method of the survey carried out to collect data.

3.1 The three ecoregions in West Africa and their related ecosystems

The coastal zone of the Coastal and Marine Regional Programme (CMRP)\(^{21}\) extends from Mauritania to Sierra Leone and is over 3,200 kilometres long. Three major marine ecoregions are encountered: 1) an ecoregion dominated by upwelling\(^{22}\) along the coasts of Mauritania and northern Senegal; 2) a second composed of estuaries and mangroves that ranges from central Senegal to Sierra Leone at its southern perimeter; and 3) a third ecoregion consisting of the volcanic archipelago of Cape Verde. The presence of an "upwelling" and the continuous provision of nutrients from large estuaries, make this region one of the most productive in the world in terms of marine biomass.

---

\(^{21}\) The CMRP is a regional programme that covers marine conservation initiatives from Mauritania to Sierra Leone and aims to advise on the implementation, operation and monitoring of such initiatives in the region.

\(^{22}\) Upwelling: deep ocean waters rich in nutrients rise to the surface along the coast.
The Senegalese-Mauritanian ecoregion (1) is characterized by an arid climate. This ecoregion is made up of a variety of ecosystems: beaches, seagrass meadows, rocky bottoms, mud flats in the delta areas, and mangroves. The beach ecosystem is the most represented. Beaches stretch along much of the coast of Mauritania and the northern part of Senegal. They are usually accompanied by a dune. In some places, especially in the north of Mauritania, the coastline is rocky. In other places, and especially in estuaries and mouths of rivers, mangroves can be found (from extended hyper-salinized land that attests to the former presence of mangroves – known as “tannes” to *Avicennia* and *Rhizophora* mangroves). Mangroves generally coexist with the dune, off the coastal strip. Mangroves are present from Tidra Island in the Bay of Arguin in Mauritania to northern Senegal, but they are more abundant in southern Mauritania and northern Senegal. Their area is constrained by the lack of nutrients from rivers in the Arguin region, in comparison to the southern part of the region (Hughes and Hughes, 1992).
The ecological functions provided by these ecosystems are diverse. Seagrass meadows grow on sandy and sheltered areas. They stabilize soft substrates, promote oxygenation of the water and are an important breeding area and nursery for marine species and migratory birds (which over-winter in the area). Large areas of wetlands and mudflats help ensure nutrient cycling. They contribute to the capture of contaminants; they ensure water purification and are also a key area to most marine species in the area, as spawning and nursery areas, and as feeding grounds for birds. Mangroves provide a wide variety of ecological functions: they make a large contribution to water purification and decontamination of organic waste, support the retention of sediments, reduce coastal erosion, serve as a nursery for larvae and juveniles of marine species and capture atmospheric carbon.

![Examples of upwelling ecoregion scenes: Saint-Louis beach (Senegal), the Senegal river estuary (Senegal) and a rocky coast in Bank of Arguin (Mauritania) (credits: T. Binet)](image)

The estuaries and mangroves ecoregion that ranges from the Saloum delta south of Dakar to Sierra Leone is a huge deltaic and estuarine complex (Cormier-Salem, 1994; 1999). Sometimes called "Rivers of the South" because of the many rivers that flow into the ocean, it consists of a series of coastal plains and a river system that is wide open to the ocean. Seawater can reach the hinterland waters up to a hundred kilometres from the coast thanks to a high tide amplitude exceeding four metres. The coastline consists of a great complex of small channels. This has the effect of increasing the length of the coast: the coast of Guinea-Bissau extends linearly for 264 kilometers, but extends to more than 1,000 km if estuaries and coastal islets are included (Hughes and Hughes, 1992).
On the river banks of channels and up to the limit of high tide, a vegetation of mangroves (*Rhizophora* and *Avicennia*) is encountered. This is the mangrove ecosystem *sensu stricto*. The mangrove ecosystem is composed of halophilic tropical plants and distributed up to 50 km inland along rivers. The accumulation over time of sedimentary deposits carried by flooding and captured by mangrove aerial roots system has resulted in the creation of a subtidal mudflats rich in organic elements. The coast also has many beaches and sandbars (sometimes covered with seagrass) on the intertidal (located between high and low tide levels) or subtidal zone. Along the coast, rocky islets are present, some of which are visible at low tide. Estuaries and channels are very important for hydrodynamic and ecologic functions: they largely contribute to nutrient cycling; the retention of contaminants and water purification; they are a support area for marine life (spawning, nursery and feeding). Rocky bottoms contribute to shore stabilization by allowing spits to form off the coast which can help protect beaches.

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*Figure 3-3: Typical scenes of a mangrove and estuary ecoregion in Guinea-Bissau: Rio Cacheu (left and centre), Bijagos archipelago (right) (credits: T. Binet)*

Bathed by the Canary Current, the Cape Verde volcanic archipelago ecoregion is unlike any other ecoregion in West Africa. The Cape Verde archipelago consists of eight islands, located approximately 600 km west of Senegal. The coasts of these islands are mostly made of cliffs and rocks of volcanic origin, which alternate with beaches (Schwartz, 1992). The beaches are a nesting site for several species of sea turtles. On sandy bottoms, seagrass meadows include varieties of *Zooestera* sp., marine phanerogams and kelp sp. Cape Verde is also considered as one of the 10 hotspots reefs in the world (Spalding et al., 2001). There are no

23The description of the mangrove ecosystem *sensu stricto* is seen as important in the characterization of ecosystems in West Africa. Many environmentalists however solely associate mangrove ecosystems with estuarine ecosystems. Here we distinguish these two ecosystems given the different services they provide.
bio-constructed coral reefs on the archipelago, but there are several coral communities that have developed on rocky bottoms. Further offshore, the very small size of the continental shelf severely limits the development of wildlife underwater (Ibid).

Figure 3-4: Typical scenes of the Cape Verde archipelago ecoregion: Santa Luzia and Santo Antao
(credits: Hellio/Van Ingen and T. Binet)

3.2 Characterization of coastal and marine ecosystems in West Africa

This section provides further detail on the ecosystems encountered in the three ecoregions which are the basis of this study. These ecosystems are hereinafter referred to under the generic term MCEs for "marine and coastal ecosystems". The Table 3-1 describes the MCE of each of the three West African ecoregions. The presence of each ecosystem within the ecoregion is shown by the red colour (a white cell shows the absence of the ecosystem in the considered ecoregion).

Table 3-1: Identification of coastal and marine ecosystems in the three West African ecoregions

<table>
<thead>
<tr>
<th>Marine ecoregion</th>
<th>Upwelling ecoregion in Mauritania and Senegal</th>
<th>Estuaries and mangroves ecoregion in Guinea and Guinea-Bissau</th>
<th>Cape Verde volcanic archipelago</th>
</tr>
</thead>
<tbody>
<tr>
<td>Mangroves</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Estuaries and channels</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Mudflats</td>
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<td></td>
<td></td>
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<tr>
<td>Beaches</td>
<td></td>
<td></td>
<td></td>
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<tr>
<td>Seagrass meadows</td>
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</tbody>
</table>
For each ecosystem listed below, I have first provided a description of the ecosystem, its associated ecological functions and uses. I have then presented an overview of its health status in the specific ecological context of West Africa.

### 3.2.1 Mangroves

"Strictly speaking, mangrove vegetation designates certain tropical coastal plains region in which *Rhizophora* sp. are the dominant species. In the broadest sense they mean intertidal mudflats of the intertropical zone colonized by mangrove forests and salt marshes as well" (Cormier-Salem, 1994, p. 233). Mangrove ecosystems are very fragile, being a transition area between the sea water and coastal zone, periodically flooded by the tide and seasonally by rain flooding. Located in brackish water at the mouths of rivers, these ecosystems can extend in places up to fifty kilometres inland where the tidal influence is still marked. Mangrove forests cover approximately 100,000 km$^2$ worldwide (Blasco, 1991). West African mangroves are less diverse than those in the Indian Ocean, but are the largest on the African continent. It is present over large areas in Guinea-Bissau and Guinea, and also in the Gambia, Senegal and Sierra Leone (Stuart et al., 1990; Fisher and Spalding, 1993; Hughes and Hughes, 1992). It is also found in Mauritania, where it is being restored in the Diawling National Park along the Senegal River (Cormier-Salem, 2011). Mangroves are an important support to marine and terrestrial biodiversity, being breeding and feeding sites for many marine species, and an important support to fisheries (Baran, 1999).

In spite of the physiological adaptations of plant species found in mangroves that makes them so special, species diversity is very limited: only six *Rhizophora* species are present in the mangrove ecosystems:

- Three *Rhizophoraceae*: *Rhizophora racemosa* (Meyer 1818), *R. mangle* (L. 1753) and *R. harrisonii* (Leechman 1918) (sometimes considered a hybrid between the two preceding species)
- An *Avicenniacea*: *Avicennia germinans* (L.) (Stearn 1958) (syn. *A. nitida*; *A.africana*);
- Two Combretaceae: Languncularia racemosa (L) (Gaertn.f. 1805), Conocarpus erectus (L. 1753).

Only these species can survive in such extreme conditions of salt, water level variability and sediment inputs. Figure 3-5 below illustrates the overall structure of such an ecosystem. I notice the succession of stages from Rhizophora to grass and rice, and also “tannes” where rice cultivation was abandoned due to excessive soil salinization.

![Figure 3-5: Representation of West African mangrove ecosystem](image)

Source: adapted from Sow et al., 1994

Ecosystem functions of mangroves are very important for the maintenance of physical, biological and ecological systems along the coast. They form a natural barrier along the coast and thus provide shoreline protection to extreme events such as storms and flooding. They also serve as a physical barrier against the everyday erosion caused by tidal stream and river flow. In addition, they have a high capacity for absorbing pollutants (heavy metals, toxic substances), organic and inorganic materials in suspension. The root systems of the mangroves slow the flow of water, thus facilitating the deposit of sediments. The pollutants are thus deposited in sediments or are incorporated into the structure of sedimentary layers. The roots of mangroves also provide shelter and protection to juvenile vertebrate species, which find sufficient food in quality and quantity in mangroves (Blaber, 1980; Thollot, 1989). Mangroves also play an important role in carbon sequestration (Wells et al.,
which has resulted in many reforestation campaigns supported by private companies under the REDD+ mechanism.24

Salt marshes, their sizes dependent on the gradient of salinity and tidal flooding frequency, represent the cover behind the first layer of Rhizophora-dominated mangrove. Salt marshes include crassulescent cover of Sesuvium portulacastrum and Philoxerus vermicularis, Eleocharis sp., Paspalum vaginatum and populations made up of a mixture of Poaceae and Cyperaceae (Cyperus articulatus, Scirpus cubensis, and Imperata cylindrica). These salt marshes extend up to a bank that marks the transition to the continental area.

In Guinea-Bissau, mangroves covered 276,000 ha in 1980, but had declined to 210,000 ha by 2005. This represents a decrease of almost 24% in 25 years (FAO, 2007). As 45% of the population are dependent on these natural formations (Diombera, 2004), the reduction in mangrove surface area often has disastrous consequences for the environment and local communities.

Disturbances in mangrove ecosystems are mainly caused by anthropogenic factors (Kathirsan and Bingham, 2001). The human pressure on mangroves is of two types: the exploitation of mangroves for timber and deforestation for agriculture use (Cormier-Salem, 2000, 2004). Rhizophora racemosa and R. harrisonii are exploited for timber in the region, second only to forest areas cleared for palm oil cultivation. The mangrove is the prime woody material used for construction and the primary source of energy for domestic use, production of salt, and fish smoking. Mangrove wood is not only for local use but is used in urban centres in Guinea and Guinea-Bissau, and can also be exported to Senegal (Dakar)26 (Rivain, 2008).

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24REDD+ (“Reducing Emissions from Deforestation and Forest Degradation”) uses market and financial incentives in order to reduce the emissions of greenhouse gases from deforestation and forest degradation. REDD+ complements the REDD mechanism designed to promote sustainable management measures such as biodiversity conservation and poverty alleviation (UN, 2012).

25 This variety of plants can be defined as fleshy halophytes; they retain water and can grow in salty conditions.

26 As an example, the consumption of firewood in the city of Bissau and its suburbs (165,000 inhabitants, 16% of the national population) varied between 40 and 45 tonnes per day in 1990, equivalent to 14,600 to 16,500 tonnes per year or about 100 kg/capita/year (Diombera, 2004).
Mangrove wood-cutting: domestic and professional practices

Leciak (2006) warns of the necessary differentiation between commercial and traditional mangrove wood-cutting. According to him, the impact of commercial cutting increases as demand from urban areas rises. Two categories of commercial actors may also be distinguished: on the one hand, the professional cutters who sell wood as lumber or firewood, and charcoal burners who produce charcoal from mangroves. On the other hand, villagers exploit mangroves in a much more sustainable way, enabling mangroves to regenerate. Professional cutters concentrate on high density mangroves, but this exploitation has substantial consequences on the regeneration of mangroves. The trees of great size, sometimes reaching 20-25 meters high, are ancient formations. The majority of them are located on dried out mud, extracted from tidal movement daily. The substrate here has become unfit for germination as seed survival is dependent on fresh mud deposits. Cutting those trees therefore prevents the area from being repopulated later. Furthermore, these trees contribute strongly to regeneration since they produce a lot of propagules (dissemination organ of Rhizophora). Cutting these trees therefore not only threatens local areas of mangrove but also jeopardizes mangrove extension. Thus, the commercial cutting practiced in fishing camps, salt production sites, and urban peripheries in Guinea, Guinea-Bissau and Senegal (Senegal River and Saloum Delta) tends to be much more detrimental than traditional cutting and should be limited.

In addition to wood-cutting, large areas of mangroves have been converted into rice fields by local populations (Cormier-Salem, 1999; 2004). Mangrove rice production sees farmers clear the Avicennia and dig channels for water circulation. The conversion is irreversible and rice production continues each year on the same field (Leciak, 2006). After a few years, these practices lead to a decrease in productivity, and farmers abandon the field and convert a new area of mangrove nearby. Abandoned fields become “tannes”, where new mangrove growth can take up to several decades to occur.
3.2.2  Estuaries and channels

An estuary is broadly defined as the coastal area where a river meets the sea and where freshwater and saltwater mix. Specific definitions vary, however, from one author to another. Fairbridge (1980), for example, defined the estuary as an encroachment of the sea into a river bed which extends to the upper limit of the tidal influence. He distinguishes three areas: the lower marine estuary or open sea; the intermediary estuary (where intense mixing of fresh and marine waters occurs); and the upper estuary (characterized by fresh water). The gradient of salinity in these estuaries is highly dependent on upstream flows. Heavy rainfall is the main driver of changes in salinity and turbidity. Sediments from inland waters are transported to the estuary, where they disperse in the coastal area under the influence of tides and trade winds. These sediments largely contribute to the formation of large areas of mudflats in Guinea (around the islands of Tristao for example) and Guinea-Bissau (all around the Bijagos archipelago). Waves and tidal streams sort the sediments by size from sand to silt along the coast: sands are deposited on the river beds while silt is transported up to several miles away at sea before being deposited in the mudflats. In coastal waters, the plankton feed on these sediments and provides a biological filter for waters from the estuary (Bangoura, 1999), which in turn supports the development of small pelagic species (the most targeted species by fisheries). Estuaries are thus important to coastal fisheries for small pelagic species, not only for their feeding but also for their reproduction (Robertson and Duke, 1987; 1990a; Robertson and Blaber, 1992; Bangoura, 1999).
Within secondary channels of the estuarine complex, sediment deposition is dependent on the curve of channels. Sediment deposits increase on one bank while it reduces on the opposite bank (Wolanski et al., 1988). The dense mangrove ecosystem develops on the convex side, while it reduces on the concave side (Figure 3-7).

![Figure 3-7: Channels in an estuarine complex in Guinea; differences in sediment deposits are visible from one bank to another (left) (credits: Hellio/ Van Ingen)](image)

Estuaries and channels are threatened by sediment inputs from rivers upstream that are loaded with chemical pollutants of human origin. In places, estuaries are threatened by the developments of hydroseres and due to the construction of dams, such as Diama on the Senegal River (Cabo, 2010; Cormier-Salem, 2011). Sedimentation and pollution can lead to the development of “green tides” (excessive growth of algae resulting in eutrophication). These “green tides” can lead to the creation of anoxic waters where most forms of marine cannot survive. Estuaries and channels can prevent eutrophication, but sometimes their capacity to do so is exceeded as a result of excessive pollution. In this case, the survival of the estuary ecosystem is threatened (mangroves, mudflats, seagrass meadows and rocky bottoms).

### 3.2.3 Mudflats

Mudflat development often results in the clogging of small bays with fine sediments. Mudflats are subject to the tidal stream and can move under its effect. The various types of mudflats include: i) estuarine mudflats or downstream rivers, subdivided into bare mudflat downstream (slikke) and upstream (schorre) of the estuary; ii) the coastal mudflats that are present on the foreshore; and iii) subtidal mudflats (or offshore mudflats) that result from

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27 Hydrosere is the primary level of conversion of water body and its community into a land vegetation community.
the accumulation of sediments offshore. Animal life is concentrated in the top ten centimetres of mud flats, where oxygen is still available (Cormier-Salem, 1999). Below this only anaerobic bacteria can survive. They release ammonia and hydrogen sulphide, giving the characteristic odour of mudflats.

![Image of intertidal mudflats]

**Figure 3-8 : Intertidal mudflats at low tide: landing site for small-scale fisheries (Tristao, Guinea) (left) and mudflats as a source of food for birds (Rio Cacheu, Guinea-Bissau) (right) (credits: T. Binet)**

Estuarine mudflats are covered and exposed every twelve hours. They are mainly composed of soft muds of smooth appearance and without any vegetation cover. They include a rich fauna of bivalves and small gastropods that are an abundant source of food for many birds at low tide. At high tide, it is the fish and other marine life that in turn feed in the area. The *schorre* is partly covered at spring high tides, and is characterized by low halophytic vegetation and dense distributed vegetation. This area is mostly a resting area and breeding ground for birds.

Coastal mudflats are gradually colonized by mangroves. Offshore mudflats extend over large areas and are an ideal spot for the growth and development of juvenile demersal fish such as catfish. When these mudflats are uncovered at low tide, sea birds come to feed and to seek refuge, safe from predators. Coastal mudflats comprise huge concentrations of sediments carried by river flows on which the shrimps can feed. The existence of an empirical relationship between the surface of estuarine mudflats and commercial catch of adult shrimp has been demonstrated in marine mudflats in Australia (Vance et al., 1990). Shrimps come into the deltas at least once a year for spawning and next generation
individuals leave at the juvenile stage (it is these juveniles that are often caught in the coastal mudflats by local fishermen).

Mudflats can store pollutants, such as heavy metals from industrial activities, and coastal and urban waste. They play a very important role in waste treatment by locking up such pollutants. However, when pollution levels are too high (for example near urban areas or plants discharging their effluents directly into the sea) mudflats become saturated with pollutants that threaten the survival of the fauna that depend on it (shellfish and demersal fish species) (Cormier-Salem, 1999). Often these resources are critical to the food security of local populations, and the high concentration of pollutants makes the species harvested unfit for consumption.

The extant literature does not provide any detailed information on the health of mudflats in West Africa, in particular the concentration of pollutants in the mudflats, though it is thought likely that pollution levels will be significantly higher closer to the major urban and industrial centres.

3.2.4 Beaches

Beaches are present all along the West African coast in the upwelling ecosystem of northern Senegal and Mauritania. They are also present in Cape Verde, sometimes over large areas (Figure 3-9). They are much smaller in size in mangrove and estuarine ecoregions. Due to the strong currents in the area, beaches sand banks are in perpetual motion and the morphology of these ecosystems changes continually. Some beaches are disappearing due to erosion, while others close to rocky spits grow because of accumulation.

![Figure 3-9: Beaches in Langue de Barbarie national park (Senegal) and Praia Grande, east of Sao Vicente island (Cape Verde) (credits: T. Binet)]
Beaches are nursery sites for small pelagic species. They are also a key nesting ground for sea turtles throughout the West African coast, from Senegal to Guinea and Cape Verde. They are also the main settling ground for small-scale migrant fishermen that fish all along the coast of West Africa (Binet et al., 2012). They play a key role in fishing business since they are not only the main landing sites for fishermen, but are also used for the installation of fish smoking huts, fishermen’ houses and various other businesses associated with fishing (Figure 3-10). A number of fishing activities are performed directly on the beach, such as beach seining, and shellfish harvesting.

As beaches are the main sites of fishermen’s activities, they are widely exposed to various types of degradation, which directly (or indirectly) threaten species and habitats. Effluents from fishing camps are thought to be a major source of marine pollution. Most camps located on the beach have no sewage or waste treatment facilities: everything is tipped into marine waters or in the river before reaching the sea. This degradation moreover is intensifying as the villages and settlements located on the beaches grow.

Other forms of degradation exist, in relation again to the fishing camps. As is the case in Guinea-Bissau, northern Guinea and Sierra Leone, intensive smoking of fish requires large quantities of wood. According to local experts, this has led to the massive clearing of mangroves around fishing camps. This deforestation not only reduces the mangrove surface and the proportion of Rhizophora, but also accelerates the coastal erosion of beaches that do not have spits. This erosion can cause beaches to disappear and so increases the risk of displacement for the people based on such beaches. Such forced migration is being observed in southern Sierra Leone, where fishermen settled on Plantain Island are preparing to leave the island, due to severe erosion and rising waters in their camp (Figure 3-10).
3.2.5 Seagrass meadows

While many studies have been devoted to mangrove ecosystems in Africa, few have focused on other ecologically important ecosystems such as seagrass meadows (Duarte et al., 2008). Seagrass areas have been identified in several places along the coast: near mangrove areas or close to rocky beds (Figure 3-11). Seagrass meadows are probably one of the most important coastal systems in the world, encompassing 177,000 km² (Green & Short, 2003). They perform a large number of ecological functions including coastal protection, water treatment, and the regulation of nutrients, and also play a key role as nurseries and refuges for many marine species.

Seagrass meadows are highly productive ecosystems. In West Africa, it is recognized that Mauritania has the largest areas of seagrass beds (Cunha and Araújo, 2008). Most seagrass meadows are located in the National Park of Banc d'Arguin (Figure 3-11). The most important seagrass species found in the Park is *Cymodocea nodosa*. Other studies within the Park ecosystems have confirmed the presence of *Halodule wrightii* and *Zostera noltii* meadows, which are present in the intertidal zone of the Banc d’Arguin (Cunha and Araújo, 2008; Green & Short 2003; Wolff et al., 2006). In Senegal, *Cymodocea* sp. and *Halodule wrightii* have been observed in some sandy areas of protected bays around Dakar, along the Petite Côte south of Dakar, and around Sarène, in the Joal Fadiouth Bamboung-Sourou region. However, the southern limits of these species’ distribution are not clearly defined yet.

![Figure 3-10: Fishing camp settled on Plantain island (Sierra Leone) (credits: T. Binet)](image)

![Figure 3-11: Seagrass meadows at low tide in the Bijagos archipelago (Guinea-Bissau) (left) and flamingos feeding on seagrass in the national Park of Banc d’Arguin (Mauritania) (right) (credits: T. Binet)](image)
In West Africa, as elsewhere, seagrass beds are sites for feeding and nesting for many terrestrial and marine species, migratory or sedentary. Thus, marine turtles of the Atlantic coast of Africa (loggerhead, olive ridley, green turtles, and hawksbill turtles), manatees and many species of migratory birds depend on these seagrass beds for their feeding. Seagrass also protects the shore against the erosion effect of waves and tides, and plays a key role in nutrient cycling and marine biomass production because a large number of fish and shellfish depend on seagrass during their life cycle.

Knowledge about the total surface area of seagrass and its health status in West Africa is extremely limited. Some references are available for the Bank of Arguin in Mauritania (Boely et al., 1978; Cuq, 1993; Faure et al., 2000; Schaffmeister et al., 2006), but there is almost no literature on seagrass in the mangrove and estuarine ecoregions. However, degradation factors for seagrass here are the same as in the rest of the world, the most important factor being hyper-sedimentation. Estuarine sediments that settle on seagrass severely limit their growth and prevent reproduction. In highly eutrophicated waters (as is the case in calm waters not affected by tidal current) macro-algae grow rapidly on the leaves of flowering plants, also limiting their growth. An accurate census of the area covered by seagrass and its related health status is therefore required across the whole region in order to ensure better protection, especially near urban areas where chemical pollution and eutrophication are a major threat to the persistence of such a fragile ecosystem.

3.2.6 Coral bottoms

In West Africa, coral communities are only present in the Cape Verde islands. Despite the fact that this ecosystem is the most studied marine ecosystem, the coral reefs are poorly studied in Cape Verde, and are only described in a very limited number of outdated publications. In 1974, Laborel stated that the environment in the Atlantic does not allow for the formation of "bio-constructed" coral reef macro-structures (Laborel, 1974). This is due to local variations in hydrology, a dry climate and the influence of the cold Canary current that prevents coral reefs from developing. Instead, coral communities have developed on rocky beds that cover most of the surface of the shallow waters in the volcanic Cape Verde islands.
Eight species of corals have been identified in the waters of the archipelago: hydrocoral *Millepora alcicornis*; the stony corals\(^{28}\): *Siderastrea radians*, *Porites porites*, *P. astreoides*, *Favia fragun*, *Schizoculina africana*, *Madrasis pharensis* and *Tubastrea sp.* (Figure 3-12). The presence of such corals have been identified on the island of Sal Palmeira (in the Baia Murdeira and Santa Maria in the south and west of the island), on the western shore of the Bay of Boavista island, in Sao Vicente, and on the island of Santa Luzia (Laborel, 1974). Significant new locations on the area of Pedra Lume on the Island of Sal, and Praia de Lajinha and Salamansa bay in Sao Vicente have also been identified (*Ibid*).

The coral bottom ecosystem is ecologically and biologically important for Cape Verde marine life. It also contributes to the provision of economic activities such as tourism and fishing. Five of the six sea turtles species found on the Atlantic coast of Africa (*Caretta caretta*, *Eretmochelys imbricata*, *Chelonia mydas*, *Lepidochelys olivacea* and *Dermochelys coracea*) feed on coral ecosystems (Figure 3-12). They aggregate on coral communities during the breeding season which reaches its peak from July to September.

Figure 3-12: Example of coral and sponge communities developed on rocky substrate (left) and a Hawksbill turtle (right) (credits: M. Leroux)

Knowledge of Cape Verde’s coral reefs is extremely limited and most references date back several decades. Coral communities have also been reported by diving clubs, but it is difficult to know the diversity, size and also the health of these communities. It is therefore expedient to develop further research on coral reefs in Cape Verde, in particular conducting

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\(^{28}\)Stony corals (as opposed to soft corals) are corals that contribute to reef-building since they have a hard calcareous skeleton that contributes to the formation of reefs.
a thorough investigation on the extent, diversity, threats and management measures required to preserve these coral communities.

Downwind of Santa Luzia, on the east coast of Sao Vicente, a large amount of coral rubble is visible (Figure 3-13). These indicate the important presence of coral communities on this coast, as well as the impact that natural factors may have in the breaking up of coral organisms (by waves, strong currents, or storms).

![Figure 3-13: Pieces of coral on the beach at Calhau, east of Sao Vicente island (Cape Verde) (credits: T. Binet)](image)

3.2.7 Rocky bottoms

Rocky bottoms come in two distinct forms: as banks lying parallel to the coast or as spits positioned perpendicular to the coast. Rocky bottoms that take the form of small underwater walls generally originate from a mixture of gravel, pebbles and sand. They were observed at depths of -15, -30 to -35, -45 and -50 meters in Guinea, Senegal and Mauritania (Domain, 1977). Most large rocky bottoms in the upwelling ecoregion are the result of volcanic activity. They are also found on the steep slopes of the continental shelf borders (as is the case in Kayar, north of Senegal).

The second type of rocky bottoms found in the region consists of discontinuous rocky beds covered with sediment. They correspond to outcrops of a similar nature as cited above, which were called "beach rock" by Domain and Bah (1999) (Figure 3-14).

Rocky bottoms play an important role as refuges for most demersal fish species: *Pagellus coupei*, sea bream or snapper, and *Pagrus ehrenbergi*. Deeper dwelling demersal species found on rocky bottoms include other species of bream, such as deep red sea bream, *Dentex Dentex angolensis* and *macrophthdmus, D. angolensis or D. macrophthalmus* and the white grouper, *Epinephelus aeneus.*
In Cape Verde, rocky bottoms are the most representative ecosystem in coastal waters because of the volcanic nature of the islands (Figure 3-14). They are essential to marine life which can only thrive without sedimentation, since the rivers of Cape Verde provide low sediment inputs, unlike the West African ecoregions on the continent. For this reason, rocky bottoms provide a habitat for most of the species exploited by fisheries: demersal species and populations of crustaceans (lobsters and crabs).

*Figure 3-14: « Beach rock » in the Bijagos archipelago (Guinea-Bissau) (left) and a view of the West cost of Sao Vicente (Cape Verde) (right) (credits: T. Binet)*

The richness of rocky bottoms’ biodiversity is largely threatened by the overfishing of demersal species, the species that attract most commercial interest given their high value (e.g. white grouper, bream, and lobster). Catches of these species decreased by 20 to 40% between 1996 and 2007 (Table 3-2). The figures are even more alarming in the waters of Senegal, as catches of key species such as groupers show a drop of around 80% between 1990 and 2000 (Dahou and Dème, 2002).

**Table 3-2: Change in demersal species catches 1996-2007**

<table>
<thead>
<tr>
<th>Species</th>
<th>Fleets*</th>
<th>Catches for period 1996-2007</th>
<th>Exploitation level estimates</th>
</tr>
</thead>
<tbody>
<tr>
<td>Demersal species (e.g. grouper, snapper, sole)</td>
<td>CV, Gui, Ma, Mo, Sen, Sp</td>
<td>-26%</td>
<td>Moderate to intense overexploitation</td>
</tr>
<tr>
<td>Cephalopods (octopus mainly)</td>
<td>Ma, Mo, Sen, Sp</td>
<td>-31%</td>
<td>Moderate to intense overexploitation</td>
</tr>
<tr>
<td>Crustaceans (lobster, crabs, shrimp)</td>
<td>Fr, It, Mo, Sen, Sp</td>
<td>-38%</td>
<td>Fully exploited</td>
</tr>
</tbody>
</table>

82
Unlike beach ecosystems, mudflats or mangroves, rocky bottoms are less subject to erosion, except for the very friable oxidized rocks which are subject to severe erosion and so threaten the populations living on the coast. Rocky bottoms may also suffer from significant sediment deposits close to river mouths that threaten the survival of communities installed on this ecosystem.

3.3 Valuing ecosystem services in West Africa

When applying the approach detailed in the literature review chapter to West African ecosystems, it is first important to define the various values according to the TEV framework that the ecosystems listed above provide. The use values in particular are not the same across each ecosystem in terms of the ecological functions provided and the related uses by populations. The non-use values are considered to be homogeneous across ecosystems that form the coastal habitats in MPAs and CAs. The details of use values for each ecosystem are as follows:

- **Direct use values:**
  - 1) Estuaries: used as fishing sites for human consumption, medicinal use;
  - 2) seagrass: used as fishing sites for human consumption, pharmaceutical use;
  - 3) mangroves: used as fishing sites for human consumption, pharmaceutical use, exploitation of wood for domestic and commercial uses, production of salt, hunting;
  - 4) mudflats: used as fishing sites for human consumption;
  - 5) beaches: used as fishing sites for human consumption, pharmaceutical use;
  - 6) rocky bottoms: used as fishing sites for human consumption, pharmaceutical use;
  - 7) coral bottoms: used as fishing sites for human consumption, for ornamental purposes or for present and future pharmaceutical use.

- **Indirect use values:**
1) Estuaries: contribution to the formation, maintenance and protection of
beaches and coastal systems, regulation of coastal water quality by filtering
or fixing sediments and pollutants from land; support to marine
biodiversity;

2) Seagrass: contribution to the formation, maintenance and protection of
beaches and coastal systems; climate regulation and sequestration of
atmospheric carbon, regulation of coastal water quality by filtering or fixing
sediments and pollutants from land; support to marine biodiversity through
their nursery and refugee role for numerous species of fish and shellfish;

3) Mangroves: contribution to the formation, maintenance and protection of
beaches and coastal system, climate regulation and sequestration of
atmospheric carbon, regulation of coastal water quality by filtering or fixing
sediments and pollutants from land; support to marine biodiversity through
their nursery and refugee role for numerous species of fish and shellfish;

4) Mudflats: regulation of coastal water quality by filtering or fixing
sediments and pollutants from land; support to marine biodiversity;

5) Beaches: regulation of coastal water quality by filtering or fixing
sediments and pollutants from land; support to marine biodiversity;

6) Rocky bottoms: contribution to the formation, maintenance and
protection of beaches and coastal systems; support to marine biodiversity;

7) Coral bottoms: contribution to the formation, maintenance and protection
of beaches and coastal systems; climate regulation and sequestration of
atmospheric carbon.

3.4 Calculation method for the conservation benefits in the West African
MPA
When data are lacking, it is not possible to carry out a valuation of the ecosystems for the
same area before and after the establishment of the MPAs in order to estimate the benefits
brought by conservation. The alternative option is to compare the value of MCEs
simultaneously in the MPAs and in neighbouring sites that are not protected. This method
enables us to estimate the conservation benefits within an MPA as opposed to an
unprotected site during the only period of the research. As a consequence, the site elected
for comparison will present comparable geomorphological, socioeconomic, ecological characteristics to the MPA selected (see section 3.2 in chapter 1).

In practice, the process for valuation of conservation of MCEs in MPAs will be as follows:

1. estimating the TEV per unit area for each of the studied ecosystems in the MPA and in the comparison site;
2. multiply the TEV per unit area by the surface area of each ecosystem in the MPA;
3. multiply the TEV per unit area for unprotected ecosystems by the same surface area of ecosystems in the MPA;
4. the two aggregated values will then be compared for the same surface area of ecosystems; and then
5. compared with the difference in the costs associated with the MPA (MPA management costs only).

The question of discounting and time is also of importance in my study. Most CBAs use discounting in their methodology and this can be tested in the specific case of the West African MPA. However, the use of discounting is subject to critiques, especially with regards to the choice of the discount rate (TEEB 2010). A great variety of discount rates, including zero and negative rates, can be used that highly influences the final results of the economic valuation. For this reason, discounting is discussed in the TEV results chapter and I prefer to provide all costs and benefits on an annual basis, in order to enable an instant comparison of benefits of management schemes applied to sites. The annual CBA results are therefore expressed in euros for the year 2013.

3.5 The choice of Marine Protected Areas (MPAs) and comparison areas
The economic valuation of marine and coastal ecosystems in West Africa was carried out on a sample of MPAs representative of the region. The method used is based on a comparison of the ecosystem value in an MPA with the value of the same ecosystem outside the MPA (what is called “comparison area” below). The following section describes the MPAs that have been chosen for this study. For each MPA, details of comparison area are also presented.
3.5.1 Choice of MPA

The choice of MPAs can be considered as representative of West Africa if it covers the three ecoregions described above. For this reason the MPAs selected for this study are:

- For the upwelling ecoregion: the National Park of the Langue de Barbarie, north of Senegal;
- For the estuaries and mangroves ecoregion: the Mangroves National Park of Rio Cacheu and the Urok Community MPA in Guinea-Bissau; and the Tristao MPA in Guinea; and
- For the ecosystem volcanic archipelago: Santa Luzia MPA.

The Table 3-3 summarizes ecosystems found in these MPAs.

<table>
<thead>
<tr>
<th>MPA Ecosystems</th>
<th>National Park of the Lange de Barbarie (Senegal)</th>
<th>Santa Luzia (Cape Verde)</th>
<th>Communitarian MPA of Urok (Guinea-Bissau)</th>
<th>Mangroves Park of Rio Cacheu (GB)</th>
<th>Tristao MPA (Guinea)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Estuaries and channels</td>
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<tr>
<td>Seagrass meadows</td>
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<tr>
<td>Mangroves</td>
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<tr>
<td>Mudflats</td>
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<td>Beaches</td>
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<tr>
<td>Coral bottoms</td>
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<td></td>
<td></td>
</tr>
<tr>
<td>Rocky bottoms</td>
<td></td>
<td></td>
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<td></td>
<td></td>
</tr>
</tbody>
</table>
The MPAs selected for this evaluation are spread across the three ecoregions of West Africa (Figure 3-15). The Figure 3-4 provides more detailed data on the MPAs in terms of date of creation and economic activities.

![Map of West Africa and details of sampling of the MPAs surveyed](image)

*Figure 3-15: Map of West Africa and details of sampling of the MPAs surveyed*

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29 The different maps presented in the report have been prepared by Vincent Turmine in French. It has not been possible to get these maps converted into English at the time the thesis was prepared.
**Table 3-4: Key information on the selected MPA**

<table>
<thead>
<tr>
<th>Ecosystems</th>
<th>National Park of the Langue de Barbarie (Senegal)</th>
<th>Communitarian MPA of Urok (Guinea-Bissau)</th>
<th>Mangroves Park of Rio Cacheu (GB)</th>
<th>Tristao (Guinea)</th>
<th>Santa Luzia MPA (Cape Verde)</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Geo-data</strong></td>
<td>15°55 N, 16°30 W</td>
<td>Between 15°35′12″-15°35′10″N and 11°51′40″-2°02′23″W</td>
<td>Between 12°10′-12°25′N and 15°55′-16°27′W</td>
<td>Between 15°25′-14°50′W and 11°-10°45′N</td>
<td>16°45′41″ N and 24°44′38″ W</td>
</tr>
<tr>
<td><strong>Other protection statuses</strong></td>
<td>One part of this MPA is also included in the Saint-Louis MPA (15°50′.5 - 15°58′.5 N and 16°31′.5 - 16°48′.5 W) created on November 2004 by decree 2004-1408; however, this MPA is not effective</td>
<td>Urok MPA is part of the Bolama-Bijagos Biosphere Reserve</td>
<td>Rio Cacheu is also a Ramsar site (since June 2002)</td>
<td>Tristao has also been a Ramsar site since 1992</td>
<td>No</td>
</tr>
<tr>
<td><strong>Managing authority</strong></td>
<td>Ministry of Environment and protection of nature – National Park Directorate</td>
<td>Institute for Biodiversity and Protected Areas (IBAP), and local populations represented by the Management Committee of Urok (CGU).</td>
<td>Institute for Biodiversity and Protected Areas (IBAP)</td>
<td>Ministry of Environment/National Protected Areas Directorate (DNAP) and Ministry of Fisheries/National Centre of Fisheries Sciences of Boussoura (CNSHB)</td>
<td>Environment General Directorate</td>
</tr>
<tr>
<td><strong>MPA populations</strong></td>
<td>On the sandbar between ocean and Senegal estuary there are only two tourist camps; on the other side of the estuary the MPA encompasses 33 villages.</td>
<td>Total population estimates are 3,080 inhabitants (internal census 2007) including 1,991 on Formosa, 562 on Nago, 257 on Chedià and distributed in 33 villages.</td>
<td>Around 7,120 inhabitants on the northern part of Rio (27 villages); around 830 inhabitants on the southern part (14 villages)</td>
<td>7,000 inhabitants over 4 districts and about 30 villages on the islands</td>
<td>Uninhabited island; island close to Santa Luzia: Sao Vicente 67,163 inhabitants in 2000</td>
</tr>
<tr>
<td><strong>Area</strong></td>
<td>20 km²</td>
<td>545 km²</td>
<td>886 km²</td>
<td>620 km²</td>
<td>35 km²</td>
</tr>
<tr>
<td><strong>Economic and subsistence activities</strong></td>
<td>Subsistence and commercial fishing, salt production in mangroves, ecotourism (birdwatching, boat tours in the Park), beach tourism (e.g. swimming, windsurfing), agriculture.</td>
<td>Subsistence and commercial fishing, subsistence shellfish picking, wood collection for fish and oyster smoking, oyster farming on ropes in mangroves.</td>
<td>Subsistence and commercial fishing, palm oil production, agriculture, salt production, mangrove (commercial and for domestic use) wood-cutting</td>
<td>Subsistence and commercial fishing, fish smoking, palm oil production, agriculture, salt production, mangrove (commercial and for domestic use) wood-cutting</td>
<td>Subsistence and commercial fishing, tourism (boat tours and diving), sport fishing</td>
</tr>
</tbody>
</table>
3.5.2 Choice of comparison areas

Areas not protected by an MPA which are to be compared with MPAs are called “comparison areas” (CAs). These are chosen according to their ecological, geomorphological and socioeconomic characteristics, which must be as comparable as possible to those of the MPA. In order to select the CA, three criteria have been identified. They include: distance from MPA; geomorphology of the site; similar economic and subsistence uses. The first criterion is essential: the MPA and the CA should be close enough so as to ensure homogeneity of major ecological features. Short distances also avoid the influence of weather (e.g. rainfalls, temperatures) and extreme events (e.g. floods, storms) on the MPA-CA comparison. However, MPA and CA should not be too close in order to avoid reciprocal influences (in this case, the benefits of MPAs may be difficult to estimate since the CA can also benefit from the redistributive effect of the MPA for fish species and tourism developments generated by the MPA). In consultation with national MPA experts in the various countries, the choice was made to consider 5 km as the minimum distance between an MPA and its CA.

The CA should also have similar geomorphological characteristics to those of the MPA. Thus, the CA must include a major delta if the MPA includes one, the CA should be an island if the MPA is an island, among others. Ensuring similar geomorphological features often leads to comparable biotopes and biocenoses. In cases where the geomorphological characteristics are not identical, one should ensure that ecosystems included in the MPA compare well with those of the MPA in terms of their ecological function (and the services that are provided based on these functions).

Lastly, the MPA and the CA must be comparable in terms of the potential economic and subsistence uses of their MCEs by the local populations. People in both MPA and CA should practice the same activities. This is a way to ensure that: i) ecosystems provide the same livelihood opportunities to populations of both the MPA and the CA; ii) there is no special activity in one site that can create extra value or, alternatively, generates an extra economic loss to one site; iii) ecosystems of the CA are ready to be exploited, while ecosystems within the MPA are protected for religious or cultural reasons (e.g. existence of sacred sites or cultural preference for farming rather than fishing).

Considering these key criteria for the choice of CA, the sites selected are as follows:
- For Langue de Barbarie MPA: the CA chosen includes a part of the langue extending from 5 km north of the MPA up to the southern limit of Saint Louis city. This site is the only one that bears comparison with the MPA because of the special geomorphological features of the MPA (being close to the Senegal River mouth).

- For Urok MPA: the chosen CA is the island of Galinas, which includes the same MCEs and has comparable uses. It is also part of the Bijagos archipelago Biosphere, but has no special management measures (unlike Urok);

- For Rio Cacheu MPA: the CA selected is located on another river estuary, Rio Cacine (south of Guinea-Bissau). Although quite distant from Cacheu, this site is part of the same ecoregion and includes the same ecosystems. It is a large estuary (like Rio Cacheu) and is located very close to the national border which has an influence on its economic activities. However, with regards to other uses, they are similar.

- For Tristao MPA: the selected CA is located around Kanfarandé on the other side of the estuary of the Rio Kogon, northwest of Kamsar. While this area is an archipelago, there are no such island groups near Tristao, and this estuary is the only one with such similar ecosystems and economic activities.

- For Santa Luzia MPA: the selected CA is located west of Sao Vicente Island. It is extremely difficult to select a CA which compares well to Santa Luzia because this MPA is uninhabited and is the only one with such characteristics in the northern part of Cape Verde archipelago. However, apart from this, the CA includes the same ecosystems and sees the same economic activities undertaken.

Appendix 1 provides maps of the various MPAs included in the study (red boxes) and their related CA (yellow boxes).

3.5.3 Geographical boundaries
For the economic valuation of the MCEs, one should consider only the marine and coastal ecosystems and the services they provide. The terrestrial boundary of such ecosystems is located at the level of the spring high tides, which therefore includes the intertidal area. The population considered here includes the villages within the sites, and possibly those at a very short distance from it (and under its direct influence for the provision of their livelihood). This also includes the allochton population (population originating from outside the area, such as migrant fishermen) that can settle in the site, even if only for a short period of time.
3.5.4 Surface areas of marine and coastal ecosystems

Having defined the MCEs that are present in the three main ecoregions of West Africa and chosen the MPAs and CAs that will be scrutinized during the study, it is then important to understand the surface areas of the ecosystems in the studied sites. In some cases, details about surface areas are available for MPAs. In my case, however, no such information exists. As it was impossible to carry out field missions in all MPAs in order to characterize all ecosystems in these sites, a remote sensing method was therefore applied to satellite pictures in order to recognize ecosystems and later estimate surface areas for each ecosystem.

Remote sensing and more specifically the processing of satellite images are an important tool for the mapping of land use and land cover. This processing method is also used for coastal ecosystem mapping. The optical specifications of satellite sensors distinguish short infra-red from medium infra-red wavelengths and between visible wavelengths. This differentiation between wavelengths allows the classification of surface areas in an automated manner and is made possible thanks to the physiological characteristics of vegetation or mineral objects. These have different radiometric responses that can be identified by the analysis of the visible and infrared spectrum. Thus, factors like the density of coastal vegetation and the presence of sandbanks in shallow waters result in different responses on satellite sensors. Satellite image processing was used in this case to both recognize the MCEs in the MPA and CA and also to evaluate the density of vegetation within ecosystems30.

Table 3-5 presents the surface areas of ecosystems as identified by satellite image modelling. The detailed geographical distribution of ecosystems in each studied site is provided in Appendix 2.

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This work was carried out by Vincent Turmine, GIS expert and research associate at CEMARE, who prepared the maps of ecosystems in the MPA and CA of the study which are presented below.
Table 3-5: Ecosystems surface for the MPA and CA (km²)

<table>
<thead>
<tr>
<th></th>
<th>Estuaries and channels</th>
<th>Seagrass meadows</th>
<th>Mangroves</th>
<th>Mudflats</th>
<th>Beaches</th>
<th>Rocky bottoms</th>
<th>Coral bottoms</th>
<th>Total</th>
</tr>
</thead>
<tbody>
<tr>
<td>Langue Barbarie MPA</td>
<td>7.0</td>
<td>0.5</td>
<td>0.8</td>
<td>2.8</td>
<td>5.6</td>
<td>0.0</td>
<td>0.0</td>
<td>16.7</td>
</tr>
<tr>
<td>CA Saint Louis</td>
<td>10.6</td>
<td>0.7</td>
<td>1.7</td>
<td>1.1</td>
<td>2.5</td>
<td>0.0</td>
<td>0.0</td>
<td>16.6</td>
</tr>
<tr>
<td>Rio Cacheu MPA</td>
<td>113.6</td>
<td>1.3</td>
<td>424.0</td>
<td>37.0</td>
<td>3.1</td>
<td>0.6</td>
<td>0.0</td>
<td>579.6</td>
</tr>
<tr>
<td>Rio Cacine CA</td>
<td>108.5</td>
<td>6.4</td>
<td>151.0</td>
<td>36.4</td>
<td>6.5</td>
<td>1.4</td>
<td>0.0</td>
<td>310.2</td>
</tr>
<tr>
<td>Urok MPA</td>
<td>48.5</td>
<td>2.0</td>
<td>77.7</td>
<td>72.0</td>
<td>0.8</td>
<td>0.4</td>
<td>0.0</td>
<td>201.4</td>
</tr>
<tr>
<td>Galinas island CA</td>
<td>25.5</td>
<td>1.4</td>
<td>43.8</td>
<td>43.5</td>
<td>0.7</td>
<td>0.5</td>
<td>0.0</td>
<td>115.4</td>
</tr>
<tr>
<td>Tristao MPA</td>
<td>232.8</td>
<td>1.6</td>
<td>134.5</td>
<td>151.0</td>
<td>5.9</td>
<td>1.0</td>
<td>0.0</td>
<td>526.9</td>
</tr>
<tr>
<td>Kanfarandé CA</td>
<td>76.0</td>
<td>1.4</td>
<td>69.6</td>
<td>96.0</td>
<td>1.2</td>
<td>0.7</td>
<td>0.0</td>
<td>244.9</td>
</tr>
<tr>
<td>Santa Luzia MPA</td>
<td>0.0</td>
<td>1.3</td>
<td>0.0</td>
<td>0.0</td>
<td>4.0</td>
<td>31.6</td>
<td>0.8</td>
<td>37.7</td>
</tr>
<tr>
<td>West Sao Vicente CA</td>
<td>0.0</td>
<td>1.5</td>
<td>0.0</td>
<td>0.0</td>
<td>5.4</td>
<td>30.4</td>
<td>0.6</td>
<td>37.9</td>
</tr>
</tbody>
</table>

3.6 The socioeconomic context of the case study

This section introduces the socioeconomic context that prevails along the coast of West Africa and within the MPAs and CAs in particular. This context is related to a few economic activities that depend on the MCEs and, as such, are described in the following subsection. These activities include: commercial and subsistence fishing, sport fishing, salt production, mangrove wood-cutting and coastal tourism.

3.6.1 Commercial fishing

The 1980s marked a turning point in the development of the fisheries sector. Historically, fishing had occurred upon a local seasonal cycle, punctuated by an annual return to the home village during the rice planting and harvest seasons. They subsequently adopted what Cormier-Salem (1995, 2000) calls ‘route’ fishing – the migration route being punctuated with stops in coastal cities where catches are landed. After the 1980s, while fishing is still practiced as a complementary livelihood activity in coastal communities, it has become increasingly important as a commercial activity based on migratory practices and targeting both domestic (in urban areas) and export markets (Chauveau, 1984; Pavé and Charles-Dominique, 1999). Fishermen now migrate within the country or between countries in the sub-region to either follow the seasonal movements of some small pelagic species or to visit more productive fishing grounds. In Guinea-Bissau and northern Guinea, Senegalese
migrant fishermen will go to sea for 10 to 15 days and fill the holds of their boat with several tonnes of demersal species such as noble grouper ("thiof" in Senegalese) and breams, which are of high commercial value for the European Union market before they go back to Senegal to land. An equally important fishery is a small pelagic fishery exploited by Guinean and Sierra Leonean fishermen. Installed in camps, they fish and process locally before sending smoked bags of small pelagics to Conakry and Freetown and further inland to domestic markets.

![Image](image_url)

**Figure 3-16: Senegalese fishers back from fishing in the Bijagos archipelago (Guinea-Bissau) and landing in Senegal (credit: T. Binet)**

The studied MPAs and their related CAs also experience a high presence of (mostly migrant) commercial fishermen. Settled in beach camps or living onboard their pirogues (small boats), they fish intensively before returning to their home settlements to land. Alternatively, some have settled in these MPAs permanently and may return to their area once a year. One striking example of these migrant fishing camps can be found in the Tristao MPA. It concentrates more than 3,000 settlers on the beach, who depend exclusively on the small pelagic fishery: most are fishermen, but some are in charge in charge of smoking ethmalosa, transporting them to the mainland, or running food and fishing equipment stores (some are even hairdressers!) (Figure 3-17).
Commercial fishing in the region is characterized by a great variety of targeted species. These can be classified into demersal species (that depend on the sea bottom for their survival) and pelagic species (that live in open water or close to the surface and do not depend on the sea bottom for their survival). Demersal species include those of the sciaenid family (*Galeoides decadactylus, Pseudotholitus elongatus brachygnatus, Pseudotholitus senegalensis, Pseudotholitus typus*), catfish (*Arius Sp*) and sea breams (*Dentex sp.*, *Pagellus bellotii, P. caeruleostictus*). Other species include sharks and rays. Demersal species are of higher value than pelagic species. For this reason, they are almost exclusively kept for export to the EU market. Some demersal species (such as catfish and Cassava croaker - *Pseudotolithus senegalensis*) reach the local market (Guinea-Bissau and Guinea in particular) where they are sold at a high price. The small pelagic species (for the local market and for export to the inland and landlocked West African countries) are round and flat sardinelles (*Sardinella aurita and S.maderensis*), and ethmalosa (called bonga in Guinea) (*Ethmalosa fimbriata*) (Figure 3-18). These small pelagics are almost exclusively processed (smoked or dried) before being transported to market.
This great variety of targeted species has led to the development of various fishing techniques and gears. The main fishing gears can be grouped into four classes: gill nets, seines, lines and traps, but it is worth noting that 95% of the total number of fishing gears encountered involve the use of monofilament nets (Ecoutin et al., 1999). Their use is considered inexpensive and very effective, despite its devastating environmental impact (known as "ghost fishing" 

3.6.2 Subsistence fishing

Subsistence fishing generally targets the same species as commercial fishing. The reason for this is that a part of the commercial catch is kept for family subsistence needs, while the rest of the catch is sold to wholesalers and processors, or directly to consumers on the local market. Though these catches represent a substantial part of the total catches of fishermen, they are not accounted for in the national statistics. As a consequence, it is difficult to estimate subsistence fishing levels in the studied sites without going into the field for an in-depth survey.

However, some communities are not engaged in commercial fishing and fish exclusively to feed their families. This is the case of some villages in the Tristao MPA, most villages in the Bijagos archipelago, and some villages of the Langue de Barbarie MPA (according to local

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31Nets lost or abandoned at sea are responsible for "ghost fishing": these nets continue to 'fish' even if they are lost at sea. This fishery is mainly caused by "monofilament" nylon nets whose low cost and difficulty to repair are contributing factors which promote their abandonment at sea. These nets have been banned for almost two decades in Senegal but this ban is not enforced at all, and the "monofilament" nets have never been as popular or commonly found in pirogues as today.
fisheries experts). In the case where fishing is only intended to feed the family, the gears used may differ from those of the commercial fishery. Subsistence fishers use more traditional devices such as traps, lines, and nets that are lifted, dragged, or pushed into position by fishermen. Fishermen in Guinea-Bissau use baskets to trap the fish, casting nets and harpoons (see Figure 3-19) (Charles-Dominique, 1994; Ecoutin et al, 1999). However, the species caught are similar to those of commercial fisheries: mostly small pelagics and mullets, but also catfish and other demersal fish (e.g. sea breams, groupers).

![Figure 3-19: The collection of arches; the use of baskets for fishing in the MPA Cacheu (Guinea-Bissau) (illustration: T. Caroff)](image)

### 3.6.3 Sport fishing

Recreational (or sport) fishing includes fishing activities carried out by tourists from the beach near their accommodation or through day tours onboard fishing boats. Sport fishing onboard fishing boats is widely practiced in the Bijagos archipelago in Guinea-Bissau and Cape Verde, where many tourists come (especially from Europe) for this activity.

In Guinea-Bissau, the main destination for sport fishing is the Bijagos Archipelago. Anglers travel to the Bijagos for a week or two, sometimes more. They go fishing every day and
mostly all day long. Sport fishing is their only reason for travelling to the islands. They stay on the main island of Bubaque (Casa Africana and Dauphins hotels), the main entrance to the archipelago is by sea or plane, on the neighbouring island of Rubane (Tubaron, Ponta Antcaka lodges) or in camps specialized in fishing on the west and south of the archipelago (in Orango – Parque Hotel Orango, Caravela – Camp Ker, Joao Vieira – Camp Bossard and the Carnage camp - islands).

Sport fishing in Cape Verde is performed aboard vessels specialized in trolling and (occasionally) with fixed or drifting lines. The main target species for trolling are swordfish and marlins (Figure 3-20), wahoos, yellowfin tunas, mahi-mahis and the various oceanic shark species present in the coastal waters. Line fishing targets large species such as groupers, kingfish, rays and various species of jacks.

![Figure 3-20: Sport fishing boat and blue marlin captured (credit: www.pechesportivecapvert.com)](image)

3.6.4 Salt production
Salt production is an important source of income for the coastal populations of West Africa. It is widely collected in the large mangrove and estuaries ecosystems, such as the Senegal River estuary and the Saloum Delta. Salt is produced on hyper-salinized bare soil ("tanne") where the mangrove habitat has been totally removed. Salt production sites are typically found in rice fields that have been abandoned because of their hyper-salinization. These salted mudflats are flooded during high tides and the soil therefore gets overloaded with salt during the dry season from January to June.

The most common salt production technique in Guinea consists of extracting the top layer of the dried soil and washing it with saltwater in order to get a highly concentrated brine. This brine is then boiled until all the water is evaporated and the salt can be collected. The
production yields are about 30 to 40 kg of salt for 100 to 150 litres of brine from 5-6 hours of boiling (Ecoutin et al., 1999. Soenen and Traineau, 1991).

The fact that salt production sites are located in mangrove ecosystems has another advantage. The remaining mangrove habitat close to the sites is also a source of fuel for the fire. This practice has therefore had a huge impact on wood cutting: about 3 kg of *Avicennia* wood is required to produce one kilogram of salt in continuous production conditions (Ecoutin et al., 1999).

**Figure 3-21: Salt production camp – Kanfarandé area (Guinea) (credit: A. Doumbouya)**

### 3.6.5 Mangrove wood-cutting

Mangrove wood can be used in four different ways:

- Household needs: cooking, heating, preparation of palm oil, and construction of houses;
- Smoking fish: surveys and literature references have established that more than 3 kg of wood is needed to produce one kg of smoked fish (Ecoutin et al., 1999.);
- Traditional salt production by boiling brine;
- Traded as firewood, charcoal and poles for construction to urban centres.

Estimates of timber contained in relatively intact mangrove areas are large, with those stocks close to urban areas being of high economic value. This exploitation is one of the main threats to the balance of the mangrove ecosystem, though it is also an important economic activity. In Guinea, it was established that 93,000 tonnes of *Avicennia* are cut annually for salt production, 58,000 tonnes of *Rhizophora* for smoking fish, and 206,000 tonnes are cleared for sale for domestic uses. This represented a minimum of 357,000...
tonnes of mangrove wood cut annually in the late 1990s (Ecoutin et al, 1999). It is very likely that the volumes cut have increased greatly in the years since due to coastal urbanization and the concentration of population in the coastal zones.

### 3.6.6 Medical exploitation

The medicinal uses are confined to the family or the village community. The local population regularly uses plants from marine ecosystems, mostly from mangroves. For example, the mangrove *Avicennia* is used to treat malaria (see Figure 3-22).

![Collection of *Avicennia* for treatment of malaria](credit: A. Borot)

The medicinal use of MCEs is mostly associated with religious practices by the local population. It was found that the boundary between religious and medicinal use of such products is tenuous. Thus, the medicinal use is, in most cases, both physical and spiritual. For example, mangroves are used during excision ceremonies (still a very common practice in the Tristao MPA). During the ceremony, bandages made with various plants from the mangrove ecosystem are applied to children to disinfect their wounds. These products are also believed to heal their souls (according to the local population interviewed on this issue).

### 3.6.7 Tourism

Seaside tourism is present in very specific sites along the coast of West Africa: Saint Louis, Casamance rivermouth and the Gambian coast. Other regions have few tourists throughout the year. Coastal tourism is only present in four of the studied sites: in the Langue de
Barbarie MPA and its CA in Senegal and in Santa Luzia MPA and its CA in Cape Verde. Tourism in other sites is absent or limited to a few individuals only per year.

The tourism infrastructure located in the Langue de Barbarie MPA is on the sandbar located between the ocean and the river Senegal. There are two camps installed on the sandbar that accept tourists for stays of several days (the average stay being two to three days), and offer the comfort of tourist resorts (individual bungalows with bathrooms, restaurants, bars, boat tours, and windsurfing equipment for rent).

![Tourist camp « El Faro » on the sand bar of Langue de Barbarie MPA (Senegal) (credits: T. Binet)](image)

The tourism infrastructure on South Saint Louis CA is sited on the sand bar north of the beach and was opened in 2003. There are nine hotels on the sandbars south of Saint Louis city. Many tourists who come to St Louis stay in one of those hotels which offer all of comforts typically associated with international resorts (e.g. bungalows, swimming pools, etc).

The Santa Luzia MPA has tourists (nationals and foreigners) who come to the island by boat on day trips. Their exact number is uncertain. However, it is possible to obtain leaflets from several different companies in Mindelo (Sao Vicente) advertising day trips to Santa Luzia, so this activity is not perhaps negligible. In addition, several diving centres offer diving opportunities in the MPA. Fishermen also regularly take tourists onboard their boats for a day trip and leave them on the island for a few hours.

Coastal tourism is very important in the CA of West Sao Vicente. The village of Sao Pedro has two resorts located on the waterfront, including the Hotel Foya Branca which is a large-
scale resort. The tourism activities in the area include bathing (though the strong waves and wind make it dangerous) and snorkelling which is practiced nearby in the CA. Walks along the coast and windsurfing are also important activities in the area.

3.7 Survey design

This section provides further details of the survey. An ethical review has been undertaken for this survey and ethical approval has been given by the Ethical Committee of the University (the Ethical Review is presented in Appendix 3).

The objective of the survey was threefold: i) to collect information on the use of various ecosystems and undertake a direct evaluation; ii) to collect socioeconomic data; iii) to apply choice experiments. The information obtained informed the valuation of the TEV of MCE services.

3.7.1 Populations: residential, allochton and visiting

For the survey, I needed to better define the different populations that are considered for use or non-use valuation of MCE services. Populations in the studied sites may be divided into: i) the resident population; ii) the population of non-native users or allochton population (temporary residents of the site, such as migrant fishers); and iii) tourists. People who have non-use value of MCEs (here defined as the population of non-users) are also found in each of these three categories of people.

3.7.1.1 Resident population

The resident population is the one that lives in the study site permanently. This population practice many activities in connection with MCEs such as fishing, shellfish collection, salt production and cutting of mangrove wood for smoking fish. Fishing communities can be classified into two categories: commercial and subsistence fishers.

3.7.1.2 Allochton population

The local population is subject to strong migratory movements in the region, especially as regards fishing activity. There is thus a non-native migrant population that has settled in the study areas for working purposes (fishing or post-harvesting activities). This population was considered in the survey because of its important impact on the uses of the study areas.
Users living outside of the study sites can also temporarily visit the MPA for specific activities. This can be the case for uninhabited areas such as Santa Luzia in Cape Verde where three communities who settled on a different island (Sao Vicente) traditionally exploit the fishery resources of Santa Luzia.

3.7.1.3 Tourist population

In general, the MCEs attract three types of tourists: sport fishers; tourists for seaside holidays; and eco-tourists. The sport fishers visit the MPA for a very specific purpose. These anglers are limited in number and are found only in the waters of the Bijagos archipelago (Urok MPA), Langue de Barbarie and Santa Luzia. For the others, they just enjoy the beauty of the MPA site and the climate that allows bathing activities, relaxation and observation of flora and fauna (e.g. the avifauna, as in the National Park of the Langue de Barbarie).

To date, little information exists on tourists visiting the MPA in West Africa. It was therefore necessary to approach the MPA managers and business hotels and hikers in the area to gather information on these tourists and evaluate their economic weight (e.g. total number, length of stay, activities performed, budget, etc).

3.7.1.4 Estimated population per site

The
Table 3-6 presents the details of data collected on population in study areas.
Table 3-6: Details of population in studied sites

<table>
<thead>
<tr>
<th>Study area</th>
<th>Village number</th>
<th>Total population</th>
<th>Resident population</th>
<th>Allochton population</th>
<th>Annual population of tourists</th>
</tr>
</thead>
<tbody>
<tr>
<td>National Park Langue de Barbarie (Senegal)</td>
<td>7 villages</td>
<td>4,470</td>
<td>3,670</td>
<td>0</td>
<td>800</td>
</tr>
<tr>
<td>Comparison site for National Park Langue de Barbarie (Senegal)</td>
<td>5 villages</td>
<td>3,202</td>
<td>3,102</td>
<td>0</td>
<td>100</td>
</tr>
<tr>
<td>Urok MPA (Guinea-Bissau)</td>
<td>22 villages</td>
<td>2,320</td>
<td>2,262</td>
<td>58</td>
<td>0</td>
</tr>
<tr>
<td>Comparison site Urok MPA (Guinea-Bissau) : Galinas Island</td>
<td>6 villages</td>
<td>1,189</td>
<td>1,157</td>
<td>32</td>
<td>0</td>
</tr>
<tr>
<td>Rio Cacheu National Park (Guinea-Bissau)</td>
<td>27 villages</td>
<td>16,622</td>
<td>16,058</td>
<td>564</td>
<td>0</td>
</tr>
<tr>
<td>Rio Cacheu National Park (Guinea-Bissau) : Comparison site: Rio Cacine</td>
<td>25 villages</td>
<td>13,191</td>
<td>12,590</td>
<td>601</td>
<td>0</td>
</tr>
<tr>
<td>Santa Luzia MPA (Cape Verde)</td>
<td>Uninhabited island</td>
<td>610 to 710</td>
<td>0</td>
<td>400 to 500</td>
<td>About 200</td>
</tr>
<tr>
<td>Santa Luzia MPA comparison site: western part of Sao Vicente island</td>
<td>1 village</td>
<td>1,550</td>
<td>1,400</td>
<td>100</td>
<td>50</td>
</tr>
<tr>
<td>Tristao MPA (Guinea)</td>
<td>34 villages</td>
<td>About 9,000</td>
<td>6,000</td>
<td>3,000</td>
<td>0</td>
</tr>
<tr>
<td>Tristao MPA comparison site: Kanfarandé (Guinea)</td>
<td>25 villages</td>
<td>About 1,5000</td>
<td>10,000</td>
<td>5,000</td>
<td>0</td>
</tr>
</tbody>
</table>

Source: extraction from national population statistical databases

### 3.7.2 Sampling plan

The selected criteria to check the representativeness of the sampling plan with regards to the parent population included population structure, sex ratio and socio-occupational classification. However, given the limited information available on the population of the areas studied (except for total population), it was impossible to produce a sampling plan for any given specific socio-economic area prior to the survey.

As a result, the sampling method adopted is to select all respondents above 15 years old in one of every three successive houses in selected streets of the various villages. This method
is recognized as creating the most representative samples of populations in data-poor situations (Wattage, pers. Comm.). The selection of respondents has to be surveyed at different hours during the day. Hence, during the day the men tend to go out to work and are not in the village when investigations could be conducted, so it is the women who respond. Early morning or evening surveys will therefore target men to ensure a consistent sex ratio with the parent population. The surveyor also needs to consider seasonal migration of population in and out the site in order to be representative (e.g. seasonal migrant fishermen).

The survey was carried out in all streets of the village in all villages of the selected site. If the population was above one third of the sampling size, then the survey was carried out once in every three streets, but still in as many villages as possible (avoiding the most remote small villages that are very difficult to reach).

The number of surveys per site of 200 ensured a margin of error ranging from 2.5% to 10%. This is suitable for a statistical treatment of a total population of between 1500 and 8000 people (Glenn and Wattage, n.d.). However, to reduce the margin of error in the case of the most populated areas, the choice was made to increase the number of surveys to 250 questionnaires per site.

3.7.3 Questionnaire and survey material

The questionnaire used in the survey attempted to identify the perception of MCEs by the resident population, allochton and tourists. In order to simplify the survey, one survey was used for all three populations identified. The statistical treatment then differentiates between the three populations identified.

The survey material was composed of the questionnaire (for use and non-use valuation) and a portfolio of text and pictures that introduces content and the objective of the survey as well as the various scenarios proposed to the respondent (for non-use valuation). The questionnaire and the presentation portfolio are presented in Appendix 4 and 5. These two documents were translated into Portuguese for investigations in Cape Verde and Guinea-Bissau. The material was in French for the other sites.

After a brief introduction (section 1) to the objectives and the context of the study, a general information section (Section 2) asked the person about his relationship with the area (e.g. resident or visitor, origin, or length of stay in the area). Section 3 of the questionnaire aimed
to gather an idea of the level of general knowledge of West African MCEs. In particular, the respondent was asked to recognize pictures of specific ecosystems. Section 4 went deeper into the knowledge of MCEs while asking about the pressures of the MCEs. Section 5 dealt with the MCEs protection and the level of awareness of the respondent about the restoration of degraded ecosystems. Section 6 was dedicated to non-use valuation. Section 7 revolves around the direct uses and Section 8 provided additional socio-economic information on the person questioned.

The presentation of scenarios for choice experiments was on the same page in order not to create bias in choice by individuals. The presentation was done through the use of pictures or pictograms. Using pictograms enabled me to adopt a guided vision for the attribute level. It prevents the bias that can be created by various picture qualities, and ensures the homogeneity of perceptions by surveyed individuals (Earnhart, 2001).

The questionnaire and presentation portfolio were tested with 10 to 15 random individuals selected in the first site prior to the survey, in order to ensure full understanding of the survey and to correct for any potential unclear content. All individuals demonstrated a very good understanding of the questions asked and provided sound answers to the questions, which confirmed that the questionnaire and surveyors were ready for the survey.

3.7.4 Survey method
The survey was carried out with the help of students from national universities. For each of the countries visited, between three and five students were recruited to administer the questionnaires.

All students were trained before going in the studied sites. It was very important that they understood the concepts used in the questionnaire (such as biodiversity and ecosystems) in order to be able to explain it when in the field. As a consequence, half a day was dedicated prior to the survey in order to read the questionnaire with all surveyors, explain the concepts and questions and clarify any misunderstandings. Also, details about the sampling method were given so that the survey may be representative of the parent population.

3.8 Profiles of respondents
Given the lack of socioeconomic data in the studied sites within national statistical databases, it was not possible to establish any sampling plan prior to the field survey so as
to document the socioeconomic features of the studied sites. For this reason I carried out one post-survey assessment of socioeconomic data to check about comparable data within the MPAs and their related CA. The socioeconomic profiles of survey respondents are presented in the figures and tables below. The pairs of sites are presented side by side in order to enable better visual comparison for a number of socioeconomic characteristics: age, average size of households, level of education, main household activity, average income, and origin (autochthon population or visitor). Figure 3-24 reflects the age structure of the sampled populations. It does not show any visible difference in age groups obtained between the MPA and the CA: each age group is present in comparable proportions for each MPA and its comparison area.

![Figure 3-24: Age structure of respondents by site](image)

The main activities of the population of MPAs and their CAs are illustrated in Figure 3-25. Fishers and farmers account for the largest percentage of jobs in the sites, ranging from 40% to 70% of the total jobs. It appears that the occupations are distributed almost similarly between the MPA and CA in Senegal, Guinea-Bissau and Cape Verde. In Guinea, the main difference is that fishing activities account for more than 50% of economic activity for Tristao MPA and less than 25% for the comparison site in Kanfarandé. This difference can
be explained by the presence of the Katchek fishing camp which represents almost one third of the MPA total population. In addition, most fishermen in Kanfarandé are subsistence ones and practice many other activities depending on the season. Hence they rarely define themselves as fishermen.

Figure 3-25: Main activities of respondents by site

N.B.: the main activity is the activity that is most practiced (in terms of number of days over one year).

Figure 3-26 shows the average income per household. The proportions for each income range again show significant similarities between MPAs and their CAs.
N.b.: for sites in Guinea, income ranges were converted into CFA Guinean Francs at a rate of $1 = 10$ Guinean Francs CFA; for Cape Verde; a conversion rate of $1.00$ CFA = $160$ Cape Verdean Escudos was used.

The level of education of the local population surveyed is shown in Figure 3-27, and again the data show that the MPA and CAs are comparable. This is especially true for the percentage of people who have never been to school. This level is very high for sites in Senegal, where the population is also older than in the other West African sites (around $75\%$ of the surveyed population), which may explain this high proportion.
Overall, the socioeconomic data collected on the sampled population suggest strong similarities between MPAs and comparison areas in terms of their socioeconomic characteristics: age, average income, and education level. Importantly, there is not any strong difference in average household revenues, which could have caused major differences in the willingness to pay evaluation. The differences observed for the main activities might be caused by the fact that the populations have multiple activities and found it difficult to consider one as a main activity\textsuperscript{32}. However, these differences are not likely to significantly affect the results of the evaluation. It is thus acceptable, in my view, to consider the comparison areas as sufficiently similar areas for the purpose of comparing the economic values of their ecosystems (at least from a socioeconomic perspective).

\textsuperscript{32}Another explanation would be that the activities classification was not appropriate to the respondents’ profiles, and this caused misunderstanding among respondents and biases in their answers.
The survey has also enabled me to collect population information to be used for economic valuation: household sizes, proportion of autochton/allochton populations, and origins of tourists. The average household size in sites is shown in Table 3-7. Some differences in size of households have been observed, especially for Langue de Barbarie MPA (8.9 person per household on average) and its comparison area in Saint Louis (11.1 people per household on average). Also, the differences are significant in Guinea-Bissau, where the differences between the MPA and the comparison site range from 20 to 25%. For other sites (Guinea and Cape Verde), the differences are negligible and households have equivalent sizes.

Table 3-7: Average size of households by studied site

<table>
<thead>
<tr>
<th>Site (MPA or comparison area)</th>
<th>Average size of household (individuals)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Langue Barbarie MPA</td>
<td>8.9</td>
</tr>
<tr>
<td>Saint Louis CA</td>
<td>11.1</td>
</tr>
<tr>
<td>Cacheu MPA</td>
<td>10.2</td>
</tr>
<tr>
<td>Cacine CA</td>
<td>12.5</td>
</tr>
<tr>
<td>Urok MPA</td>
<td>7.5</td>
</tr>
<tr>
<td>Galinas CA</td>
<td>9.2</td>
</tr>
<tr>
<td>Tristao MPA</td>
<td>10.1</td>
</tr>
<tr>
<td>Kanfarandé CA</td>
<td>10.9</td>
</tr>
<tr>
<td>Santa Luzia MPA</td>
<td>5.8</td>
</tr>
<tr>
<td>Sao Vicente CA</td>
<td>5.9</td>
</tr>
</tbody>
</table>

The proportion of indigenous population and visitor population (divided into the immigrant population resident and the tourist population) are shown in Figure 3-28. This proportion varies considerably from one area to another in Cape Verde and, to a lesser extent, in Guinea. The population in Santa Luzia MPA is a visitor population only because the islands of Santa Luzia are uninhabited. In the Tristao MPA, the important commercial fishing camp in Katchek consists of about 3,000 people, including 2,000 fishermen, most of whom are allochton. They are migrating from other parts of Guinea (e.g. Conakry, Boffa), Sierra Leone and Senegal. In the case of the Langue de Barbarie MPA in Senegal, the visiting population is almost exclusively made up of tourists, and is very different when compared to the CA.
The surveyed population of tourists showed an exclusivity of European tourists: France (56%), Belgium (24%), Italy (9%), and Greece (11%). The differences of origins observed between MPAs and their related CAs are insignificant, except in the case of Santa Luzia and Tristao. However, these differences do not invalidate the MPA-CA comparison, which is only based on socioeconomic features. In the case of Santa Luzia however, data handling and treatment was adapted in order to reflect the fact that the MPA is uninhabited.

![Graph showing proportions of autochton population and visitors](image)

**Figure 3-28: Proportions of autochton population and visitors**

### 3.9 Conclusion of the chapter

This chapter has served as an introduction to my study. First, it has provided the ecological background for this study. The three ecoregions of West Africa have been described, as well as the ecosystems they comprise: estuaries and channels, mangroves, seagrass meadows, mudflats, beaches, rocky bottoms, coral bottoms. Each ecosystem has been described in detail and an overview of its health status in the region provided. This ecological
information is useful for the characterization and calculation of use values, both direct and indirect (in the two following chapters).

Second, the calculation methods applied specifically to the West African MPAs for the three components of the TEV, as well as the MPA conservation benefits, have been described. The proposed methods take into account the main features for research carried out in this area, namely poor data availability and practical difficulties related to field data collection.

Third, the MPAs selected for this study have been described. My sample includes five MPAs (one in an upwelling ecoregion, three in estuarine and mangrove ecoregions, and one in a volcanic archipelago of the Cape Verde ecoregion). This sample is thus a representative sample of the three ecoregions. For each of the MPAs, an associated comparison area (CA) has been identified. The CAs for each MPA are located close to them (except for Rio Cacine), and include the same geomorphological and socioeconomic characteristics, so as to provide close comparisons for this study.

Fourth, a geographical and ecological characterization of the MPAs and CAs of the study has been undertaken, enabling us to delimit these areas and estimate the surface areas for each ecosystem. The total surface area of the studied sites represents more than 2,000 km$^2$, of which more than 900 km$^2$ is mangroves, 620 km$^2$ are estuaries and mangroves, and mudflats extend to 440 km$^2$. The largest MPA is Rio Cacheu (Guinea-Bissau) with 580 km$^2$.

Fifth, the chapter describes the socioeconomic context of the West African coastline and of the MPAs of the sample in particular. The main economic activities of the region have been characterized: commercial, subsistence and sport fishing, salt production, mangrove wood-cutting, medicinal uses and tourism.

Sixth, the chapter has presented the approach for the implementation of the survey. It has presented the various populations considered, the questionnaire and sampling method for the survey. It has also presented the profiles of respondents, as measured throughout the survey.

Thus, this chapter provides an overview of the case study of the thesis. It sets the context and the method for economic valuation adapted to the characteristics of the West African MPAs. The four next chapters present the results of the valuation undertaken.
4 Chapter 4: direct use values

4.1 Introduction
The characterization of the case study within chapter 3 has set the scene for the valuation exercise. This chapter and the three that follows detail the results and analysis for the three components of the TEV and the subsequent MPA conservation benefits. In this first chapter, I present the results for the first component of the TEV, the direct use values. The presentation of results in a second subsection is accompanied by some background in order to expedite the interpretation of these results.

4.2 Types of direct uses
The types of direct uses studied in this chapter draws on all uses in relation to the MCE of the MPAs, as described in the socioeconomic activities. They consist of uses related to fishing activities, be they for commercial, subsistence or recreational purposes. Other extractive uses include salt production and mangrove wood cutting for sale or household consumption (cooking, house building). The direct use values associated with non-extractive activities include those related to tourist boat tours, and also to other tourist activities such as swimming, bathing, and sailing. Medicinal uses are not considered here since there is no economic activity attached to it.

Also, the induced uses (which are most often associated to direct use values) in West Africa revolve around aquaculture and the use of mangrove wood. Aquaculture is present in two forms in the West African region: oyster and shrimp farming. However, these activities are not present in the studied sites and, for this reason, are not considered here. In a few years though, shrimp farming will be occurring in Cape Verde as shrimp farms are being established at a fast pace (see Figure below).
The induced use of wood cutting for other activities has already been accounted through mangrove wood-cutting direct use and is therefore not considered here as induced use.

### 4.3 Specific method applied

When market data is not available, it is not possible to estimate the value of provisioning services through direct market valuation, using consumer and producer surpluses. Rather, the most common method to estimate direct use value is to estimate the added value of one production type, such as fisheries. For situations in which few economic data is available on the market, the gross value-added (GVA) provides a good estimate of the ecosystem service provided. The GVA of one good $i$ is equal to the difference between the values of product $Pi$ minus the sum of $j$ intermediate consumption costs (IC) associated with the production of $i$:

$$
VAi = Pi - \sum_{n=0}^{j} ICn
$$

**Eqn 4-1: Value added of one good $i$**

The GVA can be used for valuing direct use values such as fisheries, wood cutting, salt production, and nautical activities in West African MCEs. Data on the average price of marketed commodities available on the local market can be collected over a year for different seasons. Prices for the past five years are then also estimated in order to obtain an average price. Intermediary costs are estimated through interviews with producers (e.g.
fishers, wood cutters, salt producers, owner of hotels and campsites). Based on these, direct use values can be calculated.

Table 4-1 details the data collection methods used for each of the direct use values.

**Table 4-1: Details of calculation methods for direct and induced use values**

<table>
<thead>
<tr>
<th>Use</th>
<th>Values</th>
<th>Data collection and method for calculation</th>
</tr>
</thead>
<tbody>
<tr>
<td>Direct use</td>
<td>Commercial fisheries</td>
<td>Estimation of gross added value based on annual catch volumes available at national department of fisheries and based on estimates from national fisheries experts (GVA= turnover – intermediary costs)</td>
</tr>
<tr>
<td></td>
<td>Subsistence fisheries</td>
<td>Estimation of GVA based on the commercial price of the considered species. Catches to be estimated from field surveys, interviews with fishermen to be validated by national experts</td>
</tr>
<tr>
<td></td>
<td>Recreational and sport fishing</td>
<td>Estimation of GVA of sport fishing operators based on interviews with professionals</td>
</tr>
<tr>
<td></td>
<td>Salt production</td>
<td>Estimation of production volumes and average price of salt from surveys carried out in salt production sites in mangroves</td>
</tr>
<tr>
<td></td>
<td>Commercial mangrove wood cutting</td>
<td>Estimation of production volumes and average price of mangroves from surveys carried out with professional wood cutters</td>
</tr>
<tr>
<td></td>
<td>Mangrove wood cutting for domestic use (e.g. cooking, house building)</td>
<td>Estimation of production volume per household per year and average price of mangrove wood sold for domestic use on the local market</td>
</tr>
<tr>
<td></td>
<td>Other extracting uses: exploitation of natural resources for medicinal purposes</td>
<td>Estimation of volume and average price paid to local collectors</td>
</tr>
<tr>
<td></td>
<td>Tourist activities associated with MCEs (e.g. nautical excursions, boat tours, bird watching)</td>
<td>Information collected through interviews with tourism operators; estimation of the participation of MCEs in tourist activities</td>
</tr>
<tr>
<td>Induced use</td>
<td>Situations in which MCEs provide production factors for one direct use (fish farming for instance)</td>
<td>Estimation of annual turnover and intermediary costs based on interviews</td>
</tr>
</tbody>
</table>
Having valued the total direct use values for each site, I then had to identify the extent to which each ecosystem is used for the various activities considered. The survey carried out in studied sites has enabled me to quantify this, asking each respondent to describe their activities within each ecosystem and the frequency of this activity (e.g. daily, once a week) (See questions 26 to 33 in Appendix 4). At the end, I had identified the number of times the activity was practiced in each ecosystem per year. Based on this information and the total number of days of the activity undertaken per year, it was then possible to deduce the percentage of use for each ecosystem[^33]. These percentages are presented in the Table 4-2.

[^33]: For instance, if fishing was practiced by an individual once a month in rocky bottoms and once a week in mangroves and every day in estuaries: the percentage of use will be 365/(365+12+52)=85% in estuaries (3% on rocky bottoms and 12% in mangroves).
<table>
<thead>
<tr>
<th>Countries</th>
<th>Site</th>
<th>Activity</th>
<th>Estuaries and channels</th>
<th>Seagrass meadows</th>
<th>Mangrove</th>
<th>Mudflats</th>
<th>Beaches</th>
<th>Rocky bottoms</th>
<th>Coral bottoms</th>
</tr>
</thead>
<tbody>
<tr>
<td>Senegal</td>
<td></td>
<td>Subsistence fishing</td>
<td>29%</td>
<td>9%</td>
<td>18%</td>
<td>7%</td>
<td>26%</td>
<td>11%</td>
<td>0%</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Commercial fishing</td>
<td>29%</td>
<td>7%</td>
<td>16%</td>
<td>15%</td>
<td>22%</td>
<td>11%</td>
<td>0%</td>
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<tr>
<td></td>
<td></td>
<td>Sport fishing</td>
<td>18%</td>
<td>16%</td>
<td>16%</td>
<td>16%</td>
<td>16%</td>
<td>18%</td>
<td>0%</td>
</tr>
<tr>
<td></td>
<td>Langue de Barbarie MPA</td>
<td>Mangrove wood cutting</td>
<td>0%</td>
<td>0%</td>
<td>100%</td>
<td>0%</td>
<td>0%</td>
<td>0%</td>
<td>0%</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Pharmaceutical uses</td>
<td>18%</td>
<td>22%</td>
<td>22%</td>
<td>16%</td>
<td>19%</td>
<td>3%</td>
<td>0%</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Salt production</td>
<td>0%</td>
<td>0%</td>
<td>100%</td>
<td>0%</td>
<td>0%</td>
<td>0%</td>
<td>0%</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Coastal leisure (e.g. bathing, sailing)</td>
<td>30%</td>
<td>0%</td>
<td>0%</td>
<td>0%</td>
<td>70%</td>
<td>0%</td>
<td>0%</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Tourist boat tours (fauna and flora observation)</td>
<td>22%</td>
<td>40%</td>
<td>13%</td>
<td>1%</td>
<td>23%</td>
<td>0%</td>
<td>0%</td>
</tr>
<tr>
<td>Saint Louis CA</td>
<td></td>
<td>Subsistence fishing</td>
<td>39%</td>
<td>10%</td>
<td>10%</td>
<td>9%</td>
<td>24%</td>
<td>8%</td>
<td>0%</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Commercial fishing</td>
<td>39%</td>
<td>10%</td>
<td>10%</td>
<td>10%</td>
<td>23%</td>
<td>9%</td>
<td>0%</td>
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<tr>
<td></td>
<td></td>
<td>Sport fishing</td>
<td>30%</td>
<td>60%</td>
<td>0%</td>
<td>0%</td>
<td>10%</td>
<td>0%</td>
<td>0%</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Mangrove wood cutting</td>
<td>0%</td>
<td>0%</td>
<td>100%</td>
<td>0%</td>
<td>0%</td>
<td>0%</td>
<td>0%</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Salt production</td>
<td>0%</td>
<td>0%</td>
<td>100%</td>
<td>0%</td>
<td>0%</td>
<td>0%</td>
<td>0%</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Coastal leisure activities (e.g. bathing, sailing)</td>
<td>42%</td>
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<td>0%</td>
<td>0%</td>
<td>68%</td>
<td>0%</td>
<td>0%</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Tourist boat tours (fauna and flora observation)</td>
<td>28%</td>
<td>11%</td>
<td>8%</td>
<td>21%</td>
<td>32%</td>
<td>0%</td>
<td>0%</td>
</tr>
</tbody>
</table>

Table 4-2: Breakdown of uses by ecosystem
<table>
<thead>
<tr>
<th>Countries</th>
<th>Site</th>
<th>Activity</th>
<th>Estuaries and channels</th>
<th>Seagrass meadows</th>
<th>Mangrove</th>
<th>Mudflats</th>
<th>Beaches</th>
<th>Rocky bottoms</th>
<th>Coral bottoms</th>
</tr>
</thead>
<tbody>
<tr>
<td>Guinea-Bissau</td>
<td>Cacheu MPA</td>
<td>Subsistence fishing</td>
<td>35%</td>
<td>3%</td>
<td>32%</td>
<td>11%</td>
<td>17%</td>
<td>3%</td>
<td>0%</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Commercial fishing</td>
<td>32%</td>
<td>2%</td>
<td>24%</td>
<td>26%</td>
<td>14%</td>
<td>2%</td>
<td>0%</td>
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<tr>
<td></td>
<td></td>
<td>Sport fishing</td>
<td>47%</td>
<td>6%</td>
<td>26%</td>
<td>0%</td>
<td>19%</td>
<td>0%</td>
<td>0%</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Mangrove wood cutting</td>
<td>0%</td>
<td>0%</td>
<td>100%</td>
<td>0%</td>
<td>0%</td>
<td>0%</td>
<td>0%</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Pharmaceutical uses</td>
<td>26%</td>
<td>4%</td>
<td>27%</td>
<td>26%</td>
<td>15%</td>
<td>3%</td>
<td>0%</td>
</tr>
<tr>
<td>Guinea-Bissau</td>
<td>Cacine CA</td>
<td>Salt production</td>
<td>0%</td>
<td>0%</td>
<td>100%</td>
<td>0%</td>
<td>0%</td>
<td>0%</td>
<td>0%</td>
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<tr>
<td></td>
<td></td>
<td>Subsistence fishing</td>
<td>32%</td>
<td>4%</td>
<td>27%</td>
<td>22%</td>
<td>14%</td>
<td>1%</td>
<td>0%</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Commercial fishing</td>
<td>34%</td>
<td>2%</td>
<td>27%</td>
<td>23%</td>
<td>13%</td>
<td>1%</td>
<td>0%</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Sport fishing</td>
<td>34%</td>
<td>3%</td>
<td>31%</td>
<td>21%</td>
<td>10%</td>
<td>1%</td>
<td>0%</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Mangrove wood cutting</td>
<td>0%</td>
<td>0%</td>
<td>100%</td>
<td>0%</td>
<td>0%</td>
<td>0%</td>
<td>0%</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Salt production</td>
<td>0%</td>
<td>0%</td>
<td>100%</td>
<td>0%</td>
<td>0%</td>
<td>0%</td>
<td>0%</td>
</tr>
<tr>
<td>Guinea-Bissau</td>
<td>Urok MPA</td>
<td>Subsistence fishing</td>
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4.4 Results

4.4.1 Commercial fishing

4.4.1.1 Calculation method

MPAs and CAs offer thorough illustrations of regional fisheries. They include all the fisheries described here and their related fishing gears. Pelagic species are not usually captured in the MCEs considered here because they are fished away from the shore, close to the surface. However, these species greatly depend on the MCEs for their reproduction and juvenile stages. In addition, they benefit from nutrient inputs from the MCEs for their feeding. As a consequence it is fair to consider them as part of the fisheries in the MCEs.

In contrast, tunas caught in the region (as well as some highly migratory species) are not included in computing Gross Added Value (GVA) because they are more indirectly dependent than small pelagics on the MCEs. Thus, they are related through the trophic chain to MCEs but do not have one part of their life cycle actually within one of these. This trophic linkage and whether it is appropriate to include offshore migratory species such as tropical tunas (Yellowfin, Skipjack and Bigeye tunas) in my study could be further discussed. However, the fisheries considered here only include those that are directly related to MCEs (and exclude those are indirectly linked to MCEs), namely the commercial demersal fisheries, the small pelagics fisheries and also the commercial collection of shellfish on the intertidal shore.

The calculation of GVA requires knowledge of the catches, the average price per kilogram and the operational costs associated with fishing. National statistics on capture fisheries were used to provide a first approximation of quantities caught in the studied areas. However, these data were most often incomplete, outdated, and collected across different and much larger areas (district areas). It was therefore very difficult to exploit these data. However, it is still possible to use such data for cross-checking with the studied sites estimates compiled with national experts on the field. The annual average prices of recent years and the average operational costs for each fishery were obtained from interviews with fishermen in the main landing sites associated with the studied sites. Interestingly, these were relatively homogeneous across countries. Operational costs averaged 10% of the selling price of the fish, irrespective of whether pelagic or demersal fisheries were being considered in all the continental sites. These were different in the case of Cape Verde
though, where operational costs were higher because of the fishing practices that consisted of daily trips to the islands and subsequent higher costs in fuel.

### 4.4.1.2 Senegal

In the Langue de Barbarie MPA and its CA, the opening of the breach within the sandbar between the Senegal River mouth and the sea in 2003 has had a major effect on the fisheries. The entrance of saltwater into the brackish waters of the estuaries has increased the presence of some ocean species, while traditional fisheries, that targeted species such as threadfins (*Galeoides decadactylus*), mullets and sardinellas, experienced a sharp decrease in their catches.

According to local fisheries experts and fishermen themselves, catches have remained at a relatively low level since 2003. Fishing in the estuary is not a major economic activity. Catches are mainly composed of pelagic species (mullet, sardine, ethmalosa, or threadfins). Fishing is carried out with surface driftnets or casting nets. Average annual catches over the past five years were estimated at 7.5 tonnes for the Langue de Barbarie MPA and 3.75 tonnes for the comparison area of south Saint Louis. The average first-sale price has been estimated at 250 CFA francs per kilogram (equivalent to 0.38 euros per kg).

### 4.4.1.3 Guinea-Bissau

Commercial fishing is practiced all over the continental shelf that extends from the continental coast to about 160 km offshore beyond the Bijagos Archipelago, encompassing a surface area of more than 53,000 km$^2$. The main fishing gears used in commercial fishing in Guinea-Bissau include:

- surface driftnets to catch mullets and ethmalosas;
- gillnets for fishing catfish, sharks and rays, and soles;
- open water driftnets for big fish (barracudas, threadfins, sharks);

Surface driftnets are used primarily by migrant fishermen from Sierra Leone and Guinea, which mainly target ethmalosa. Gillnets were introduced by Senegalese migrant fishermen targeting demersal species (e.g. sea bream, sole), with an emphasis on sharks for their fins to be sold on the Asian markets.

A recent study (Gonzalez, 2010) was used to validate the estimates made during field surveys on the current total production by study site and the average price. This study
reported large catches in 2009 in the coastal areas of Cacheu and Cacine of 1,271 tonnes (902 of demersal species and 369 of small pelagics) and 5,589 tonnes respectively (2,348 tonnes of demersal and 2,967 tonnes of small pelagics). However, these data do not seem representative of the 2010 and 2011 situation. During field visits, it appeared that almost all non-native fishermen, after spending several years in the area, had left the Rio Cacine for northern Guinea (and the Tristao MPA), thereby drastically decreasing the total annual catches reported. For the Cacheu MPA, recent estimates showed a lesser decrease in total catches due to on-going fishing activity in the area by some migrant groups who had stayed in the estuary to continue fishing. Finally, the average catches over the past five years were estimated at 750 tonnes of demersal and 600 tonnes of small pelagics in the Cacheu MPA and 700 tonnes and 500 tonnes of demersal and small pelagics in the Cacine CA. The average price of small pelagic fish was estimated at 250 CFA francs per kg and that of demersal at 600 CFA francs per kg (equivalent to 0.38 euros/kg and 0.92 euros/kg respectively)

The literature on the Urok islands and Galinas comparison area mainly considers the whole Bijagos archipelago and the islands where migrant fishers are found (Uracane, Uno, and Caravela). A recent study, however, enabled us to obtain an update of the census of fishermen in the Urok MPA (Savary-Bellon, 2009). For Galinas, no specific reference could be identified. Commercial fishing in the waters of the Bijagos archipelago is homogeneous, and is undertaken by:

- indigenous commercial fishermen that use longlines, bottom driftnets for demersal fish and surface driftnets for small pelagics; they fish seasonally for commercial purposes, sell their catch to migrant fishers, and bring them to the local market in the most populated Bijagos island of Bubaque or the capital Bissau; and
- migrant professional fishermen settled in the archipelago (islands of Bolama, Bubaque or Uracane) that practice fishing all year long; they use purse seines for capturing small pelagics, bottom longlines and driftnets for demersal species.

Catch data from the commercial fishery in the coastal waters of Urok MPA and Galinas CA in the past five years is very difficult to estimate, because of the extreme mobility of commercial fishers that make them hard to spot. However, by estimating the capture at the scale of the archipelago and through interviews with fisheries experts in the country as well as commercial fishermen, it was possible to define the importance of these waters for
fishing (in contrast to other areas within the archipelago) and therefore estimate the proportion of catches that could be attributed to fishing in these areas. Overall, these waters are not key fishing grounds for commercial fishing. Estimates suggest catches are about 10 tonnes of demersal and 38 tonnes of small pelagics for Urok, and eight tonnes of demersals and 30 tonnes of pelagics for Galinas. The average prices are based on the average prices in the capital Bissau, which are 500 CFA francs per kg for demersal (0.77 euros/kg) and 250 CFA francs per kg for small pelagics (0.38 euros/kg).

### 4.4.1.4 Guinea

In northern Guinea, according to local fisheries experts, commercial fishing has become one of the most important economic activities over the years. A large migrant fishing camp has developed on the beach in Katchek on the south-western coast of the Tristao main island. This camp specialized in fishing ethmalosa, a few miles off the Tristao MPA. The estimated commercial fishery in the MPA (which includes the catch of the other villages of the MPA but excludes migrant fishing carried out outside the MPA boundaries) was estimated at about 8,500 tonnes per year over the past five years. Catches of demersal are much lower at 130 tonnes a year. The comparison area of Kanfarandé has no such migrant fishing camp, but catches are important in the estuary of the Rio Nunez where the main cities of Kamsar and Kanfarandé are located, and where local commercial fishermen are based. There, the main fishery is targeting demersal, catfish being a priority as it is highly appreciated on the domestic market as a smoked product. Total catches were estimated at 1,500 tonnes of demersal and 1,500 tonnes of pelagic.

### 4.4.1.5 Cape Verde

Commercial fishing in Cape Verde is generally undertaken onboard small boats (Figure 4-2). Fishing activities target mainly demersal species (grouper, moray, toothed, porgy, red snapper, parrot fish, ling) and large pelagic species such as tunas (skipjack, yellowfin, bluefin tuna), barracudas, swordfishes, mahi-mahis, and jacks. The most intense period for commercial fishing runs from September to November (INDP, 2007).
Fishing in the MPA of Santa Luzia and its islets is carried out by fishermen from villages on the neighbouring islands: Sinagoga (Santo Antao), Calhau and Praia Grande, Salamansa and São Pedro (on the island of São Vicente), and Tarrafal (São Nicolau). In these villages, the majority of people depend on fishing for their livelihood (between 70% and 90% of the income of the village is provided by commercial fishing). Fishermen from Sinagoga remain in the MPA for an average 8 to 10 days during the trade winds season. Those of São Vicente go to the MPA throughout the year with a more pronounced presence during the summer, going to Santa Luzia for shorter periods of 2 to 3 days. Fishermen from Salamansa (which is the closest village to Santa Luzia) may go fishing in the MPA on a daily basis, 6 days a week.

The estimated catches per year in Santa Luzia MPA are nine tonnes: 5.3 tonnes of demersal fish and 3.7 tonnes of tuna. The average price of fish is 200 CVE per kg of tuna (equivalent to 1.80 euros) and 220 Cape Verdean escudos (CVE) per kg of demersal fish (2 euros). Thus, the value of commercial fishing catches in the MPA is 1,166,000 CVE for demersal and 740,000 CVE for tuna, a total of 1,906,000 CVE (17,300 euros).

In addition, operational costs were estimated at about 30% of the revenue generated. This high proportion of fishing costs (when compared to those of the other regions of West Africa) can be explained by the high fuel cost required to reach the MPA, and the relatively low catches per trip at sea as compared to fishermen in the other MPAs of the region (most harvest less than 100 kg per boat per trip at sea).

Commercial fishing in the comparison area to the west of Sao Vicente (Sao Pedro village) is practiced by Sao Pedro fishermen and some fishermen from Mindelo or the neighbouring island of Sao Nicolau. The national statistics for 2003 reported 53 fishing boats, of which 27 had an engine. However, my field survey has shown a greater proportion of motorized boats.
(about 90%). The estimated catch per year for this area is 120 tonnes of demersal fish and 180 tonnes of tuna. Operational costs and average selling prices remain similar to those of the MPA.

Figure 4-3: Sao Pedro beach, island of Sao Vicente (Cape Verde) (credit: Nato)

4.4.1.6 Shellfish collection

The collection of arches (*Anadara senilis*) and oysters (mainly *Crassostrea gasar*) is an activity that goes far back into the history of the development of coastal societies (Ecoutin et al., 1999). Today, harvesting shellfish on the mudflats and beaches is a very minor activity in northern Senegal and Cape Verde (and for this reason will not be considered here). It is however a very intense activity in Guinea and Guinea-Bissau. Here, arches form the basis of animal protein in the diet (Tiniguena, 2003). In the Bijagos archipelago, arches and oysters are banned from sale for religious reasons outside the islands. As a result, they are reserved for home consumption. All arches harvested in the Bijagos should thus be recorded as subsistence fishing.

Unlike the arches, most of the oysters are sold on the local market and are therefore considered as commercial fishing here. They are harvested from the roots of the mangrove or, in the case of rock oysters, collected on the rocks. They are mostly sold as sundried products. It takes about 15 kg of fresh oysters for one kilogram of dried oysters. In Guinea-Bissau, other shellfish are harvested in the mudflats and beaches (gastropods such as “gandim” - *Pugilina morio* - and “contchubedja” - *Cymbium*). The Table 4-3 shows the shellfish species encountered in this large ecosystem. These species have been heavily exploited by the families of migrant fishermen (see below picture and comment). As a consequence, the resource has been significantly reduced in both number and average size. The *Cymbium* has become rare (average harvest of less than 5 kg per day for a collector in
Urok), while the “lingrons” (*Tagellus adamsoni*) have been overexploited and stocks collapsed in Urok in 2004, and have still not recovered.

On an islet in the Urok MPA, the massive piles of shells instantly indicate intense shellfish collection took place in the past. This was carried out by the migrant population (Senegalese women) who travelled with their husband fishermen. Such activity is now forbidden in Urok, although it is likely that this massive exploitation has happened on other unprotected islands.

Figure 4-4: Shellfish piles on an islet located in the vicinity of Urok MPA (credit: T. Binet)
Table 4-3: Species of molluscs found in Guinea and Guinea-Bissau (Regalla et Baldé, 2008)

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<tr>
<td>9</td>
<td><em>Senilia senilis</em></td>
<td>Combé</td>
<td>21</td>
<td><em>Natica marochiensis</em></td>
</tr>
<tr>
<td>10</td>
<td><em>Ensis goreensis</em></td>
<td>Canivete de Goré</td>
<td>22</td>
<td><em>Bullia miran</em></td>
</tr>
<tr>
<td>11</td>
<td><em>Gari bomii</em></td>
<td>Boné</td>
<td>23</td>
<td><em>Cymbium pepo</em></td>
</tr>
<tr>
<td>12</td>
<td><em>Crassostrea tulipa</em></td>
<td>Ostra</td>
<td>24</td>
<td><em>Cymbium glans</em></td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>25</td>
<td><em>Natica tigrina</em></td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>26</td>
<td><em>Natica turtoni</em></td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>27</td>
<td><em>Nerita senegalensis</em></td>
</tr>
</tbody>
</table>

Annual harvests were estimated during field surveys because no past data was found on such practices, with the exception of one report that suggested a volume of 2,000 tonnes of oysters harvested in Tristao, although this seemed largely overestimated (Doumbouya, 2008). Estimates of the daily harvest of oysters based on field surveys ranges from 3 to 6 baskets per harvester (each basket is from 4 to 15kg). Given 200 women work 80 days per year on this activity, the annual production of Tristao is therefore about 640 tonnes of fresh oysters. These values were confirmed by the observation of shell piles around the villages as
all oyster shells are removed before being dried by harvesters in the village, so all shells remain in the village. In this way, it is therefore possible to estimate the total production of the village when the age of the pile is known.

Volumes were estimated at about 160 tonnes for Cacheu, 100 tonnes for Cacine, 30 tonnes for Urok, 10 tonnes for Galinas, 400 tonnes for Tristao and 300 tonnes for Kanfarandé. These volumes and unit prices are reported in the Table 4-4. The considered volume of dried oyster is calculated based on a ratio of 1 kg per 15 kg of fresh oyster.

Table 4-4: Gross value-added of commercial shellfish harvesting

<table>
<thead>
<tr>
<th>Countries</th>
<th>Site</th>
<th>Main species</th>
<th>Collection (kg)</th>
<th>Average price (CFA/kg)</th>
<th>Operational costs (CFA/kg)</th>
<th>GVA (CFA)</th>
<th>GVA (€)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Guinea-Bissau</td>
<td>Cacheu MPA</td>
<td>Oysters</td>
<td>160,000</td>
<td>2,000 (dried)</td>
<td>200</td>
<td>19,200,000</td>
<td>29,538</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Other shellfish</td>
<td>40,000</td>
<td>500</td>
<td>50</td>
<td>1,800,000</td>
<td>27,692</td>
</tr>
<tr>
<td></td>
<td>Cacine CA</td>
<td>Oysters</td>
<td>90,000</td>
<td>2,000 (dried)</td>
<td>200</td>
<td>10,800,000</td>
<td>16,615</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Other shellfish</td>
<td>10,000</td>
<td>500</td>
<td>50</td>
<td>4,500,000</td>
<td>6,923</td>
</tr>
<tr>
<td></td>
<td>Urok MPA</td>
<td>Oysters</td>
<td>30,000</td>
<td>2,000 (dried)</td>
<td>200</td>
<td>3,600,000</td>
<td>5,538</td>
</tr>
<tr>
<td></td>
<td>Galinas CA</td>
<td>Oysters</td>
<td>10,000</td>
<td>2,000 (dried)</td>
<td>200</td>
<td>1,200,000</td>
<td>1,846</td>
</tr>
<tr>
<td>Guinea</td>
<td>Tristao MPA</td>
<td>Oysters</td>
<td>640,000</td>
<td>900 (dried)</td>
<td>90</td>
<td>34,560,000</td>
<td>53,169</td>
</tr>
<tr>
<td></td>
<td>Kanfarandé CA</td>
<td>Oysters</td>
<td>300,000</td>
<td>900 (dried)</td>
<td>90</td>
<td>16,200,000</td>
<td>24,923</td>
</tr>
</tbody>
</table>

4.4.1.7 Synthesis

The Table 4-5 summarizes the data on the gross value-added of the commercial fisheries in each of the study sites.
<table>
<thead>
<tr>
<th>Countries</th>
<th>Site</th>
<th>Targeted species</th>
<th>Catches (kg)</th>
<th>Average price (CFA/kg if not detailed)</th>
<th>Operational costs (CFA/kg if not detailed)</th>
<th>GVA (CFA if not detailed)</th>
<th>GVA (€)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Senegal</td>
<td>Langue Barbarie MPA</td>
<td>Pelagic fish</td>
<td>7,500</td>
<td>250</td>
<td>25</td>
<td>1,687,500</td>
<td>2,600</td>
</tr>
<tr>
<td></td>
<td>Saint Louis CA</td>
<td>Pelagic fish</td>
<td>3,750</td>
<td>250</td>
<td>25</td>
<td>843,750</td>
<td>1,300</td>
</tr>
<tr>
<td>Guinea-Bissau</td>
<td>Cacheu MPA</td>
<td>Demersal fish</td>
<td>750,000</td>
<td>600</td>
<td>60</td>
<td>405,000,000</td>
<td>623,000</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Pelagic fish</td>
<td>600,000</td>
<td>250</td>
<td>25</td>
<td>135,000,000</td>
<td>207,700</td>
</tr>
<tr>
<td></td>
<td>Cacine CA</td>
<td>Demersal fish</td>
<td>700,000</td>
<td>600</td>
<td>60</td>
<td>378,000,000</td>
<td>581,540</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Pelagic fish</td>
<td>500,000</td>
<td>250</td>
<td>25</td>
<td>112,500,000</td>
<td>173,080</td>
</tr>
<tr>
<td></td>
<td>Urok MPA</td>
<td>Demersal fish (threadfins, catfish)</td>
<td>10,000</td>
<td>500</td>
<td>50</td>
<td>4,500,000</td>
<td>7,000</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Pelagic fish (mullet, ethmalosa)</td>
<td>38,000</td>
<td>250</td>
<td>25</td>
<td>8,550,000</td>
<td>13,100</td>
</tr>
<tr>
<td></td>
<td>Galinas CA</td>
<td>Demersal fish (threadfins, catfish)</td>
<td>8,000</td>
<td>500</td>
<td>50</td>
<td>3,600,000</td>
<td>5,500</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Pelagic fish (mullet, ethmalosa)</td>
<td>30,000</td>
<td>250</td>
<td>25</td>
<td>6,750,000</td>
<td>10,400</td>
</tr>
<tr>
<td>Countries</td>
<td>Site</td>
<td>Targeted species</td>
<td>Catches (kg)</td>
<td>Average price (CFA/kg if not detailed)</td>
<td>Operational costs (CFA/kg if not detailed)</td>
<td>GVA (CFA if not detailed)</td>
<td>GVA (€)</td>
</tr>
<tr>
<td>-----------</td>
<td>-----------------------</td>
<td>-----------------------------------</td>
<td>--------------</td>
<td>----------------------------------------</td>
<td>---------------------------------------------</td>
<td>----------------------------</td>
<td>-----------</td>
</tr>
<tr>
<td>Guinea</td>
<td>Tristao MPA</td>
<td>Demersal fish (catfish, sea breams)</td>
<td>130,000</td>
<td>500</td>
<td>50</td>
<td>58,500,000</td>
<td>90,000</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Pelagic fish (sardinellas, ethmalosa)</td>
<td>8,500,000</td>
<td>250</td>
<td>25</td>
<td>1,912,500,000</td>
<td>2,942,300</td>
</tr>
<tr>
<td></td>
<td>Kanfarandé CA</td>
<td>Demersal fish (catfish, sea breams)</td>
<td>1,500,000</td>
<td>500</td>
<td>50</td>
<td>675,000,000</td>
<td>1,038,500</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Pelagic fish (sardinellas, ethmalosa)</td>
<td>1,500,000</td>
<td>250</td>
<td>25</td>
<td>337,500,000</td>
<td>519,200</td>
</tr>
<tr>
<td>Cape Verde</td>
<td>Santa Luzia MPA</td>
<td>Demersal fish</td>
<td>252,000</td>
<td>220 CVE/kg</td>
<td>65 CVE</td>
<td>39,060,000 CVE</td>
<td>355,000</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Tunas</td>
<td>318,800</td>
<td>200 CVE/kg</td>
<td>60 CVE</td>
<td>44,632,000 CVE</td>
<td>405,700</td>
</tr>
<tr>
<td></td>
<td>Sao Vicente CA</td>
<td>Demersal fish</td>
<td>180,000</td>
<td>220 CVE/kg</td>
<td>65 CVE</td>
<td>27,900,000 CVE</td>
<td>253,640</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Tunas</td>
<td>120,000</td>
<td>200 CVE/kg</td>
<td>60 CVE</td>
<td>16,800,000 CVE</td>
<td>152,700</td>
</tr>
</tbody>
</table>
Using the table above and the values obtained for commercial fishing and shellfish harvesting at the different sites, it is then possible to calculate the total contribution of each ecosystem to the commercial extraction of fish and shells.
Table 4-6: Gross value-added of commercial fishing and harvesting

<table>
<thead>
<tr>
<th>Countries</th>
<th>Site</th>
<th>Total (€)</th>
<th>Estuaries and channels (€/km²/yr)</th>
<th>Seagrass (€/km²/yr)</th>
<th>Mangrove (€/km²/yr)</th>
<th>Mudflats (€/km²/yr)</th>
<th>Beach (€/km²/yr)</th>
<th>Rocky bottoms (€/km²/yr)</th>
<th>Coral bottoms (€/km²/yr)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Senegal</td>
<td>Langue Barbarie MPA</td>
<td>2,600</td>
<td>107</td>
<td>364</td>
<td>501</td>
<td>141</td>
<td>103</td>
<td>N/A</td>
<td>N/A</td>
</tr>
<tr>
<td></td>
<td>Saint Louis CA</td>
<td>1,300</td>
<td>48</td>
<td>186</td>
<td>76</td>
<td>114</td>
<td>120</td>
<td>N/A</td>
<td>N/A</td>
</tr>
<tr>
<td>Guinea-Bissau</td>
<td>Cacheu MPA</td>
<td>887,996</td>
<td>2,502</td>
<td>13,661</td>
<td>503</td>
<td>6,240</td>
<td>40,103</td>
<td>29,600</td>
<td>N/A</td>
</tr>
<tr>
<td></td>
<td>Cacine CA</td>
<td>778,158</td>
<td>2,438</td>
<td>2,432</td>
<td>1,391</td>
<td>4,917</td>
<td>15,563</td>
<td>5,558</td>
<td>0</td>
</tr>
<tr>
<td></td>
<td>Urok MPA</td>
<td>25,608</td>
<td>211</td>
<td>647</td>
<td>43</td>
<td>68</td>
<td>2,241</td>
<td>10,243</td>
<td>N/A</td>
</tr>
<tr>
<td></td>
<td>Galinas CA</td>
<td>17,766</td>
<td>279</td>
<td>0</td>
<td>49</td>
<td>143</td>
<td>2,030</td>
<td>1,777</td>
<td>N/A</td>
</tr>
<tr>
<td>Guinea</td>
<td>Tristao MPA</td>
<td>3,085,479</td>
<td>2,651</td>
<td>19,284</td>
<td>3,440</td>
<td>11,852</td>
<td>26,054</td>
<td>30,855</td>
<td>N/A</td>
</tr>
<tr>
<td></td>
<td>Kanfarandé CA</td>
<td>1,582,613</td>
<td>7,077</td>
<td>11,304</td>
<td>6,599</td>
<td>4,451</td>
<td>92,319</td>
<td>45,218</td>
<td>N/A</td>
</tr>
<tr>
<td>Cape Verde</td>
<td>Santa Luzia MPA</td>
<td>760,840</td>
<td>N/A</td>
<td>36,173</td>
<td>N/A</td>
<td>N/A</td>
<td>79,888</td>
<td>5,292</td>
<td>285315</td>
</tr>
<tr>
<td></td>
<td>Sao Vicente CA</td>
<td>406,370</td>
<td>N/A</td>
<td>13,136</td>
<td>N/A</td>
<td>N/A</td>
<td>31,607</td>
<td>6,814</td>
<td>14,058</td>
</tr>
</tbody>
</table>
The table shows considerable variations in value from one ecosystem to another across the sites: very low for mangrove ecosystems and estuaries in Bijagos or Senegal, while values per surface area unit can be extremely high for the beaches in Kanfarandé and Santa Luzia. These differences partially reflect the differences in values between ecosystems for the commercial fishing sites. Only partially because another factor may also be influential: the relatively high percentage of fishing activity calculated from the survey results (and presented in table above) in some ecosystems of very limited coverage in the studied sites (such as seagrass and rocky bottoms) lead to very high unitary values. This relatively high percentage of fishing activity (compared to very small surface areas of ecosystems) is caused by an overstatement by the respondents about the use of these ecosystems. For instance a percentage of only 2% of the total fishing practices in rocky bottoms in Kanfarandé represents a high unitary value of 45,000 euros/km²/yr because the surface of rocky bottoms in Kanfarandé is limited to 0.7 km².

4.4.2 Subsistence fishing

4.4.2.1 Calculation method

To estimate the gross value-added of subsistence fishing, it is necessary to evaluate the kept for subsistence separate from total commercial catches. It is conventional to use market prices to estimate the value of subsistence fishing, since most subsistence fishing species are similar to commercial fishing ones. For example, in the case of a professional fisherman who keeps 5% of his total catches for home consumption and whose catches are 10 tonnes per year, the volume consumed on capture reach 500 kg - 125,000 CFA for fish at 250 CFA per kg (eq. 0.38 euros/kg).

The estimated GVA for subsistence fisheries varies according to several criteria:

- The target species: people set aside demersal fish with high commercial value for sale (catfish, threadfins, sea breams, and soles are rarely kept for self-consumption.). Subsistence species mostly consist of small pelagics: sardinelles, ethmalosa, mullets.

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34 This overstatement is possibly due to the question asked about frequency of use (daily, once a week). This question should be completed with a question on the length of use (for the full period of fishing). Rare ecosystems can be visited once a day or once a week but for a short period of time only.
- **The proportion of non-commercial fishers**: if fishermen are exclusively subsistence fishermen this may increase subsistence fishing estimates. This can be gauged by surveying the professionalization evident within the fisheries sector, subsistence fishing estimates being adjusted to reflect the level of professionalization.

- **The use of non-commercial species**: this issue is of particular importance in the Bijagos archipelago where arches cannot be marketed, they representing the main source of protein for the local population. As a consequence the average price of arches has been estimated based on the average price in the local market in the Saloum Delta in Senegal (where the sale of arches is allowed). The average price there is 100 CFA francs per kg (eq. 0.15 euros per kg).

### 4.4.2.2 Senegal

In Senegal, the proportion of self-consumption of fisheries production is relatively low. In interviews, this was estimated to represent about 8% of total catches based on interviews with commercial fishermen (who keep about 5% of their catches for the consumption of their household) and non-professional fishermen (who have higher rates of self-consumption but are less numerous in the areas sampled). Subsistence fishing only consists of pelagic fish (sardinelles and mullets).

### 4.4.2.3 Guinea-Bissau

Subsistence fishing is an important part of fishing in the Urok MPA, with most of the rest of the catch dedicated to local markets on the islands. Subsistence fishing is estimated at 25% of all fishing activity for small pelagics and 10% for demersal fish. In Galinas however, allochton commercial fishermen are present and this tends to lower the part of the catches kept for subsistence, while fishing effort and catch per unit of effort increase. As a result, subsistence fishing is estimated at about 20% for small pelagics and 8% for demersal fish in this area. Similar rates of subsistence fishing (of 20% of small pelagics and 5% for demersal fish) apply to Cacheu MPA and Cacine CA due to the significant presence of commercial fishing in the area.

With regards to shellfish harvested for subsistence in Bijagos, arches are collected throughout the year on the mudflats. They are also used for religious ceremonies like funerals ("tocachurro") and the ceremony of transition to manhood ("canhoca"). Oysters are present on the rocky bottoms and mangroves and are collected mainly during the dry
season (household consumption and ceremonies) with little collection during the rainy season (only for ceremonies). The species was overexploited in the past in Urok causing serious damage to the local mangroves, since oysters are collected by cutting the roots where they grow. Thanks to the creation of the MPA and subsequent promotion of sustainable practices, the mangrove is now better preserved while rotation of collection sites has reduced the pressure on oyster stocks. Total harvest volumes were estimated at 350 tonnes of shellfish in Urok and 180 tonnes in Galinas. The collection of arches and oysters is also important in Cacheu MPA and Cacine CA where they were estimated at 320 tonnes and 168 tonnes, respectively.

![Figure 4-5: Casting net fishing and arches collection in the Urok MPA – Guinea Bissau) (illustration: T. Caroff)](image)

### 4.4.2.4 Guinea

In the Tristao MPA, the share of self-consumption for small pelagic is estimated at 1%, as fishing capacity per unit of capture is very high (each boat can land several tonnes of ethmalosa each day). This very low proportion is caused by the fact that fishermen live alone in the fishing camp, far from their homes, and without children and wives. In contrast,
while the autochthon population within the MPA almost exclusively fish to supply their households, these catches are marginal given the weight of commercial fishing. In the Kanfarandé CA, there are no migrant fishing camps, despite a high proportion of commercial fishers. The percentage of subsistence fishing for small pelagics here is higher (10%). For demersal species, the share of consumption is very low (3%) in Tristao and Kanfarandé because these catches are almost exclusively reserved for sale. Oysters are the main shellfish harvested for consumption in Guinea, and harvest estimates are based on this species only as the collection of arches is low. Oyster harvesting for household consumption varies from one village to another, but is estimated to average 35% and 25% for Tristao and Kanfarandé, respectively. Oysters for consumption are also dried or smoked.

4.4.2.5 Cape Verde
In Cape Verde, fishing is almost exclusively undertaken for commercial purposes. Consequently, the share allocated to household consumption is very low, given the volumes of catch per unit (more than 10 tonnes per month per boat from February to July). Thus, the share of subsistence fishing in Santa Luzia and the CA of Sao Vicente was estimated at 5% of the total catch for demersal and tuna alike.

4.4.2.6 Synthesis
Table 4-7 presents the annual aggregated gross values of subsistence fishing for fish capture and shellfish harvesting by site for each ecosystem.
Table 4-7: Annual gross value-added of subsistence fish capture

<table>
<thead>
<tr>
<th>Countries</th>
<th>Site</th>
<th>Species (main targeted species)</th>
<th>Annual commercial fishing value (€)</th>
<th>Part kept aside for subsistence</th>
<th>GVA/yr (€)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Senegal</td>
<td>Langue Barbarie MPA</td>
<td>Pelagic fish</td>
<td>2,600</td>
<td>8%</td>
<td>200</td>
</tr>
<tr>
<td></td>
<td>Saint Louis CA</td>
<td>Pelagic fish</td>
<td>1,300</td>
<td>8%</td>
<td>100</td>
</tr>
<tr>
<td>Guinea-Bissau</td>
<td>Cacheu MPA</td>
<td>Demersal fish</td>
<td>623,000</td>
<td>5%</td>
<td>31,200</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Pelagic fish</td>
<td>207,700</td>
<td>20%</td>
<td>41,500</td>
</tr>
<tr>
<td></td>
<td>Cacine CA</td>
<td>Demersal fish</td>
<td>581,540</td>
<td>5%</td>
<td>29,100</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Pelagic fish</td>
<td>173,080</td>
<td>20%</td>
<td>34,600</td>
</tr>
<tr>
<td></td>
<td>Urok MPA</td>
<td>Demersal fish</td>
<td>7,000</td>
<td>10%</td>
<td>700</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Pelagic fish</td>
<td>13,100</td>
<td>25%</td>
<td>3,300</td>
</tr>
<tr>
<td></td>
<td>Galinas CA</td>
<td>Demersal fish</td>
<td>5,500</td>
<td>8%</td>
<td>400</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Pelagic fish</td>
<td>10,400</td>
<td>20%</td>
<td>2,100</td>
</tr>
<tr>
<td>Guinea</td>
<td>Tristao MPA</td>
<td>Demersal fish</td>
<td>90,000</td>
<td>3%</td>
<td>2,700</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Pelagic fish</td>
<td>2,942,300</td>
<td>1%</td>
<td>29,400</td>
</tr>
<tr>
<td></td>
<td>Kanfarandé CA</td>
<td>Demersal fish</td>
<td>1,038,500</td>
<td>2%</td>
<td>20,800</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Pelagic fish</td>
<td>519,200</td>
<td>10%</td>
<td>51,900</td>
</tr>
<tr>
<td>Cape Verde</td>
<td>Santa Luzia MPA</td>
<td>Demersal fish</td>
<td>355,000</td>
<td>5%</td>
<td>17,800</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Tunas</td>
<td>405,700</td>
<td>5%</td>
<td>20,300</td>
</tr>
<tr>
<td></td>
<td>Sao Vicente CA</td>
<td>Demersal fish</td>
<td>253,640</td>
<td>5%</td>
<td>12,700</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Tunas</td>
<td>152,700</td>
<td>5%</td>
<td>7,600</td>
</tr>
</tbody>
</table>
Table 4-8: Annual gross added value of shellfish harvesting for subsistence purpose

<table>
<thead>
<tr>
<th>Countries</th>
<th>Site</th>
<th>Species (main targeted species)</th>
<th>Annual catches (kg)</th>
<th>Average price (CFA/kg)</th>
<th>Operational costs (CFA/kg)</th>
<th>GVA/yr (CFA)</th>
<th>GVA/yr (€)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Guinea-Bissau</td>
<td>Cacheu MPA</td>
<td>Arches and other shellfish</td>
<td>320,000</td>
<td>100</td>
<td>10</td>
<td>28,800,000</td>
<td>44,300</td>
</tr>
<tr>
<td></td>
<td>Cacine CA</td>
<td>Arches and other shellfish</td>
<td>168,000</td>
<td>100</td>
<td>10</td>
<td>15,120,000</td>
<td>23,300</td>
</tr>
<tr>
<td></td>
<td>Urok MPA</td>
<td>Arches, oysters and other shellfish</td>
<td>500,000</td>
<td>100 (2,000 for dried oysters)</td>
<td>100 (2,000 for dried oysters)</td>
<td>301,500,000</td>
<td>463,800</td>
</tr>
<tr>
<td></td>
<td>Galinas CA</td>
<td>Arches, oysters and other shellfish</td>
<td>260,000</td>
<td>100 (2,000 for dried oysters)</td>
<td>100 (2,000 for dried oysters)</td>
<td>160,200,000</td>
<td>246,500</td>
</tr>
<tr>
<td>Guinea</td>
<td>Tristao MPA</td>
<td>Oysters</td>
<td>250,000</td>
<td>900 (dried)</td>
<td>90</td>
<td>202,500,000</td>
<td>311,500</td>
</tr>
<tr>
<td></td>
<td>Kanfarandé CA</td>
<td>Oysters</td>
<td>150,000</td>
<td>900 (dried)</td>
<td>90</td>
<td>121,500,000</td>
<td>186,900</td>
</tr>
</tbody>
</table>

Based on these tables and on the ecosystem surface areas, it was then possible to calculate the unitary values of subsistence fishing by ecosystem. The results are presented in Table 4-9.
Table 4-9: Aggregated gross added values for subsistence fishing by ecosystem

<table>
<thead>
<tr>
<th>Guinea</th>
<th>Site</th>
<th>Total per site (€)</th>
<th>Estuaries and channels (€/km²/yr)</th>
<th>Seagrass meadows (€/km²/yr)</th>
<th>Mangrove (€/km²/yr)</th>
<th>Mudflats (€/km²/yr)</th>
<th>Beaches (€/km²/yr)</th>
<th>Rocky bottoms (€/km²/yr)</th>
<th>Coral bottoms (€/km²/yr)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Senegal</td>
<td>Langue de Barbarie MPA</td>
<td>200</td>
<td>10</td>
<td>40</td>
<td>40</td>
<td>10</td>
<td>10</td>
<td>N/A</td>
<td>N/A</td>
</tr>
<tr>
<td></td>
<td>Saint Louis CA</td>
<td>100</td>
<td>0</td>
<td>30</td>
<td>10</td>
<td>10</td>
<td>10</td>
<td>N/A</td>
<td>N/A</td>
</tr>
<tr>
<td>Guinea-Bissau</td>
<td>Cacheu MPA</td>
<td>117,000</td>
<td>361</td>
<td>2,700</td>
<td>88</td>
<td>348</td>
<td>6,416</td>
<td>5,850</td>
<td>N/A</td>
</tr>
<tr>
<td></td>
<td>Cacine CA</td>
<td>63,700</td>
<td>188</td>
<td>398</td>
<td>101</td>
<td>385</td>
<td>1,372</td>
<td>455</td>
<td>N/A</td>
</tr>
<tr>
<td></td>
<td>Urok MPA</td>
<td>467,800</td>
<td>2,411</td>
<td>9,454</td>
<td>1,144</td>
<td>2,209</td>
<td>70,170</td>
<td>70,170</td>
<td>N/A</td>
</tr>
<tr>
<td></td>
<td>Galinas CA</td>
<td>249,000</td>
<td>2,441</td>
<td>3,557</td>
<td>1,080</td>
<td>2,175</td>
<td>49,800</td>
<td>9,960</td>
<td>N/A</td>
</tr>
<tr>
<td>Guinea</td>
<td>Tristao MPA</td>
<td>343,600</td>
<td>443</td>
<td>2,148</td>
<td>843</td>
<td>364</td>
<td>9,284</td>
<td>13,744</td>
<td>N/A</td>
</tr>
<tr>
<td></td>
<td>Kanfarandé CA</td>
<td>259,600</td>
<td>1,161</td>
<td>1,854</td>
<td>1,008</td>
<td>541</td>
<td>36,777</td>
<td>7,417</td>
<td>N/A</td>
</tr>
<tr>
<td>Cape Verde</td>
<td>Santa Luzia MPA</td>
<td>38,100</td>
<td>N/A</td>
<td>1,810</td>
<td>N/A</td>
<td>N/A</td>
<td>4,000</td>
<td>260</td>
<td>14,288</td>
</tr>
<tr>
<td></td>
<td>Sao Vicente CA</td>
<td>20,300</td>
<td>N/A</td>
<td>660</td>
<td>0</td>
<td>0</td>
<td>1,580</td>
<td>340</td>
<td>700</td>
</tr>
</tbody>
</table>

*N/A is used for non-applicable in sites where there is no such ecosystem*
For subsistence activities, the differences between sites are important: representing tens of euros per square kilometre per year in Senegal to tens of thousands of euros per square kilometre per year for the beaches in Bijagos or Kanfarandé. These differences are caused by differences in subsistence fishing practices between countries (see the differences between Bijagos and Cape Verde). Also, as subsistence fisheries values are estimated as percentages of commercial fisheries (for fish captures), this leads to very high unitary values for some ecosystems (i.e. the high estimates of subsistence practices in beaches and rocky bottoms and the small surfaces present in these sites creates a very high unitary value for fishing in these ecosystems).

4.4.3 Sport fishing

4.4.3.1 Context and calculation method

In the Langue de Barbarie MPA, tourism operators offer to rent fishing rods for surfcasting (fishing from the shore). However, this value is not considered here since rod renting is included in the added value generated by tourist accommodation (as calculated in the section below). Cacheu, Cacine, Tristao and Kanfarandé have no sport fishing. For this use, the evaluation method consists of determining the gross value added to the activity for each of the operators in the area (see method developed in chapter 2).

4.4.3.2 Guinea-Bissau

Interviews with sport fishing operators in Bubaque and Rubane were carried out in order to estimate annual visitors and details about the main fishing sites in the archipelago. It appeared that sport fishermen never fish in the waters around Urok islands because the channels within the MPA are prohibited for sport fishing (as part of the MPA management measures). For the Galinas CA, operators from Rubane and Bubaque occasionally exploit the channels close the island. The other operators to the west and south of the archipelago never go fishing there. According to the interviews, sport fishing in Galinas represents about 5% of total output. The total gross value of recreational fishing based on the declarations of the operators was estimated to be around 630,000 euros among the four sport fishing operators. As a result, sport fishing in the Galinas CA represents a GVA of 31,500 euros.
4.4.3.3 Cape Verde

The average stay for anglers on the island of Sao Vicente is six days, and there are six sport fishing operators on the island, including the Mindelo Centre for Sport Fishing (the main operator). They have a total of nine fishing boats, making an average of 100 trips per year per boat, with boats taking between one and four fishermen onboard. On average, it is estimated that two to three fishermen are present on the boat. Gross value added of sport fishing operators is therefore estimated to be 1,800 euros per stay per person if fishing is the sole purpose of the stay (six days fishing trip), or 400 euros per person if the fishing trip is undertaken for a day only. The gross value added of sport fishing operators is therefore estimated to be 787,500 euros per year.

Based on the total GVA of sport fishing in the islands, it is possible to estimate the part that can be allocated to the MPA and the CA ecosystems. For Santa Luzia, sport fishing is not allowed in the waters of the MPA. However, since access is not controlled by authorities, most operators go fishing there from time to time. Based on declarations from fishing operators, it is estimated that 20% of the fishing trips include the MPA, which represents a total of 180 daily trips per year for the nine boats. The gross value of sport fishing in the Santa Luzia MPA is therefore estimated at 157,500 euros.

The comparison area of West Sao Vicente is much more frequented by anglers as interviews demonstrated. According to the Centre for Sport Fishing website, this spot "is sheltered from the prevailing winds and where the catches of blue marlin average between 200 and 300 pounds while some captured specimens exceed 500 pounds" (http://www.pechesportivecapvert.com). This marlin spot located in the CA represents one major area for sport fishing. Based on interviews, it was estimated that the CA attracts 50% of trips by the Fishing Centre, 30 to 40% for other sport fishing companies. The GVA of West Sao Vicente is therefore estimated at 315,000 euros per year.
Surfcasting is also frequently practiced in Cape Verde. The targeted species are both pelagic and demersal species such as rays, groupers, and amberjack. However, this practice was considered negligible based on observations in the field and interviews with national fisheries experts.

4.4.3.4 Synthesis

Table 4-10 shows the gross value added for sport fishing broken down for each ecosystem. These figures are calculated by dividing the GVA of sport fishing for each ecosystem by the surface area of each ecosystem. This provides values for sport fishing by ecosystem by unit of area.

Looking at the table, I first note that the practices of sport fishing are concentrated on highly productive ecosystems such as seagrass beds or coral reef ecosystems. On the other hand, the values show an important use of beaches, which are not necessarily the main fishing grounds according to fishing operators. This may be explained by the fact that recreational fishing also considered free fishing, such as surfcasting, and not only onboard sport fishing. Accordingly this may have created one bias in the valuation exercise and attributed some value for sport fishing to beaches\textsuperscript{35}.

\textsuperscript{35} This hypothesis is one possible explanation of the results, but it was not possible to back this up with wider references.
Table 4-10: Gross value added of sport fishing

<table>
<thead>
<tr>
<th>Countries</th>
<th>Site</th>
<th>Total per site (€)</th>
<th>Estuaries and channels (€/km²/yr)</th>
<th>Seagrass meadows (€/km²/yr)</th>
<th>Mangrove (€/km²/yr)</th>
<th>Mudflats (€/km²/yr)</th>
<th>Beaches (€/km²/yr)</th>
<th>Rocky bottoms (€/km²/yr)</th>
<th>Coral bottoms (€/km²/yr)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Guinea-Bissau</td>
<td>Galinas CA</td>
<td>31,500</td>
<td>741</td>
<td>2,250</td>
<td>0</td>
<td>145</td>
<td>0</td>
<td>6,300</td>
<td>N/A</td>
</tr>
<tr>
<td>Cape Verde</td>
<td>Santa Luzia MPA</td>
<td>157,500</td>
<td>N/A</td>
<td>12,500</td>
<td>N/A</td>
<td>N/A</td>
<td>11,813</td>
<td>1,992</td>
<td>39,375</td>
</tr>
<tr>
<td></td>
<td>Sao Vicente CA</td>
<td>315,000</td>
<td>N/A</td>
<td>20,400</td>
<td>N/A</td>
<td>N/A</td>
<td>27,000</td>
<td>6,200</td>
<td>0</td>
</tr>
</tbody>
</table>
4.4.4 Salt production

4.4.4.1 Guinea
In the Tristao MPA and the Kanfarandé CA, several production sites were visited during the field surveys. These visits enabled me to establish an average selling price of 1,000 Guinean CFA francs, about 100 CFA per kg (eq. 0.15 euros/kg). Production was estimated at 10 tonnes per site per year in Kanfarandé and only a few hundred kilograms in the Tristao MPA, where production is less important. The total annual production estimates are 200 tonnes for the Kanfarandé site and 25 tonnes for Tristao, which represent a turnover of 20 million CFA (30,000 euros) and 2.5 million CFA (3,800 euros) respectively. Operational costs of 10% of the turnover bring the GVA to 27,600 euros for Kanfarandé and 3,400 euros for Tristao.

Figure 4-7: Storage of hyper-salinized soil crust; production close to the Avicennia forest, main source of energy, in Kanfarandé (Guinea) (credit: A. Doumbouya)

4.4.4.2 Guinea-Bissau
In Guinea-Bissau, the salt production method is similar to the one in Guinea. However, evidence of production is anecdotal and no salt production sites were found during field surveys. Interviews with locals enabled me to estimate production volumes in Cacheu and Cacine at 3 tonnes and 1 ton, respectively. There is no production in Urok and Galinas.

4.4.4.3 Senegal
In Senegal, traditional salt production is similar to the method adopted in Guinea. However, there are dedicated sites where a more modern technique has been applied since the 1990s. This production is comparable to the saltworks in France and is based on solar evaporation
(see next figure). The brine is evaporated in small basins and produces about 1kg/day/m² (Ecoutin et al., 1999). The estimated production in the Langue de Barbarie MPA is 100 tonnes, which represents a turnover of 10 million of CFA francs (15,000 euros), and 200 tonnes in the comparison area of South Saint Louis (30,000 euros) Operational costs are estimated to be 10% of turnover.

Figure 4-8: Salt production in the Langue de Barbarie MPA as pictured on the entrance sign of the MPA (credit: T. Binet)

4.4.4.4 Synthesis

Table 4-11 shows the gross value added for each site. It should be noted that the value is directly attributed to the mangrove ecosystem which is the only ecosystem involved in this economic activity. Salt production has a very high GVA in Senegal sites. This is due to intensive salt production (100 tonnes and 200 tonnes respectively) in small areas of mangroves. In Guinea, salt production is much less important, with some sites spread over an immense area of mangrove in Kanfarandé and Tristao.
Table 4-11: Gross value added of salt production in mangroves

<table>
<thead>
<tr>
<th>Countries</th>
<th>Site</th>
<th>Production (kg)</th>
<th>Average price (CFA francs/kg)</th>
<th>Operational costs (CFA francs/kg)</th>
<th>GVA (CFA francs)</th>
<th>GVA (€)</th>
<th>Unitary GVA in mangrove (€/km²/yr)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Senegal</td>
<td>Langue de Barbarie MPA</td>
<td>100,000</td>
<td>100</td>
<td>10</td>
<td>9,000,000</td>
<td>13,800</td>
<td>16,627</td>
</tr>
<tr>
<td></td>
<td>Saint Louis CA</td>
<td>200,000</td>
<td>100</td>
<td>10</td>
<td>18,000,000</td>
<td>27,600</td>
<td>16,235</td>
</tr>
<tr>
<td>Guinea-Bissau</td>
<td>Cacheu MPA</td>
<td>3,000</td>
<td>100</td>
<td>10</td>
<td>270,000</td>
<td>400</td>
<td>1</td>
</tr>
<tr>
<td></td>
<td>Cacine CA</td>
<td>1,000</td>
<td>100</td>
<td>10</td>
<td>90,000</td>
<td>140</td>
<td>1</td>
</tr>
<tr>
<td>Guinea</td>
<td>Tristao MPA</td>
<td>25,000</td>
<td>100</td>
<td>10</td>
<td>2,250,000</td>
<td>3,400</td>
<td>25</td>
</tr>
<tr>
<td></td>
<td>Kanfarandé CA</td>
<td>200,000</td>
<td>100</td>
<td>10</td>
<td>18,000,000</td>
<td>27,600</td>
<td>397</td>
</tr>
</tbody>
</table>

4.4.5 Mangrove wood-cutting

4.4.5.1 Calculation method

To estimate the GVA of mangrove wood-cutting, the average selling price of wood is first calculated. For all sites, the average price per kilogram is around 50 CFA (eq. 0.08 euros). Operational costs were estimated at 10% of the selling price of the wood.

4.4.5.2 Senegal

The Langue de Barbarie MPA and its comparison area have mangroves which have been degraded but are still exploited by professional cutters. There are 30 professional cutters active in the MPA and 150 in the CA (not all of the latter being active). After meeting with these cutters, it appeared that they earn about one million CFA francs annually, with a selling price of wood of around 50 CFA francs per kg. The wood is primarily intended for sale in Saint Louis and is used for domestic use and partly for smoking fish. The mangrove cutting carried out is estimated to be 600 tonnes of wood in the MPA and 1,000 tonnes in the CA. These figures also consider the cutting of mangrove for domestic use.
4.4.5.3 Guinea-Bissau and Guinea

In Guinea-Bissau, mangrove wood cutting is very important in terms of volume and causes significant damage, although mangroves appear to be in a much better health status than in Senegal. Cut volumes are estimated at 4,500 tonnes in the Cacheu MPA, providing for the domestic use of 2,800 households in the area (at an average 1.2 tonnes per household per year, which is a conservative value), and cut for sale for building purposes (1,100 tonnes). Cutting in the Cacine CA is estimated at 2,000 tonnes for domestic use (consumption by 1,000 households) and selling wood. However, in recent years, migrant fishermen living on the other side of the border (in Tristao’s fishing camp of Katchek) have bought their wood for smoking fish into Guinea-Bissau. It was difficult to pinpoint the location of cuts, but these seem to take place outside the CA boundaries, between the Cacine CA and the Tristao MPA in Guinea. However, this is said to cause tremendous damage to the mangroves according to the local populations interviewed on the subject. Some canoes were spotted during the field survey around Cacine that were thought to be cutting wood for Tristao fishing camp (Figure 4-9).

Figure 4-9: Transport and storage of mangrove wood after cutting in Guinea (credit: T. Binet)

Mangrove cutting in Urok and Galinas is less important. In Urok, cutting practices have been limited by MPA management measures and the training of populations by the Tiniguena NGO\textsuperscript{36}. The population were trained to cut only the dead or minor parts of mangrove trees in order to facilitate its regeneration. The mangrove cutting was estimated at 410 tonnes and 210 tonnes, for the 340 and the 175 households that use mangroves as their main

\textsuperscript{36}The Tiniguena NGO is an NGO based in Urok whose mission is to develop the MPA and implement the management plan. It was established following a demand from the local population relating to the recovery of the stock of one shellfish species which is important for their religious ceremonies.
source of domestic energy. The cutting of mangrove wood for commercial purposes is negligible.

In Guinea, mangrove timber is important in both areas (though lots of wood now comes from Guinea-Bissau). Wood is mainly used to supply the fishing camp of Katchek (that relies heavily on provision of mangrove wood from Tristao for its smoking of fish), and the smoking houses in Kamsar, the main city, and the landing site in the Kanfarandé CA. Salt production also requires a large amount of wood. The estimated volumes of wood cut in the studied sites are estimated to be around 1,200 tonnes in the MPA and 3,000 tonnes in the CA (assuming 1,000 and 1,500 households respectively consume 1.2 tonnes of wood per year).

![Figure 4-10: Various uses of mangrove wood: preparation of palm oil and smoking of small pelagic fish (Guinea) (credit: T. Binet)](image)

**4.4.5.4 Synthesis**

Table 4-12 shows the GVA of mangrove cutting for each site. The GVA is directly attributed to the mangrove ecosystem which only supports this activity. The values per site highlight the importance of these practices in Guinea and Guinea-Bissau (Cacheu, Kanfarande, and Cacine above all). Values per unit area are greater in the two sites in Senegal, showing that each hectare of mangrove ecosystem is more heavily exploited, though the total value is much lower than in Guinea-Bissau and Guinea.
Table 4-12: Gross value added of mangrove wood cutting

<table>
<thead>
<tr>
<th>Countries</th>
<th>Site</th>
<th>Production (kg)</th>
<th>Average price (CFA francs/kg)</th>
<th>Operational costs (CFA francs/kg)</th>
<th>GVA (CFA francs)</th>
<th>GVA (€)</th>
<th>Unitary GVA (€/km²/yr)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Senegal</td>
<td>Langue de Barbarie MPA</td>
<td>600,000</td>
<td>50</td>
<td>5</td>
<td>27,000,000</td>
<td>41,500</td>
<td>50,000</td>
</tr>
<tr>
<td></td>
<td>Saint Louis CA</td>
<td>1,000,000</td>
<td>50</td>
<td>5</td>
<td>45,000,000</td>
<td>69,200</td>
<td>40,706</td>
</tr>
<tr>
<td>Guinea-Bissau</td>
<td>Cacheu MPA</td>
<td>4,500,000</td>
<td>50</td>
<td>5</td>
<td>202,500,000</td>
<td>311,500</td>
<td>735</td>
</tr>
<tr>
<td></td>
<td>Cacine CA</td>
<td>2,000,000</td>
<td>50</td>
<td>5</td>
<td>90,000,000</td>
<td>138,500</td>
<td>917</td>
</tr>
<tr>
<td></td>
<td>Urok MPA</td>
<td>410,000</td>
<td>50</td>
<td>5</td>
<td>18,450,000</td>
<td>28,400</td>
<td>366</td>
</tr>
<tr>
<td></td>
<td>Galinas CA</td>
<td>210,000</td>
<td>50</td>
<td>5</td>
<td>9,450,000</td>
<td>14,500</td>
<td>331</td>
</tr>
<tr>
<td>Guinea</td>
<td>Tristao MPA</td>
<td>1,200,000</td>
<td>50</td>
<td>5</td>
<td>54,000,000</td>
<td>83,100</td>
<td>618</td>
</tr>
<tr>
<td></td>
<td>Kanfarandé CA</td>
<td>3,000,000</td>
<td>50</td>
<td>5</td>
<td>135,000,000</td>
<td>207,700</td>
<td>2,986</td>
</tr>
</tbody>
</table>

4.4.6 Medicinal exploitation

Such domestic use of MCE products for medicinal uses cannot be quantified. However, these will be considered within the non-use valuation when considering the biodiversity of MCEs and the current or future availability of such products for religious considerations.

4.4.7 Tourism

4.4.7.1 Context and calculation method

The method used here revolves around the evaluation of the GVA of tourism operators: the GVA of tour operators offering trips to the sites studied; and the GVA of seaside tourism operators (with an estimate made as to the part of the revenues that is generated due to the existence of the MCEs). The GVA of tourist tours in the MPA was evaluated from their spending on activities related to ecosystems in the region (e.g. boat trips, fishing, or bird watching). These were estimated through direct interviews with tourists visiting the sites. Based on about 20 answers about spending and budgets of stay, it was then possible to
calculate an average expense per tourist. This was (when possible) crosschecked with interviews with tourist operators on the number of tourist visits and estimated annual turnover of their businesses.

### 4.4.7.2 Senegal

Many tourists come for the day to visit the MPA and then leave without staying overnight. Annually, about 5,000 people go to the MPA for a day and generate a gross value added of about 20,000 CFA francs per person (30 euros), which represents a total of 10 million CFA francs (15,300 euros).

The GVA of tourism in the MPA combines the budget of stay of tourists in the MPA and the expenses related to transportation from St Louis (taxi or shuttle and the boat to cross the river). From interviews it appears that tourists come to stay in the park solely to enjoy the ecosystems of the area (the beaches and estuary of the Senegal River) because there is no other activity in the area. For this reason I calculate 80% of the budget of stay relates to the benefits provided by the MCEs. With an average attendance of 1,500 people per year for the two camps and an average budget of stay and local transportation to the camps of 80,000 CFA francs (eq. 123 euros) i.e. a GVA per person of 50,000 CFA francs - eq. 77 euros) the GVA of tourism in the area is estimated at 150,000 euros.
The GVA of tourism in the CA of South Saint Louis was estimated from the revenues spent that are related to the region’s ecosystems activities (e.g. excursions, fishing, observation of fauna and flora). However, the proportion of revenues related to the MCE is lower because the MCE is not the only reason for staying in these hotels (unlike the tourist camps in the MPA). Tourists also stay in the CA because of its proximity to Saint Louis and the opportunity it offers to visit the city. The proportion of tourists enjoying the benefits of the MCE in the CA (approximately 3,000 individuals in 1,000 families or groups) and the average budget of their stay are estimated at 120,000 CFA francs (180 euros) per family or group. As a result, the GVA of tourism in the area is estimated at 147,700 euros.

**4.4.7.3 Cape Verde**

The population visiting the MPA every year can be estimated at about 1,500 tourists. The price of the boat tour to the island is between 30 and 200 euros per person, depending on whether the trip is on a fishing boat or on a tourist motorboat (an average of 100 euros per person). The GVA of tourism in the MPA is therefore about 40,000 euros for boat tours. This value is probably underestimated because it was not possible to collect information on the
number of divers attending the island from diving centres. Also, the number of sailors visiting the island on their own vessel is unknown.

The average budget of tourists visiting the CA is 250 euros. According to a specific survey carried out with a dozen tourists met in the camps with regards to their motivations of stay, MCEs represent 40% of the willingness to stay on the site.\footnote{This could have been furthered through a more in-depth analysis about the rationale of stay for tourists.}

In addition to these expenses, about 1,000 people are engaged in optional activities directly related to the MCEs for which they pay extra (e.g. boat tours, hikes along the coastline with a guide, scuba diving)\footnote{This figure is an estimate from interview with tourism operators in the area.}. The GVA for these is about 100 euros per person.

![Figure 4-12: Foya Branca resort in Sao Pedro, Sao Vicente island (Cape Verde) (credit: NATO)](image)

### 4.4.7.4 Synthesis

Table 4-13 shows the GVA of seaside tourism and boat tours for each site considered. The value of tourism varies greatly according to the site. Tourism in the CA Sao Vicente generates significant revenues of 1.2 million euros per year due to the existence of one large resort with a large number of tourists.
<table>
<thead>
<tr>
<th>Countries</th>
<th>Site</th>
<th>Activity</th>
<th>Number of individual</th>
<th>GVA stay/trip per (CFA francs unless mentioned)</th>
<th>Contribution of the MCEs in the activity (%)</th>
<th>GVA tourism (CFA francs)</th>
<th>GVA tourism (€)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Senegal</td>
<td>Langue de Barbarie MPA</td>
<td>Coastal tourism and accommodation</td>
<td>1,500</td>
<td>50,000</td>
<td>80%</td>
<td>60,000,000</td>
<td>92,308</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Nautical activities, boat tours</td>
<td>1,000</td>
<td>20,000</td>
<td>100%</td>
<td>20,000,000</td>
<td>30,769</td>
</tr>
<tr>
<td></td>
<td>Saint Louis CA</td>
<td>Coastal tourism and accommodation</td>
<td>3,000</td>
<td>40,000</td>
<td>50%</td>
<td>60,000,000</td>
<td>92,308</td>
</tr>
<tr>
<td>Cape Verde</td>
<td>Santa Luzia MPA</td>
<td>Boat tours</td>
<td>1,500</td>
<td>75 €</td>
<td>100%</td>
<td>N/A</td>
<td>112,500</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Diving</td>
<td>400</td>
<td>100 €</td>
<td>100%</td>
<td>N/A</td>
<td>40,000</td>
</tr>
<tr>
<td></td>
<td>Sao Vicente CA</td>
<td>Coastal tourism and accommodation</td>
<td>12,000</td>
<td>250 €</td>
<td>40%</td>
<td>N/A</td>
<td>1,200,000</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Boat tours diving</td>
<td>1,000</td>
<td>100 €</td>
<td>100%</td>
<td>N/A</td>
<td>100,000</td>
</tr>
</tbody>
</table>
In terms of return per unit of area for each ecosystem, the values for each site are also highly variable (Table 4-14). The high economic value of the Sao Vicente CA has led to a very high unitary value as well. This is especially true for beaches, where seaside holiday activities are concentrated, and for coral bottoms, which are targeted by boat tours and diving activities.
### Table 4-14: Unitary values of tourism activity by ecosystem

<table>
<thead>
<tr>
<th>Countries</th>
<th>Site</th>
<th>Total per site (€)</th>
<th>Estuaries and channels (€/km²/yr)</th>
<th>Seagrass meadows (€/km²/yr)</th>
<th>Mangrove (€/km²/yr)</th>
<th>Mudflats (€/km²/yr)</th>
<th>Beaches (€/km²/yr)</th>
<th>Rocky bottoms (€/km²/yr)</th>
<th>Coral bottoms (€/km²/yr)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Senegal</td>
<td>Langue de Barbarie MPA</td>
<td>Coastal tourism and accommodation</td>
<td>3,936</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>11,598</td>
<td>N/A</td>
<td>N/A</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Nautical activities, boat tours</td>
<td>1,007</td>
<td>24,640</td>
<td>4,824</td>
<td>112</td>
<td>1,272</td>
<td>N/A</td>
<td>N/A</td>
</tr>
<tr>
<td></td>
<td>Saint Louis CA</td>
<td>Coastal tourism and accommodation</td>
<td>3,659</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>21,414</td>
<td>N/A</td>
<td>N/A</td>
</tr>
<tr>
<td>Cape Verde</td>
<td>Santa Luzia MPA</td>
<td>Boat tours</td>
<td>N/A</td>
<td>14,501</td>
<td>N/A</td>
<td>N/A</td>
<td>6,481</td>
<td>1,495</td>
<td>76,250</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Diving</td>
<td>N/A</td>
<td>62,064</td>
<td>N/A</td>
<td>N/A</td>
<td>144,444</td>
<td>10,652</td>
<td>0</td>
</tr>
<tr>
<td></td>
<td>Sao Vicente CA</td>
<td>Coastal tourism and accommodation</td>
<td>N/A</td>
<td>7,758</td>
<td>N/A</td>
<td>N/A</td>
<td>3,148</td>
<td>1,019</td>
<td>69,189</td>
</tr>
</tbody>
</table>
4.4.8  Direct use values: key results

The direct use values of the sample of 5 MPAs represent a total of 6.6 million euros. The Tristao MPA generates the most important direct use values with almost 3.5 million euros, followed by the Cacheu MPA (1.3 million euros) and Santa Luzia MPA (1.1 million euros). It is surprising to note that the ecosystem with the most important value (irrespective of surface area) is not the mangrove or the coral bottoms but rather the mudflats (2.24 million euros). This result is mainly caused by the heavy commercial fishing in Tristao which is practiced on mudflats and has a large economic value (1.79 million euros). It is then followed by mangroves (1.39 million euros), estuaries and channels (1.19 million euros) and beaches (886,000 euros). These values are shown in Figure 4-13.

![Figure 4-13: Direct use values in the studied MPAs and their CAs (euros)](image)

The values of CAs have been calculated based on the unitary value for each ecosystem multiplied by the surface area of each ecosystem in the MPA. Hence, the values are
compared with the same reference area, thus enabling comparison between MPA and CA values.

In the CAs, the most valuable ecosystems are the mangroves (2.66 million euros), estuaries (2.21 million euros) and the beaches (1.77 million euros). The comparison between MPAs and their related CAs shows a difference of direct use value of 2.4 million euros in favour of the CAs. All CAs have higher values than the MPAs they are compared to, except for the Urok MPA. This is due to intense extractive practices, such as commercial and subsistence fishing and mangrove wood cutting. However, these estimates are only provided on a yearly basis and do not consider the longer economic costs of such pressures on the resources. An economic valuation carried out over a long-term period would highlight such unsustainable practices, but this is outside the scope of my research here.

4.5 Discussion
Various shortcomings in the method and protocol adopted for direct use values calculation have emerged from the research. They can be inherent to the general approach adopted (the TEV approach) or related to the research itself (valuation methods applied and protocol chosen). The choice was made not to develop the first category of shortcomings that are generic and relate to the TEV approach. Rather, I thought it important to discuss the shortcomings and limits of the method proposed and suggest some improvements. The following discusses the limits of the direct use values calculation.

4.5.1 Data-poor situations for direct use valuation
The most important barrier to the calculation of direct use values was the lack of data available on the past levels of exploitation in order to estimate average volumes of production and price. This was overcome by the adoption of the growth added-value estimate and with successive interviews with local experts in order to estimate the required data. Further valuation should better consider this data-poor situation for improved quality and accuracy of values.

4.5.2 Non-monetised economies and subsistence activities
The apparent absence of a monetised economy in some sites was also a limit to my method. In the Bijagos for instance, the economy there is mostly non-monetised and the trade is based on exchange of products or based on subsistence activities only. During my visits in the sites, however, I found that some commodities were marketed (e.g. dried
oysters or wood carvings to be sold to the mainland markets), which made me think that money was not totally absent for these communities and that it could be possible to consider subsistence activities on the same price basis as commercial activities (such as for fisheries or wood-cutting). The only alternative option for non-monetised societies (applied in the Pacific islands for instance) consists in estimating value based on protein equivalent. I found that this option was not acceptable here because fish and shellfish are key to the food diet of Bijagos populations (and also bear religious value); they cannot be substituted by rice or other agricultural products. It then seemed sounder to consider the commercial market as the best way to value these subsistence activities. The same applied to wood-cutting activities since mangrove is vital for the traditional housing and cooking and cannot be replaced by a wood-equivalent.
5 Chapter 5: indirect use values

5.1 Introduction
The aim of this chapter is to estimate the indirect use values of MPAs, and compare them to those of CAs. The chapter first presents the indirect uses considered in the MPAs and CAs and then provides further details on their magnitude. The chapter then gives an overview of the specific calculation methods used for each of these indirect uses in the specific context of the case study.

5.2 Types of indirect uses
The types of indirect uses assessed here include the following: coastal protection and erosion control; fisheries biomass production; carbon sequestration; and water and waste treatment.

With regards to coastal protection, marine and coastal ecosystems (MCEs) are natural barriers for the shore against erosion by waves, current and extreme events such as tsunamis and storms. They reduce coastal erosion by absorbing 70 to 90% of the wave energy (Wells et al., 2006), and lessen the damage in case of severe weather (e.g. hurricanes, tropical storms). Six ecosystems are considered here: seagrass meadows, mangroves, beaches, mudflats, rocky reefs and coral bottoms.

MCEs are also responsible for fish biomass production. They support a rich biodiversity and are involved in the life cycles of a large selection of marine animals (e.g. crustaceans, fish and molluscs). They play a role as nurseries during the larval or juvenile cycle. They also provide habitat to numerous species during their adult stage as well as acting as a source of shelter or main food. They can be the main sites for breeding and nesting (Barbier, 2007; Rönnback, 1999; Baran, 1995). Some species which benefit from these functions are of commercial interest through commercial and subsistence fisheries. Thanks to these various productivity functions, MCEs thus contribute to the maintenance of these fisheries and their values. This indirect use is considered here in addition to the fisheries’ direct uses, in order to introduce some degree of sustainability when valuing fisheries.

Third, carbon dioxide (CO₂) is captured by marine and coastal ecosystems. This marine and coastal carbon (also known as “blue carbon”) is sequestered and stored by various

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39 Coastal protection offered by rocky and coral bottoms will be discussed when these ecosystems are compared to rocky and coral reefs.
ecosystems including mangroves, seagrass meadows and saltmarshes (and coral reefs to a lesser extent) (Box 2). This function has gained increasing attention from policymakers thanks to its potential for emission compensation (Laffoley, 2009). Carbon capture, following the adoption of the Kyoto Protocol in February 2005, now has an economic value. Countries that have ratified the Protocol are committed to reducing their emissions of greenhouse gases, including CO$_2$. In fact, each country has implemented various practices to reduce emissions of Greenhouse Gases (GHG), using different mechanisms under the Kyoto Protocol (such as REDD+, NAMAs, CDM – see Appendix 6 for more details on the compliance market).

**Box 2: the carbon sequestration process**

Appendix 6 contains a briefing note on blue carbon and the opportunities for blue carbon projects development in MPAs.$^{40}$

The ocean is often cited as the most important carbon sink on earth because of its high capacity to absorb CO$_2$. Over the last two centuries, the oceans have stored about 500 Gt of CO$_2$ out of the 1,300 Gt of CO$_2$ emitted by human activities, nearly 40% of emissions (Metz et al., 2005). Storage of blue carbon can happen in one of three ways. First by simple dissolution of CO$_2$ in water according to the reaction:

$$CO_2 + H_2O \rightarrow H_2CO_3 \rightarrow HCO_3^- + H^+ \rightarrow CO_3^{2-} + 2H^+$$

It can also happen through the uptake of CO$_2$ in the photosynthesis of phytoplankton and algae in upper metres of surface water, or by biogenic calcification of tests or limestone skeletons, using the calcium dissolved in seawater ($Ca^{2+}$) and carbonate ion (CO$_3^{2-}$) to give calcium carbonate ($CaCO_3$) according to the reaction:

$$Ca^{2+} + CO_3^{2-} \leftrightarrow CaCO_3$$

In the short term the calcification process produces CO$_2$ but it is quickly trapped by photosynthetic organisms in reefs as well as by the buffering effect of the ocean.

Fourth, marine ecosystems are very sensitive to the quality of coastal waters, though they contribute significantly to their purification. The role of coastal ecosystems in water and waste treatment is threefold (MEA, 2005):

$^{40}$This Appendix is adapted from a briefing note and keynote speech prepared for a workshop held in Montenegro in 2013 on the opportunities for blue carbon project development in the Mediterranean MPAs.
- They act as filters of coastal waters, fix sediments that are transported in the water and hence reduce the turbidity of waters. This reduces the deposition of sediments on other neighbouring habitats (coral bottoms and seagrass meadows) (which require clear water for their development). Through their stilt roots, mangroves act as filters of inland waters, limiting the amount of suspended matter discharged into the oceans.

- They ensure the cycling of water through tidal recirculation, thus enabling treatment by micro-organisms contained in the water; and

- They act as buffers to terrestrial chemical pollution.

5.3 Specific method

The availability of data for ecological functions is very limited. The only available data at local level is related to fisheries biomass production. For this reason, the transfer of benefit method is used in most of the indirect uses considered. Pascual et al. (2010) distinguishes 4 different types of benefit transfers (BTs): 1) the unit BT that involves estimating the value of an ecosystem at a destination site by simply using the origin value and multiplying by the surface unit; 2) the adjusted unit BT that involves making simple adjustments to the transferred unit values to reflect differences in site characteristics (generally income or price levels); 3) the value or demand function transfer that uses functions estimated through valuation applications (for travel cost and hedonic pricing) in an origin site with information on parameter values for the destination site to transfer values; the parameter values calculated in the destination site are thus plugged into the value function to calculate a transferred value that is adapted to the studied site; and 4) the meta-analytic function transfer that uses a value function estimated from multiple study results with information on parameter values for the destination site to estimate values. The complexity of applying these methods increases in the order in which they are presented. But the accuracy of the values calculated increases accordingly.

The benefit transfer method used in my study has been limited to the ‘adjusted unit transfer’, which was thought necessary to transfer values from developed countries (for most transferred values) to developing countries in West Africa. The adjustment was made through the GDP per capita, which was easily available for all countries (different from income per households or price levels information, which have been suggested as adjustment parameters by Pascual et al. (2010), but which are not available in West Africa). The two other function transfer methods require more local data on parameter
values to be applied, which was the most important limitation in my case. As a result, the implementation of such methods was difficult in this region. Further data collection is required for specific parameters to be able to apply these methods.

In my study, however, I have developed a more accurate method than the ‘adjusted unit transfer’ one to estimate indirect use values in data-deficient sites. This was applied for the fisheries biomass production. I have carried out a meta-analysis of ecological parameters (on the biomass production in my case) and then applied the value function to my site. This was used for the production function method but could possibly be used for other methods such as replacement costs (e.g. for coastal protection or water treatment): ecological or hydro-dynamical parameters from neighbouring areas transferred to the destination sites and replacement cost value estimated through field surveys.

The Table 5-1 details methods used for the calculation of indirect use values.

<table>
<thead>
<tr>
<th>Use</th>
<th>Values</th>
<th>Data collection and method for calculation</th>
</tr>
</thead>
<tbody>
<tr>
<td>Indirect use</td>
<td>Formation, maintenance and protection of beach, estuaries and coastline</td>
<td>Transfer of values for coastal protection of mangroves, seagrass and coral reefs (values will be adjusted to the economic context of West Africa)</td>
</tr>
<tr>
<td></td>
<td>Biodiversity and ecosystem productivity; fisheries biomass production</td>
<td>Estimation of fisheries biomass for the main targeted species and percentage of adult in stock</td>
</tr>
<tr>
<td></td>
<td>Climate regulation and carbon sequestration</td>
<td>Estimation of captured carbon by ecosystems and value based on the global market for carbon</td>
</tr>
<tr>
<td></td>
<td>Water and waste treatment</td>
<td>Transfer of values calculated with replacement cost method (adjustment to the economic context of West Africa)</td>
</tr>
</tbody>
</table>

Once the value is know, it is also important to weight it according to the health status of the ecosystem considered, which influences the service delivery. This is the only way to distinguish between protected and unprotected site for values that have been transferred to a similar economic context. In order to take these differences into account when transferring values, coefficients are applied based on health status using an index (0 to 5, with 0 as totally degraded habitat and 5 as habitat in excellent condition). The Table 5-2 shows the correspondence of the health status of ecosystems and the coefficient to be applied for coastal protection of mangrove.
In the specific case of mangroves, the health status can be characterized by the dominant species of the ecosystem and the percentage of vegetation cover: a mangrove in excellent condition is dominated by *Rhizophora*, a moderately productive mangrove (50%) is a mangrove whose major species is *Avicennia*. A highly degraded mangrove (10% of service yield) is bare of vegetation, such as “tanne”.

For other ecosystems, such as seagrass or mudflats, it is very difficult to estimate their health status by measurement in the field. Besides, there are no references providing such information for West Africa. As a consequence, values calculated for the services of these ecosystems are not indexed to their health statuses.

5.4 Results

5.4.1 Coastal protection and erosion control

5.4.1.1 Seagrass meadows

Seagrass meadows absorb some of the energy of waves and therefore limit their impact on the shore. Some seagrass species are organised in “mattes” (such as *Posidonia oceanica*) that form large areas very close to the surface. These substantially diminish the power of waves and reduce the speed of the current along the shore. A study by Fonseca and Cahalan (1992) on the wave reduction by four types of seagrass shows that wave energy is reduced by 40% after passing over a seagrass meadow.

On coastlines where erosion is intense, artificial structures can also be put in place to limit the action of the waves. These breakwater structures can be erected on the seabed or anchored as floating structures. They are placed in parallel to the coast, impeding swell and therefore reducing wave energy by 40-50% (Samat, 2007).

The estimated value of protection can be extracted from the costs of installation and maintenance of these breakwaters (cf. replacement cost method in chapter 2). However, this value can only be applied for seagrass meadows that are exposed to the waves,
which is not the case for all meadows. For instance in the Langue de Barbarie MPA and the Cacheu MPA, seagrass meadows are often sheltered from waves as most meadows are located in the estuary. They are however highly exposed to tidal currents which can create as much erosion and sediment transport as waves and, in this regard, can be partly considered as providing a service of erosion control as well.

For Cape Verde, seagrass is very important because the islands have no other barrier against the ocean swell, which is very strong in the offshore area as it is hit by heavy swells all year round (and especially during the trade winds season from November to April). Only the windward seagrass (about 50% of the total seagrass area) is exposed to waves and thus contributes to coastal protection. The leeward seagrass is less important for this service. However, leeward seagrass can, as for the estuaries, contribute to reducing the power of currents and can reduce the effects of waves that go around the island and hit the shore sideways.

In the French Caribbean islands, the average annual cost of breakwater protection is approximately 714 euros per linear metre of coastline or 714,000 euros per km of coastline (Failler et al. 2010). In West Africa, I have considered the replacement cost of seagrass ecosystem based on the relative cost of living in these countries, as measured by the GDP/capita/yr index (19,607 euros for Martinique, 2,595 euros in Cape Verde, 1,212 euros in Senegal, 454 euros in Guinea-Bissau, and 833 euros in Guinea). Based on this index and the calculation of service per unit of GDP/capita/yr, values are estimated. The average value for the protection of seagrass per unit of GDP/capita is 36 euros which represents 92,340 euros/km/yr for Cape Verde, 43,632 euros/km/yr for Senegal, 16,344 euros/km/yr for Guinea-Bissau and 29,988 euros/km/yr for Guinea.

The distance covered by seagrass is estimated based on the recognition that seagrass meadows extend along the coastline but not necessarily perpendicular to the coastline. For the distance of seagrass to be considered, I therefore used an average ratio of two metres in length of seagrass for one metre of linear coastline in width.

5.4.1.2 Mangroves

Mangroves are the most important ecosystem for coastal protection. Forming a physical barrier to wind and water, they greatly reduce the impacts of erosion. These ecosystems also moderate flooding (Spurgeon et al., 2004). For example, the areas most affected by the tsunami in Thailand in 2004 were those where mangroves had been removed by human activities. The dense root system limits sedimentation transported by runoff
from upstream land being washed away to the sea. The accumulation of sediments in the mangrove stabilizes the bank.

The replacement cost study by Spurgeon et al. (2004) in Samoa, estimated the value of mangrove protection at 188,438 euros/km²/yr. Another study on the coast of Belize evaluated this same function at 227,146 euros/km²/yr (Cooper et al., 2008). Considering the income per capita of these countries expressed in GDP/capita/yr (4,320 euros for Samoa, 6,480 euros for Belize, 1,212 euros for Senegal, 454 euros for Guinea-Bissau, and 833 euros for Guinea), I can establish protection values for Senegal, Guinea and Guinea-Bissau. The average value of protection per unit of GDP/capita/yr is 39 euros, and the estimated values are 47,268 euros/km²/yr for Senegal, 17,706 euros/km²/yr for Guinea-Bissau and 32,487 euros/km²/yr for Guinea.

As specified in the methodology section, mangroves with predominant Rhizophora contribute 100% of possible coastal protection services. Avicennia contributes much less to physical protection during the dry season (they are not below sea level at this time of the year). However, it still fixes the sediments through its root system. It does however play a particularly important role in the fight against erosion during the rainy season, when washouts occur upstream. It is thus appropriate to consider that an ecosystem with a predominance of Avicennia offers 40% of the maximum value of estuarine and coastal protection. When the mangrove has been removed, the soil can be either bare or grassy. I consider that this degraded mangrove which is poorly vegetated offers only 10% of the value of protection against erosion (as it does provide a flat surface that allows sediment to settle in the case of minor washouts). Bare deforested mangrove does not contribute to coastal and estuarine protection at all.

Based on these results, Table 5.3 details the values for each site.
Table 5-3: Coastal protection value of mangroves in studied sites

<table>
<thead>
<tr>
<th>Countries</th>
<th>Site</th>
<th>Unitary value for 100% protection (€/km²/yr)</th>
<th>Rhizophora surface area (km²), 100% of protection</th>
<th>Avicennia surface area (km²), 40% of protection</th>
<th>Grassy deforested mangrove surface area (km²), 10% of protection</th>
<th>Value per site (€)</th>
<th>Unitary value per site (€/km²/yr)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Senegal</td>
<td>Langue de Barbarie MPA</td>
<td>47,268</td>
<td>0.1</td>
<td>0.2</td>
<td>0.5</td>
<td>11,013</td>
<td>13,269</td>
</tr>
<tr>
<td></td>
<td>Saint Louis CA</td>
<td>47,268</td>
<td>0.0</td>
<td>0.6</td>
<td>1.1</td>
<td>16,544</td>
<td>9,732</td>
</tr>
<tr>
<td>Guinea-Bissau</td>
<td>Cacheu MPA</td>
<td>17,706</td>
<td>131.0</td>
<td>119.0</td>
<td>174.0</td>
<td>3,470,376</td>
<td>8,185</td>
</tr>
<tr>
<td></td>
<td>Cacine CA</td>
<td>17,706</td>
<td>41.0</td>
<td>45.0</td>
<td>65.0</td>
<td>1,159,743</td>
<td>7,680</td>
</tr>
<tr>
<td></td>
<td>Urok MPA</td>
<td>17,706</td>
<td>36.6</td>
<td>23.4</td>
<td>17.7</td>
<td>845,107</td>
<td>10,877</td>
</tr>
<tr>
<td></td>
<td>Galinas CA</td>
<td>17,706</td>
<td>7.8</td>
<td>12.5</td>
<td>23.5</td>
<td>268,246</td>
<td>6,124</td>
</tr>
<tr>
<td>Guinea</td>
<td>Tristao MPA</td>
<td>32,487</td>
<td>27.7</td>
<td>35.5</td>
<td>71.4</td>
<td>1,591,844</td>
<td>11,832</td>
</tr>
<tr>
<td></td>
<td>Kanfarandé CA</td>
<td>32,487</td>
<td>3.3</td>
<td>31.9</td>
<td>34.3</td>
<td>634,588</td>
<td>9,124</td>
</tr>
</tbody>
</table>

5.4.1.3 Beaches

Beaches play an important role in erosion control. They help stabilize sediments and allow retention of the soil. Indeed, vegetated dune beaches can help to hold the shoreline, through the root systems of plants that grow there, (Landry et al, 2003). Although this service has not been evaluated directly, many studies have estimated the benefits derived from the implementation of programmes against erosion which preserve beaches and dunes (Landry et al, 2003. Kriesel and Landry, 2004, Huang et al, 2007. Whitehead et al, 2008, Morgan and Hamilton, 2010; Barbier et al, 2011). In particular, the service of erosion control was estimated at 4.45 US dollars/household/yr based on the cost of a programme to preserve a beach in the United States (Huang et al., 2007). This value was estimated through contingent valuation; however this depends on a variety of variables beyond the cost of living in one country (see methodology chapter on the advantages and limits of the contingent valuation method), and so makes the value of the transfer value method to such a different country very uncertain. This value will hence not be applied in this case.
5.4.1.4 Mudflats and rocky bottoms

In mudflats, sediments are not retained by the root systems as in the case of mangroves and nothing impedes tidal currents as with seagrass meadows. Accordingly, they are not thought to contribute to erosion control and are thus not considered here.

The bedrock is involved in the reduction of the wave, just as mangroves or coral reefs do (Failler et al., 2011). On sandy or muddy shores, rocky points along the shore prevent coastal erosion and transport by waves and tidal currents of the soft substrates. Yet, no reference in the international literature was found that provided an estimated value for this type of service. As a result it was not possible to provide an economic value for such a service in West African MPAs which have such rocky points. This should be further investigated in future work on the case of West African ecosystems.

5.4.1.5 Coral bottoms

If coral reefs provide significant protection against erosion, coral bottoms as they are present in Cape Verde offer much less protection against erosion and the effect of waves since they do not provide a barrier to waves. In the case of a storm, coral formations on bedrocks have only a minor impact on the strong swell (though they can reduce its energy in shallow areas). The coral bottoms therefore play a similar role to rocky bottoms. However, as for the rocky bottom ecosystem, there is no value in the international scientific literature on such service and so it will not be considered here.

5.4.2 Fishing biomass production

The evaluation of this service focuses solely on the species of fisheries interest because it is very difficult to estimate the value of a species that is not commercially exploited. Also, only the individuals that have reached adulthood and were taken with the fishing gear of artisanal fisheries (traps, nets, lines) are considered here. This represents the biomass of MCEs that can be exploited by fisheries each year.

The ecosystems contribute to the productivity of a much greater biomass that is not captured by fisheries. This biomass productivity is not accounted for here. However, as this unexploited biomass may also contribute to fisheries biomass by providing food for exploited species, the fisheries biomass productivity calculated here should be considered as a minimum.

Productivity estimates are made based on data from the international literature and, when possible, from the references available on West African ecosystems. Four
ecosystems are considered here: the channels and estuaries, seagrass meadows, mangroves, mudflats, and coral bottoms.

To avoid double counting, I deduct the biomass already caught annually by all fishermen (commercial, subsistence and sport fishermen). When the biomass exploited by fisheries is higher than the estimated fisheries biomass productivity, this latter value is zero (as the value cannot be negative). Three factors may explain why the biomass caught is greater than the fishable biomass: a) due to transfers of fisheries biomass from neighbouring ecosystems to the ecosystem considered b) due to the exploitation of fish not included in the fisheries biomass (or bycatch of juvenile non-commercial species), or c) because the estimates of biomass productivity are less than the actual productivity of West African ecosystems. In these three cases, the value of the biomass is set to zero and the annual productivity of the ecosystem considered nil.

5.4.2.1 Estuaries and channels

The importance of estuaries as nurseries for communities of juvenile fish is well known (Baran et al., 1995, 1999). According to Baran (1995, p. 186), "several factors combine in complex ways to explain the abundance of post-larvae and juveniles in estuaries and the nursery role played by them. The key two factors are the concentration of trophic resources and the turbidity. The post-larval and juvenile fish are abundant in the estuary because of trophic resources [-namely food-] that are varied and adapted to their poor ability to capture (...). Turbidity is considered another important explanatory factor because it greatly limits the juvenile predation by larger specimens." Other explanatory factors include the fact that the shallow intertidal areas also limit predation (Blaber 1980; Kneib, 1987) and that the diversity of habitats offers multiple refuges for juveniles.

This very important ecological role of estuaries in protecting juveniles suggests estuaries also contribute to the increase of the adult population in this ecosystem. However, the abundance of juveniles does not necessarily reflect an abundance of adults (Beck et al., 2001). Thus even if, as Albaret (2006) notes, the specific richness (in terms of the number of different species) of estuaries and channels of West Africa is significant (see Table 5.4), the fisheries biomass is still less than that of mangroves.
Table 5-4: Number of species for some estuaries in West Africa (adapted from Albaret, 2006; Diouf, 1996)

<table>
<thead>
<tr>
<th>Site (country)</th>
<th>Number of species</th>
<th>Sources</th>
</tr>
</thead>
<tbody>
<tr>
<td>Langue de Barbarie (Senegal)</td>
<td>111</td>
<td>Diouf et al., 1996</td>
</tr>
<tr>
<td>The Gambia river (the Gambia)</td>
<td>89</td>
<td>Daget, 1960; Dorr et al., 1985</td>
</tr>
<tr>
<td>Rio Buba (Guinea-Bissau)</td>
<td>92</td>
<td>Kromer et al., 1994</td>
</tr>
<tr>
<td>Fatala (Guinea)</td>
<td>102</td>
<td>Baran, 1995</td>
</tr>
</tbody>
</table>

Villanueva (2004) estimated the total raw biomass weight $B$ for fisheries species using ecotrophic modelling (through *Ecopath* software). This biomass was estimated based on the annual catch $Y$ and fishing mortality $F$ ($B = Y / F$). Biomass for fisheries in the Gambia River and Sine-Saloum Delta is $9.1 \text{ t}/\text{km}^2$ for fish. Crabs and shrimp, other exploited groups, represents $7.2 \text{ t}/\text{km}^2$ in the Gambia River, and $4.7\text{ t}/\text{km}^2$ in the Sine-Saloum. In total, the average value of estimated biomass for estuaries and channels calculated from the literature is about $15 \text{ t}/\text{km}^2$, or $8,650 \text{ euros}/\text{km}^2/\text{yr}$ ($15,000 \text{ kg per km}^2 \text{ at } 0.57 \text{ euros/kg}$). This value extracted from the literature is considered to be identical for the purposes of this study for all estuaries ecosystems in the West Africa region and will be used here as the fisheries biomass production value.

In Table 5.5, the unitary value of unexploited fisheries biomass is obtained by deducting the annual fisheries value in estuaries from the value of total fisheries biomass.

---

$^4$The average price of fish caught is 375 CFA/kg (eq. 0.57 euros) which reflect the equally split proportion of species between demersal species of high value (at 500 CFA/kg) and small pelagic species of low value (at 250 CFA/kg).
Table 5-5: Value of fisheries biomass of estuaries

<table>
<thead>
<tr>
<th>Countries</th>
<th>Site</th>
<th>Total fisheries biomass productivity (€/km²/yr)</th>
<th>Annual fisheries value in estuaries (€/km²/yr)</th>
<th>Unitary value of unexploited fisheries biomass (€/km²/yr)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Senegal</td>
<td>Langue de Barbarie MPA</td>
<td>8,650</td>
<td>117</td>
<td>8,500*</td>
</tr>
<tr>
<td></td>
<td>Saint Louis CA</td>
<td>8,650</td>
<td>48</td>
<td>8,600</td>
</tr>
<tr>
<td>Guinea-Bissau</td>
<td>Cacheu MPA</td>
<td>8,650</td>
<td>2,749</td>
<td>5,900</td>
</tr>
<tr>
<td></td>
<td>Cacine CA</td>
<td>8,650</td>
<td>2,626</td>
<td>6,000</td>
</tr>
<tr>
<td></td>
<td>Urok MPA</td>
<td>8,650</td>
<td>2,622</td>
<td>6,000</td>
</tr>
<tr>
<td></td>
<td>Galinas CA</td>
<td>8,650</td>
<td>3,460</td>
<td>5,200</td>
</tr>
<tr>
<td>Guinea</td>
<td>Tristao MPA</td>
<td>8,650</td>
<td>3,094</td>
<td>5,600</td>
</tr>
<tr>
<td></td>
<td>Kanfarandé CA</td>
<td>8,650</td>
<td>8,238</td>
<td>400</td>
</tr>
</tbody>
</table>

*the figures in this column (and columns with the same title in tables below) are rounded.

The results of these unitary values are very similar except for Kanfarandé CA. The most important values are observed in Senegal, followed by Guinea-Bissau and Tristao. The results highlight the high productivity of estuaries and their importance for fisheries. In Kanfarandé, since the largest part of fisheries occurs in estuaries, the unitary value is very low.

This method could be improved with an in-depth research and quantification of the precise biomass production in the ecosystems studied (or all least a quantification of estuaries production by large ecosystems – in Senegal on the one hand and in Guinea and Guinea-Bissau on the other hand). Some estuaries, because of their specific hydrological regime and morphology, are likely to be well above 8,650 euros/km²/yr.

5.4.2.2 Seagrass meadows

Seagrass meadows support a much lower fisheries biomass than those recorded for other ecosystems. Robertson and Duke (1990) note that seagrass in northern Australia have a biomass production of fish between 4 and 10 times lower than those of mangroves. Thayer et al. (1987) also note that the mangroves of Florida have a fish abundance that is 35 times greater than the adjacent seagrass meadows. The seagrass
ecosystem is estimated to produce a fisheries biomass of 1.9 t/km² per year (Martin and Coopers, 1981; Gullström and Dahlberg, 2004). Given the prevalence of demersal species in seagrass, unlike estuaries, the average price fixed for the value of fishable biomass was set at 450 CFA/kg (eq. 0.69 euros).

Table 5-6: Value of fisheries biomass productivity of seagrass meadows

<table>
<thead>
<tr>
<th>Countries</th>
<th>Site</th>
<th>Fisheries biomass production value(€/km²/an)</th>
<th>Fisheries value in seagrass meadows (€/km²/yr)</th>
<th>Unitary value of unexploited fisheries biomass (€/km²/yr)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Senegal</td>
<td>Langue de Barbarie MPA</td>
<td>1,310</td>
<td>404</td>
<td>900</td>
</tr>
<tr>
<td></td>
<td>Saint Louis CA</td>
<td>1,310</td>
<td>216</td>
<td>1,100</td>
</tr>
<tr>
<td>Guinea-Bissau</td>
<td>Cacheu MPA</td>
<td>1,310</td>
<td>15,740</td>
<td>0</td>
</tr>
<tr>
<td></td>
<td>Cacine CA</td>
<td>1,310</td>
<td>2,830</td>
<td>0</td>
</tr>
<tr>
<td></td>
<td>Urok MPA</td>
<td>1,310</td>
<td>10,101</td>
<td>0</td>
</tr>
<tr>
<td></td>
<td>Galinas CA</td>
<td>1,310</td>
<td>12,157</td>
<td>0</td>
</tr>
<tr>
<td>Guinea</td>
<td>Tristao MPA</td>
<td>1,310</td>
<td>21,432</td>
<td>0</td>
</tr>
<tr>
<td></td>
<td>Kanfarandé CA</td>
<td>1,310</td>
<td>13,158</td>
<td>0</td>
</tr>
<tr>
<td>Cape Verde</td>
<td>Santa Luzia MPA</td>
<td>1,310</td>
<td>50,483</td>
<td>0</td>
</tr>
<tr>
<td></td>
<td>Sao Vicente CA</td>
<td>1,310</td>
<td>34,196</td>
<td>0</td>
</tr>
</tbody>
</table>

In this case, the value of fisheries estimated in direct use values is most often above the estimated biomass production based on the references from the literature. As a result, fisheries biomass production is down to zero in most sites. The Senegalese sites are the only ones that have a value different from zero. It is therefore important to investigate further the biomass production of seagrass in West African since the current references seem to very much underestimate this service.

5.4.2.3 Mangroves

For mangroves, the very high organic matter production of trees and plants (about 1 kg/m²/yr, as compared to the 5 kg of organic matter production per m² per year of the most productive ecosystem on earth - the rainforest) greatly enrich the coastal waters and support development of rich marine life (Dabin, 1980). This high amount of
nutrients explains the important breeding and nursery role of mangrove for species that spend their adult lives in other ecosystems such as rocky bottoms, mudflats or seagrass meadows (Bann, 1998). Fish biomass of large mangrove ecosystems in Florida, Queensland, the Solomon Islands and New Caledonia are estimated to be respectively 13.26t/km², 8.20t/km², 11.60t/km² and 20.67t/km² (Thayer et al.1987, Baber et al.1989, Thollot 1992). Also Rönnback (1999) estimated the total biomass of fish associated with *Avicennia* and *Rhizophora* microhabitats to be approximately 10.4t/km², with levels generally ranging from 4 to 25t/km². I can estimate the biomass of fish in West African mangrove to average 13t/km², but this biomass includes juvenile fish (which are estimated to account for 30% according to Thollot (1992)).

As a result, the fisheries biomass of mangrove is 9.1 t/km² per year or 5,250 euros/km² with an average price of 375 CFA/kg (eq. 0.58 euros) (species are split almost equally between high valued species and low valued ones). The biomass production of mangroves only considers surfaces of *Rhizophora* and *Avicennia* as areas of mangroves; the deforested mangrove does not play any significant role in biomass production. As a result, the unitary value of unexploited fisheries biomass is to be multiplied with the ratio of areas of *Rhizophora* and *Avicennia* by the total surface of mangrove (column 3). The results are reported in Table 5-7.

<table>
<thead>
<tr>
<th>Countries</th>
<th>Site</th>
<th>Fisheries biomass production value (€/km²/yr)</th>
<th>Fisheries value in mangroves (€/km²/yr)</th>
<th>Ratio of surfaces of <em>Avicenia</em> et <em>Rhizophora</em> by total surfaces of mangroves (km²)</th>
<th>Unitary value of unexploited fisheries biomass (€/km²/yr)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Senegal</td>
<td>Langue de Barbarie MPA</td>
<td>5,250</td>
<td>541</td>
<td>0.38</td>
<td>1,700</td>
</tr>
<tr>
<td></td>
<td>Saint Louis CA</td>
<td>5,250</td>
<td>86</td>
<td>0.35</td>
<td>1,800</td>
</tr>
<tr>
<td>Guinea-Bissau</td>
<td>Cacheu MPA</td>
<td>5,250</td>
<td>568</td>
<td>0.59</td>
<td>2,800</td>
</tr>
<tr>
<td></td>
<td>Cacine CA</td>
<td>5,250</td>
<td>1,492</td>
<td>0.57</td>
<td>2,100</td>
</tr>
<tr>
<td></td>
<td>Urok MPA</td>
<td>5,250</td>
<td>1,187</td>
<td>0.77</td>
<td>3,100</td>
</tr>
<tr>
<td></td>
<td>Galinas CA</td>
<td>5,250</td>
<td>1,129</td>
<td>0.46</td>
<td>1,900</td>
</tr>
<tr>
<td>Guinea</td>
<td>Tristao MPA</td>
<td>5,250</td>
<td>4,283</td>
<td>0.47</td>
<td>500</td>
</tr>
<tr>
<td></td>
<td>Kanfarandé CA</td>
<td>5,250</td>
<td>7,607</td>
<td>0.51</td>
<td>0</td>
</tr>
</tbody>
</table>

Table 5-7: Value of fisheries biomass production of mangroves
The unitary values are comparable in most sites and range from 1,700 euros/km²/yr for Langue de Barbarie MPA to 3,100 euros/km²/yr for Urok MPA. The values in the Guinean sites are much lower: 500 euros/km²/yr for Tristao and zero for Kanfarandé. These latter results may be caused by the heavy exploitation by fisheries in mangroves of both Tristao and Kanfarandé sites.

Differences in values between MPAs and CAs are apparent in Guinea-Bissau. These values are higher in the MPAs for two reasons: a) fisheries exploitation is less important in the MPAs (especially in the case of Urok) and b) the ratio of surfaces of *Rhizophora* and *Avicennia* to the total area of mangrove is higher in MPA sites than in the CA sites, thanks to the better health status of the mangrove ecosystem.

### 5.4.2.4 Mudflats

Mudflats support a high biomass of molluscs which are exploited by subsistence fishing to a large extent. According to Rönnback (1999), the biomass of molluscs in the mudflats located in the intertidal area close to mangroves is about 70 t/km²/yr. However, these estimates only consider the most productive mudflats (for which I have not calculated the surface area). The other parts of the mudflats are likely to be much less productive. As a result, given this high value of production, the production of mollusc biomass should be further investigated before being accounted for here.

### 5.4.2.5 Coral bottoms

The average value of the fishable biomass for coral communities on bedrock in Martinique is 9,454 g for 200m² (Failler et al., 2010), which represents about 47.5 t/km². This volume of biomass production equals to 96,000 euros/km²/yr if I consider an average value of 220 CVE per kg. The equivalent values are shown in Table 5.8.
Table 5-8: Value of fisheries biomass production in coral bottoms

<table>
<thead>
<tr>
<th>Site</th>
<th>Fisheries biomass production (€/km²/yr)</th>
<th>Fisheries value in coral bottoms (€/km²/yr)</th>
<th>Unitary value of unexploited fisheries biomass (€/km²/yr)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Santa Luzia MPA</td>
<td>96,000</td>
<td>339,000</td>
<td>0</td>
</tr>
<tr>
<td>Sao Vicente CA</td>
<td>96,000</td>
<td>14,800</td>
<td>81,200</td>
</tr>
</tbody>
</table>

The results show a very high value for Sao Vicente CA (more than ten times the unitary values of other ecosystems!) and a null value for Santa Luzia. This extreme difference may be explained by very high biomass production value first, which is caused by a very high fisheries biomass production volume found in the literature (47.5 t/km²/yr), significantly higher than other ecosystems production (9 tonnes/km²/yr for mangroves, for instance). Also, the average price of fish in Cape Verde is four times higher than the average price for the continental West Africa which leads to a very high value of unexploited fisheries biomass, in the case where fisheries exploitation is limited (as is the case in Sao Vicente CA).

Second, this extreme difference is caused by an overstatement of fishing activity in coral bottoms ecosystems. Hence, coral bottoms are estimated to be important contributor to fisheries in Santa Luzia, according to fishermen. This may be the result of misinterpretation of the question asked on the frequency of fishing activity on the ecosystem. For this reason, coral bottoms are given a large proportion of the total fisheries value. This high value and the quite small surface of coral bottoms estimated in Santa Luzia have generated a huge unitary value of fisheries in Santa Luzia. This unitary value is higher than fisheries biomass production value and leads to an unexploited fisheries biomass value of zero.

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42 As detailed earlier, the question on the use of ecosystems should have included one aspect on the time period spent on the ecosystem and not only the frequency of use. One other explanation for overstating the fishing use on coral bottoms is that the respondents may have considered bottoms as most of the area seabed. Hence, corals are found on most of the rocky bottoms in Santa Luzia and the difference between coral bottoms and rocky bottoms may be difficult to differentiate. To avoid this, the questionnaire should have included a map of the considered ecosystems when asking question about uses.
5.4.3 Carbon sequestration

At present the cost of CO\(_2\) emission is estimated at less than 6 euros/t CO\(_2\) on the European market for carbon emission\(^{43}\). However, on the blue carbon market, there are almost only voluntary mechanisms existing. The price for one ton of CO\(_2\) on this market is about 17 euros/ton\(^{44}\) which is the value used in this study.

The following section describes the value of carbon absorption and storage by coral bottoms, estuaries, seagrass meadows and mangroves.

5.4.3.1 Coral bottoms

A healthy reef has a greater capacity to synthesize coral limestone and subsequently absorb more CO\(_2\). However, this only happens in the case of carbon saturation. Otherwise, in water that is under-saturated in carbon, the dissolution of calcium carbonate may occur, this releasing further carbon into the water (Failler et al., 2011). Many studies suggest that an increase in the amount of CO\(_2\) dissolved in water, along with an increase in temperature, would result in a significant decrease in calcification and the dissolution of calcareous structures (Ibid). As a result the construction of coral structures is a balance between bio-mineralization and dissolution of calcium.

In terms of coral settled on bedrocks (coral bottoms), calcareous algae of the genus *Halimeda* present on the algo-coral bottoms on the Atlantic coast have the ability to capture carbon and produce carbonates (Barry et al., 2013). However the distribution is unknown in the studied region which makes estimation of carbon capture by this plant uncertain. It is not considered here.

The coral reefs of the world cover some 617,000 km\(^2\), with the total amount of carbon absorbed estimated at 111 million tonnes of carbon per year (approximately 407 million tonnes of CO\(_2\) per year)\(^{45}\) (Lauvier, 2003). As the coral reefs of Santa Luzia MPA and Sao Vicente CA cover 0.8 km\(^2\) and 0.6 km\(^2\), this suggests a carbon sequestration capacity of 144 t (527 t CO\(_2\)) and 108 t (385 t CO\(_2\)) per year for these sites, respectively. However, the carbon capture capacity has been calculated for bio-constructed coral reefs, the most productive in terms of carbon capture. So, if I consider 25% as reflecting the proportion of the most productive coral bottoms (located around 10 m deep) within the total coral bottom cover in Cape Verde, the total carbon dioxide capture capacity equals 130t and

\(^{43}\text{Based on rate on the EEX website: http://www.eex.com/en/}\)
\(^{44}\text{Based on data collected in June 2013 on https://seagrassgrow.org/}\)
\(^{45}\text{To calculate the volume of CO}_2\text{ captured based on the volume of carbon, one has to multiply the volume of carbon by 3.66.}\)
100t per year for Santa Luzia and Sao Vicente, respectively. Values for these sequestrations are 2,210 euros and 1,700 euros per year.

### 5.4.3.2 Estuaries and channels

The undersea continental shelf and estuaries are major sinks for atmospheric carbon (Bouillon et al., 2008). Laffoley (2009) conducted a review of various marine carbon sinks which states that estuaries capture carbon at a rate of 50 tC/km²/yr. Based on this figure, it is possible to estimate carbon sequestration and the subsequent value of this by estuaries and channels. These values are shown in Table 5-9.

**Table 5-9: Carbon capture value by estuaries and channels**

<table>
<thead>
<tr>
<th>Site</th>
<th>Unitary carbon capture capacity (tC/km²/yr)</th>
<th>Surface (Km²)</th>
<th>Volume of carbon capture (t)</th>
<th>Volume of CO₂ capture (t)</th>
<th>Value of CO₂ capture (€)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Langue de Barbarie MPA</td>
<td>50</td>
<td>7.0</td>
<td>352</td>
<td>1,290</td>
<td>21,930</td>
</tr>
<tr>
<td>Saint Louis CA</td>
<td>50</td>
<td>10.6</td>
<td>530</td>
<td>1,943</td>
<td>33,031</td>
</tr>
<tr>
<td>Cacheu MPA</td>
<td>50</td>
<td>113.6</td>
<td>5,679</td>
<td>20,825</td>
<td>354,025</td>
</tr>
<tr>
<td>Cacine CA</td>
<td>50</td>
<td>108.5</td>
<td>5,425</td>
<td>19,892</td>
<td>338,164</td>
</tr>
<tr>
<td>Urok MPA</td>
<td>50</td>
<td>48.5</td>
<td>2,425</td>
<td>8,892</td>
<td>151,164</td>
</tr>
<tr>
<td>Galinas CA</td>
<td>50</td>
<td>25.5</td>
<td>1,275</td>
<td>4,675</td>
<td>79,475</td>
</tr>
<tr>
<td>Tristao MPA</td>
<td>50</td>
<td>232.8</td>
<td>11,641</td>
<td>42,682</td>
<td>725,594</td>
</tr>
<tr>
<td>Kanfarandé CA</td>
<td>50</td>
<td>76.0</td>
<td>3,802</td>
<td>13,939</td>
<td>236,963</td>
</tr>
</tbody>
</table>

The most valuable sites for carbon sequestration are Tristao MPA and Cacheu MPA in Guinea and Guinea-Bissau. This is due to the large surface area of their estuaries which are highly efficient for carbon sequestration. These values are less important though than other indirect use values (such as biomass production or coastal protection). However, they could be translated into real payments to the MPA (through blue carbon market mechanisms) and should therefore be considered as an important contributor to the creation of value in the MPA.
5.4.3.3 Seagrass meadows

Seagrass meadows, like estuaries, are major blue carbon sinks provided they are in good condition (Laffoley, 2009). If the meadows are in poor condition (that is to say they have stopped their growth or are covered with sediments) then they may release carbon as coral reefs do. In addition, physical destruction of “mattes” of seagrass (thick layers of a seagrass root system where organic matter is stored) may expose organic carbon to the washing away and so further release of huge volumes of carbon. Though references on the health status of seagrass in West Africa are unknown, observations suggest that seagrass meadows are in average to good health status and therefore do capture carbon.

Various studies estimated the sequestration rate of carbon by seagrass at an average of 129tC/km²/yr (Champenoy, 2008; Laffoley 2009; Chauvaud and Bouchon 1997; Agostini et al. 2003). According to Champenoy (2008), Posidonia oceanica (found mostly in the Mediterranean Sea but also in Mauritania) can absorb 6 mol of CO₂ per square metre per year, which represents 72tC/km²/yr. On the other hand, Kennedy and Björk (2009) state that seagrass contributes to 15% of the total capture of atmospheric carbon in marine ecosystems. They trap an average of 83tC/km²/yr, which is more important than most terrestrial ecosystems capture rates (Mateo et al, 2006.). A third approach has assessed the primary production of these organisms in order to estimate the annual rate of carbon capture (Agostini et al., 2003). Primary production is calculated by drying the seagrass plant and weighing it. This approach has enabled the calculation of the average carbon sequestration rate of seagrass at 231tC/km²/yr. These three approaches give an average rate of carbon capture of 129tC/km²/yr (264 tCO₂/km²/yr).

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46 This is one reason why seagrass meadows can lead to significant volumes of carbon release in the case of their destruction. To avoid this, and promote seagrass protection, blue carbon projects in MPA may be developed to limit the destruction of “mattes” and hence avoid such carbon releases.
Table 5-10 shows the amount of \( \text{CO}_2 \) captured per year by seagrass.
## Table 5-10: Carbon capture value by seagrass meadows

<table>
<thead>
<tr>
<th>Site</th>
<th>Unitary carbon capture capacity (tC/km²/an)</th>
<th>Surface (Km²)</th>
<th>Volume of carbon capture (t)</th>
<th>Volume of CO₂ capture (t)</th>
<th>Value of CO₂ capture (€)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Langue de Barbarie MPA</td>
<td>129</td>
<td>0.5</td>
<td>65</td>
<td>236</td>
<td>4,012</td>
</tr>
<tr>
<td>Saint Louis CA</td>
<td>129</td>
<td>0.7</td>
<td>90</td>
<td>331</td>
<td>5,627</td>
</tr>
<tr>
<td>Cacheu MPA</td>
<td>129</td>
<td>1.3</td>
<td>168</td>
<td>615</td>
<td>10,455</td>
</tr>
<tr>
<td>Cacine CA</td>
<td>129</td>
<td>6.4</td>
<td>826</td>
<td>3027</td>
<td>51,459</td>
</tr>
<tr>
<td>Urok MPA</td>
<td>129</td>
<td>2.0</td>
<td>255</td>
<td>936</td>
<td>15,912</td>
</tr>
<tr>
<td>Galinas CA</td>
<td>129</td>
<td>1.4</td>
<td>181</td>
<td>662</td>
<td>11,254</td>
</tr>
<tr>
<td>Tristao MPA</td>
<td>129</td>
<td>1.6</td>
<td>206</td>
<td>757</td>
<td>12,869</td>
</tr>
<tr>
<td>Kanfarandé CA</td>
<td>129</td>
<td>1.4</td>
<td>181</td>
<td>662</td>
<td>11,254</td>
</tr>
<tr>
<td>Santa Luzia MPA</td>
<td>129</td>
<td>1.3</td>
<td>163</td>
<td>597</td>
<td>10,149</td>
</tr>
<tr>
<td>Sao Vicente CA</td>
<td>129</td>
<td>1.5</td>
<td>200</td>
<td>731</td>
<td>12,427</td>
</tr>
</tbody>
</table>

In spite of a higher rate of carbon capture than estuaries, the results nevertheless show lower values of carbon storage by seagrass meadows per site, because of the small areas of seagrass cover. However, as indicated above, this carbon capture per year does not reflect the important carbon volume that is stored permanently in the “mattes”. Should these “mattes” be degraded, the seagrass would release very significant volumes of carbon. As a result, in principle I could have also considered the value of carbon stored in the soil. However, the methods used in my study allow me to estimate the present annual value of carbon flows rather than stocks, and does not enable me to calculate storage values. Hence these are not presented here.

### 5.4.3.4 Mangroves

Mangroves are also very important blue carbon sinks (Laffoley, 2009; Murray et al., 2011). According to Wells et al. (2006) mangrove ecosystems cover some 15.2 million hectares worldwide and absorb 25.5x10⁶ tonnes of carbon per year. This volume equals a rate of capture of 167.7tC/km²/yr. On the other hand, a publication by the IUCN reports that mangroves could absorb up to 139 tC/km²/yr (Laffoley, 2009). As a result
of these two figures, I have estimated the average amount of carbon absorbed by the mangroves at 153tC/km²/yr.

Table 5-11 shows the amount of carbon captured by the mangrove ecosystems. I consider here that *Rhizophora* dominant mangrove delivers 100% of the estimated carbon capture rate, while *Avicennia* dominant mangrove capture only 40% of that value, and deforested mangrove covered with grass only 10%.
<table>
<thead>
<tr>
<th>Site</th>
<th><em>Rhizophora</em> dominant mangrove surface (km²)</th>
<th><em>Avicennia</em> dominant mangrove surface (km²)</th>
<th>Deforested mangrove with grass (km²)</th>
<th>Unitary carbon capture of <em>Rhizophora</em> dominant mangrove (t/km²/yr)</th>
<th>Volume of carbon capture (t)</th>
<th>Volume of CO₂ capture (t)</th>
<th>Value of CO₂ capture (€)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Langue de Barbarie MPA</td>
<td>0.1</td>
<td>0.2</td>
<td>0.5</td>
<td>153</td>
<td>36</td>
<td>131</td>
<td>2,227</td>
</tr>
<tr>
<td>Saint Louis CA</td>
<td>0.0</td>
<td>0.6</td>
<td>1.1</td>
<td>153</td>
<td>54</td>
<td>196</td>
<td>3,332</td>
</tr>
<tr>
<td>Cacheu MPA</td>
<td>131.0</td>
<td>119.0</td>
<td>174.0</td>
<td>153</td>
<td>29,988</td>
<td>109,956</td>
<td>1,869,252</td>
</tr>
<tr>
<td>Cacine CA</td>
<td>61.0</td>
<td>45.0</td>
<td>65.0</td>
<td>153</td>
<td>13,082</td>
<td>47,965</td>
<td>815,405</td>
</tr>
<tr>
<td>Urok MPA</td>
<td>36.6</td>
<td>23.4</td>
<td>17.7</td>
<td>153</td>
<td>7,303</td>
<td>26,776</td>
<td>455,192</td>
</tr>
<tr>
<td>Galinas CA</td>
<td>7.8</td>
<td>12.5</td>
<td>23.5</td>
<td>153</td>
<td>2,318</td>
<td>8,499</td>
<td>144,483</td>
</tr>
<tr>
<td>Tristao MPA</td>
<td>27.7</td>
<td>35.5</td>
<td>71.4</td>
<td>153</td>
<td>7,497</td>
<td>27,489</td>
<td>467,313</td>
</tr>
<tr>
<td>Kanfarandé CA</td>
<td>3.3</td>
<td>31.9</td>
<td>34.3</td>
<td>153</td>
<td>2,989</td>
<td>10,958</td>
<td>186,286</td>
</tr>
</tbody>
</table>
The value of carbon capture in mangroves is very important for sites with a high proportion of Rhizophora (as in Guinea and Guinea-Bissau). These values are therefore dependent on the health status of the mangrove. This, coupled with high carbon capture rates (as for seagrass meadows) has led to very significant values of carbon captured in mangroves. Like seagrass, mangroves also permanently stores high volumes of carbon in the soil that are not accounted for in this study.

5.4.4 Water and waste treatment

Marine ecosystems are very sensitive to the quality of coastal waters, though they contribute significantly to their purification. The role of coastal ecosystems in water and waste treatment is triple (MEA, 2005):

- They act as filters of coastal waters, fix the sediments that are transported in the water and hence reduce the turbidity of waters. This reduces the deposition of sediments on coral bottoms and seagrass meadows (which require clear water for their development). Through their stilt roots, mangroves act as filters of inland waters, limiting the amount of suspended matter discharged into the oceans.
- They ensure the regeneration of water through tidal recirculation, thus enabling treatment by micro-organisms contained in the water; and
- They act as buffers to terrestrial chemical pollution.

5.4.4.1 Estuaries and channels

Estuaries and channels play a dual role in the movement of nutrients and mitigation of risks related to chemical pollution (MEA, 2005). The most significant risk to coastal waters in West Africa is nutrient overload which can cause eutrophication in quiet bays or still seas. This eutrophication leads to a loss of water quality and sometimes the development of "dead zones" (anoxic areas) that threaten the economic activities of the area (especially fishing).

Furthermore, nutrient cycling is very important because it is a support service that sustains all other ecological functions of the ecosystems (in this regard, this service could be considered as different from water and waste treatment). The only reference that could be found to this service is that of Costanza who used the replacement cost method (Costanza et al. 1997). They found that the value for this cycling was 1.47 million euros/km²/yr because it was considered as a supporting service for all other services, ecological functions (and economic activities). However, given that nutrient cycling is different from the water treatment service, it did not seem appropriate to
take this value into account here as, given its importance (over 340 million euros for Tristao), it would distort any value comparison between sites. In addition, I have concerns with using the replacement costs method as this service was not reproducible artificially (Beaumont et al., 2005). Despite this, it is important to highlight these aspects and take this service into account while considering the protection of estuaries and channels.

5.4.4.2 Mangroves
In addition to the retention of suspended particles, mangroves also have the ability to purify the waters of some of the nutrients by absorbing nitrates and phosphates and certain pollutants such as heavy metals or toxic substances (Wells et al., 2006). The existence of mangroves and their treatment capacity has an important monetary value. A study conducted in Fiji using replacement cost method (through the establishment of one water treatment station) estimated the value of this treatment at 174,200 euros/km²/yr (Lal, 2003). The transfer of value using the GDP per capita index (from Fiji to West African countries) leads to values of: 51,220 euros/km²/yr for Senegal, 19,187 euros/km²/yr for Guinea-Bissau, and 35,203 euros/km²/yr for Guinea.

In this case, I considered the total surface area of mangrove as a full provider of water and waste treatment service. Of course, the Rhizophora dominant mangrove provides water treatment throughout the year, but the other layers may also contribute to water treatment by retaining sediments during the rainy season or spring high tides. However, as this approximation is not backed by any ecological study, research is required to examine the treatment capacity of Avicennia dominant mangroves and deforested mangrove areas subsequently covered with grass.

5.4.4.3 Seagrass meadows
Seagrass meadows are also involved in water treatment since seagrass traps part of the suspended particles in its root complex (the "matte"). In addition to this role in stabilizing the substrate, seagrass also purifies the water while using minerals for its growth. Costanza et al. (1997) estimates the value of this service to be 1.27 million euros/km²/yr for a total area of 2,000,000 km² with 177,000 km² being seagrass (Waycott et al., 2009). Given that the contribution to water treatment is proportional to the seagrass area, I assume that the average value of water treatment of seagrass at a global scale is 112,672 euros/km²/yr. Weighted by the GDP per capita index in comparison to average global GDP, the following unitary values were obtained: 17,801
186 euros/km²/yr for Senegal, 6,668 euros/km²/yr for Guinea-Bissau, 12,235 euros/km²/yr for Guinea and 38,115 euros/km²/yr for Cape Verde.

The presence of algae probably also affect the water quality by absorbing a substantial portion of nutrients, up to a certain point. However, the algal ecosystem is not included in this study and the treatment service value is hence not considered here (but worth noting).

### 5.4.4.4 Coral bottoms

Coral bottom ecosystems also make a contribution to water treatment. Their symbiosis with *zooxanthellae*, using CO₂, nitrogen and phosphorus in water to produce organic matter, purifies the water of some of these nutrients. If I consider that the bio-constructed coral reefs and coral bottoms on bedrocks provide the same water purification rate, the replacement cost of the purification service to that produced by the coral reef ecosystems is 3,886 euros/km²/yr according to Costanza et al. (1997). This results in a value of 1,317 euros/km²/yr after being weighted by the GDP per capita of Cape Verde.

### 5.4.5 Indirect use values: key results

The indirect use value (IUV) of the five MPAs of the studied sample represents a total of 28.5 million euros. The highest values are those of the Cacheu MPA (15.7 million euros) and Tristao MPA (8.9 million euros), followed by the Urok MPA (3.5 million euros). The most valuable ecosystem with regards to indirect use values is mangrove, with 24.5 million euros or nearly 86% of the IUV. The second most valuable ecosystem is estuaries and channels, with 3.54 million euros or 12% of the IUV. The results are presented in Figure 5.1.
The IUV for CAs are reported in Figure 5-1 as well. As for the direct use values, the CA values are calculated based on the unitary values of ecosystems multiplied by the reference surface area of ecosystem in the MPA. This comparison shows an excess of IUVs of 2.9 million euros for MPAs. The IUVs in MPAs are superior to those of the CAs, except for the Santa Luzia MPA. These higher values for MPAs reflect the better overall ecosystem health and reduced exploitation of natural resources.

5.5 Discussion
The influence of the ecosystem health status on the delivery of indirect uses, as explored in my study, should be furthered. Such investigation on the linkages between health status and the delivery of indirect uses has never been undertaken in West Africa to my knowledge. Globally, the linkages between ecosystem health and the delivery of human well-being has been scrutinized in 1992 by authors including Costanza (Costanza et al., 1992; Haskell et al., 1992; Norton, 1992) and later translated into the impact of ecosystem health on the delivery of ecosystem services (with the
notable example of the UK National Ecosystem Assessment). However, it is not possible to find a valuation exercise that aimed to transcribe the various statuses of ecosystem health into various ecosystem values. Hence, most valuation studies establish any differences within the same ecosystem, be it totally degraded or in a pristine state. I found this approximation really far from reality and, as a result, have proposed a way to integrate the variations of health status for one ecosystem within the value calculation. This has enabled me to translate the variations of health status of ecosystems into variations of value. However, the method proposed could certainly be improved. The proposed table for translation of health status to percentage of delivery (Table 2-5) is very approximate and not based on local scientific surveys. Also, it is the only indirect use that is weighted according to the health status. All indirect uses and ecosystems should combine this method but this should be backed by scientific research on the ecological functions at stake.

Furthermore, the non-consideration of health status in indirect use valuation could create even more biases in cases of benefit transfers. Thus, most for value transfers only the service and the ecosystem are considered, along with the socioeconomic context of origin, before transferring the value. They never consider the health status for the ecosystem from which they transfer the value. In this research as well, I have not taken this aspect into consideration. This aspect should be further explored to fine-tune the benefit transfer method.
6 Chapter 6: non-use values

6.1 Introduction
The aim of this chapter is to estimate the third main component of the TEV, namely the non-use values of MPAs, and compare them to those of CAs. This chapter first presents the non-use uses considered in the MPAs and CAs and provides further details on their magnitude. The chapter then gives an overview of the specific calculation methods used in the specific context of the case study. It also discusses the results found.

6.2 Description of non-use values
As detailed in chapter 2, non-use values are related to the satisfaction of knowing that a species or ecosystem exists (existence value) or knowing that future generations (bequest value) or other people (altruist value) will have access to such benefits provided these benefits are managed in a sustainable way (Krutilla, 1967). Non-use values therefore relate to current and future values.

Existence and bequest value in West Africa have a great importance because of the traditions associated with the ecosystems for local populations, and the willingness of these populations to see their children pursue their traditions after them (see Figure below). Everywhere in Africa, nature is a symbol, divided into three categories: plants, animals and minerals (Dakouri, 2001). Nature is also a genitor and the place where Gods live (Ibid). These beliefs have led to divinatory practices of local populations with regard to nature (Fall, 1999), a certain syncretism according to the fact that the population in the study areas is broadly either Muslim (to a larger extent) or Christian. The cultural traditions of West African associated to nature materializes in sacred sites. These natural scared sites are linked to a system of beliefs and a specific system of resources management (Duchesne, 2002). There are many examples of sacred sites that have been preserved from exploitation. This is the case in the forests of Fouta Diallon in Guinea (Diallo and Diallo, 1999). In some of these sites, nobody is permitted to enter and all uses are prohibited. Hence, the values born by cultural and religious traditions, and especially in sacred sites, cannot be estimated by direct use values, but rather by non-use valuation.
Figure 6-1: Religious zoomorphs masks in the Bijagos Archipelago (credit: T. Caroff)

6.3 Method applied

Non-use values of marine and coastal ecosystems have almost exclusively been estimated through contingent valuation method (Ayob et al., 2001; Hargreauen-Allen et al., 2004; Hundloe et al, 1987; Pham and Tran, 2003; Spash, 2000; Spurgeon et al., 2004; Subade, 2005). Bishop et al. (2011) has used the multi-attribute method as an alternative method. Also, Van Beukering et al., (2007) and Taylor (2011) are the only two examples found in the literature that aimed to estimate non-use values through choice experiment method. In our case, and according to the limits of contingent valuation method, I have used the choice experiment method to estimate non-use values of MCEs.

The determination of non-use valuation through choice experiments follows a complex process which has been described by Hanley et al. (2001). That process is adapted here to the specificities of my study.

6.3.1 Define the attributes and their related levels

This first step requires a strong qualitative study, which draws on observations and interviews carried out during one mission in the field. Organisation of focus groups
with all users and non-users of the MCEs, as well as face-to-face interviews with experts on MCEs are required. This will enable the completion of the following four activities that are described below.

6.3.2 Identify target populations
The target population for this valuation are the individuals living (either permanently or temporarily) in the studied site. These individuals have the greater non-use value for these ecosystems. While it is difficult to account for non-use value among people who may use the ecosystems I am considering (Spurgeon et al., 2004), it is nevertheless necessary to capture these values if possible.

Within the local population, I therefore determine the groups of people whose preferences for non-use value of MCEs are relatively homogeneous. In general, residents and tourists are separated. In some of my sites, however, the number of tourists is insignificant when compared to residents, and so will be discounted.

6.3.3 Understand the social and cultural representation associated with MCEs
It is important to identify and understand the social and cultural representation of MCEs within the population. David et al. (2007) emphasize particular images of virginity, purity, abundance or fertility. In my case, it is necessary to highlight the reasons for attachment to MCEs, the typical features of the site, the perception of the status of ecosystems, the expectations of protection, conditions of membership management measures and protection, amongst others.

6.3.4 Attributes describe the method of choice experiments
The challenge here is to identify attributes truly independent of their use in MCEs. This ensures that respondents will vote only on the values of non-use, despite their level of experience or interaction with the MCEs. This can be tested by using pictures of the beauty, diversity or richness of MCEs. Another difficulty is to identify which attributes apply to both residents and visitors.

The monetary attribute must represent the monetary contribution required for the specific implementation of a programme of specific conservation and management. I note here that tax is among the best means to introduce this attribute. Indeed, if the voluntary contribution is usually the preferred solution for a survey on non-use value (through contingent valuation), a compulsory payment associated with a particular scenario may represent a more tangible value for the respondent. However, the tax details must be framed adequately to be perceived as being to the advantage of the surveyed population. Otherwise, the survey might not be representative (over or under
representation of the choice of “status quo” attribute levels). Tax for residents could take the form of a property tax, with a daily holiday levy for visitors, as most studies have done before (Dachary-Bernard, 2004). However, there may be sites where the local economy is non-monetised and based on in-kind exchanges. In this case, the surveyor will have to provide a framework within which such a tax could be implemented. The understanding of the framework will be tested during the preliminary field mission.

6.3.5 Define the attribute levels
After highlighting attributes, strictly speaking, the next step is to identify socially acceptable attribute levels for the populations surveyed. It is essential at this stage to refer to the health of MCEs of the community in question, the pressures that threaten it, and the management measures and conservation programmes currently in place. These factors should be taken into account as they define the status quo for individuals (here, the future of each of the non-monetary attributes associated with MCEs without any intervention). It enables us to check that the perceptions of individuals are consistent with reality and ensures the credibility of the status quo scenario.

Attribute levels must represent the impact of the various measures implemented as part of the scenario considered. They may be of differential importance, depending on the number of attributes, but they should always be compared to the “status quo”. In practice, the number of attribute levels is limited (as is the number of attributes) by the maximum number that the surveyed population can envisage (generally, no more than 2 levels of difference from the “status quo”).

6.3.6 Elaboration of the experimental plan
The experimental plan aims to develop combinations of attributes that form the scenarios to be presented during the investigation. In the simplest case where the number of attributes and levels is sufficiently small, it is possible to generate a complete process, for which all possible combinations are assembled and presented to respondents. Most of the time however, it is necessary to select scenarios to present a partial analysis. This process must meet three criteria:

- Orthogonality: the effects of all attributes must be orthogonal (or independent of the utility function);
- The equilibrium of level representation: each level attribute should appear the same number of times;
- The equilibrium value: experiences generated must minimize the variance-covariance parameters and estimated choice model.
This latter step requires the use of SAS software (or other similar statistical software) and is based on the method described by Zwerina et al. (1996) for minimizing the error-D.

6.4 Results

6.4.1 Improvement of ecosystems and their related functions

The purpose of the choice experiment method applied here is to obtain monetary indicators for various scenarios concerning the use and maintenance of MCEs over time. The non-use value is deducted from the scenario that favours the ecosystem conservation most (the "greenest" scenario).

As detailed earlier, the investigation consisted of face-to-face interviews with local populations (both residents and allochton populations) and tourists (Figure 6-2).
<table>
<thead>
<tr>
<th>1</th>
<th>Terrestrial activities: increase 20%</th>
<th>Regulations: on industrial maritime activities only</th>
<th>Biodiversity: increase 20%</th>
<th>Cost</th>
</tr>
</thead>
<tbody>
<tr>
<td>2</td>
<td>Terrestrial activities: status quo</td>
<td>Regulations: on all maritime activities</td>
<td>Biodiversity: decrease 20%</td>
<td>Cost</td>
</tr>
<tr>
<td>3</td>
<td>Terrestrial activities: status quo</td>
<td>No regulation on maritime activities</td>
<td>Biodiversity: increase 20%</td>
<td>Cost</td>
</tr>
<tr>
<td>4</td>
<td>Terrestrial activities: status quo</td>
<td>Regulations: on all maritime activities</td>
<td>Biodiversity: status quo</td>
<td>Cost</td>
</tr>
<tr>
<td>5</td>
<td>Terrestrial activities: status quo</td>
<td>Regulations: on industrial maritime activities only</td>
<td>Biodiversity: decrease 20%</td>
<td>Cost</td>
</tr>
<tr>
<td>6</td>
<td>Terrestrial activities: decrease 20%</td>
<td>Regulations: on all maritime activities</td>
<td>Biodiversity: increase 20%</td>
<td>Cost</td>
</tr>
<tr>
<td>7</td>
<td>Terrestrial activities: decrease 20%</td>
<td>No regulation on maritime activities</td>
<td>Biodiversity: decrease 20%</td>
<td>Cost</td>
</tr>
<tr>
<td>8</td>
<td>Terrestrial activities: increase 20%</td>
<td>No regulation on maritime activities</td>
<td>Biodiversity: status quo</td>
<td>Cost</td>
</tr>
<tr>
<td>9</td>
<td>Terrestrial activities: decrease 20%</td>
<td>Regulations: on industrial maritime activities only</td>
<td>Biodiversity: status quo</td>
<td>Cost</td>
</tr>
</tbody>
</table>

Figure 6-3: Details about the nine proposed scenarios of the portfolio
After reading each of the nine scenarios, the respondent was required to pick one, according to his/her priorities (terrestrial activity, regulation of marine activities, level of biodiversity and cost incurred). From the responses, a multiple linear regression with three variables was applied with the "status quo" scenario as a reference for non-monetary attributes (regulation of land and marine activities and level of biodiversity).

This regression allows me to express the willingness-to-pay (WTP) of individuals with different levels of attributes, as follows:

$$WTP = A + \alpha_1 S_1 + \alpha_2 S_2 + \beta_1 B_1 + \beta_2 B_2 + \mu_1 C_1 + \mu_2 C_2$$

Eqn 6-1: Willingness-to-pay of individuals for three various attributes

With:

- $A =$ constant
- $S_1 =$ decrease by 20% in terrestrial activities
- $S_2 =$ increase of 20% in terrestrial activities
- $B_1 =$ ban on both commercial and subsistence activities
- $B_2 =$ ban on commercial activities only
- $C_1 =$ decrease of biodiversity level by 20%
- $C_2 =$ increase of biodiversity level by 20%
Based on statistical treatment, the following equations were found:

<table>
<thead>
<tr>
<th>Site</th>
<th>WTP equation</th>
</tr>
</thead>
<tbody>
<tr>
<td>Cacine CA</td>
<td>$WTP = 2098 + 3236.S_1 - 1416.S_2 + 42.B_1 + 879.B_2 + 821.C_1 - 529.C_2$</td>
</tr>
</tbody>
</table>

Eqn 6-2: Equations of WTP for all studied sites

A positive constant $A$ means that people have a positive willingness to pay to keep the current situation (status quo) (if the constant $A$ is negative, it means that people are not willing to pay to keep the situation as it is today). However, this latter situation was not encountered in any of the studied sites. This means that the population surveyed is generally contented with the current situation, although it is not possible to conclude as to the specific reason why. Also, the WTP for the “status quo” is greater in MPAs than it is in the CAs, except for Guinea.

To further analyse the perceptions of respondents, it is necessary to look at the coefficients of the variables. The coefficients reflect the willingness of respondents to pay for the action to happen. For example, when the coefficient of $S_1$ is positive, it means that the respondents are willing to pay for a decrease in land-based activities that are detrimental to MCE (and therefore contribute to a better state of the MCE). The results of the survey show that in most cases the coefficient $S_1$ is positive while the $S_2$
Coefficient is mostly negative, meaning that respondents favour a reduction in land-based activities that have an impact on marine ecosystems.

The regulation of marine activities also has a positive influence on willingness to pay (WTP): respondents are willing to see the marine activities that impact upon MCE controlled in 6 sites out of 10, demonstrating a determination for a stronger limitation on economic activities in the marine and coastal ecosystems.

For the third attribute, the level of biodiversity, the coefficients obtained are mostly negative regardless of the option chosen (decreased or increased level biodiversity). Only in the case of the Urok MPA did I observe a positive WTP for increasing biodiversity and a negative WTP for a reduction in biodiversity, which translates into a preference among respondents to see an increased level of biodiversity. The first interpretation of this result is that this biodiversity attribute is secondary in choice of preferred scenario compared to other attributes such as limitations on marine and terrestrial activities.

Another interpretation of this result may stem from a lack of understanding of the word “biodiversity” by respondents, leading them to disregard this attribute in their choice of scenario. This was noted as a potential limitation to the exercise during the field mission: the training session for surveyors showed a general misunderstanding by surveyors of the concept of biodiversity, even if most surveyors were students at the national university. I had several long discussions during the surveys about how to translate the word biodiversity: should it be translated as “life in the sea”? or “Plants and animals that live in the sea”? Biodiversity was presented to surveyors as the marine life that bears specific meaning to cultural and religious practices (e.g. the shellfish that are used for ceremonies in the Bijagos) and to aesthetic practices for artists, poets and storytellers (based on the diversity of colours, sizes and shapes). It was very difficult not to include use values in this description which would have caused one major bias in the non-use value estimation. For instance, if biodiversity was associated with the diversity of catches for fishers or the availability of various fish for subsistence, the results of this exercise could not have been quantified as non-use value.

6.4.2 Willingness-to-pay for conservation

Applying these equations to all scenarios (for each scenario, the variable are affected a coefficient 0 or 1 depending on its realization), I obtain the willingness of individuals to pay for each of the scenarios. The following sections detail the WTP for each MPA site.
and their comparison areas. The results are followed by hypotheses to explain these observed results. Unfortunately, it was not possible to find references for these sites which provided elements of explanations for such results of WTP between a protected and a neighbouring unprotected site.

Table 6-1: WTP of scenarios in the Langue de Barbarie MPA and its CA

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<th>Scenario</th>
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<th>B1</th>
<th>B2</th>
<th>C1</th>
<th>C2</th>
<th>WTP for the MPA (¥)</th>
<th>Rank</th>
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**Green**: best scenario for ecosystem conservation

**Violet**: scenario which gained most WTP (if different from the green scenario)

For the Langue de Barbarie MPA, the WTP for the greenest scenario is 6.7 euros and 7.1 euros for the comparison area. For the MPA, this generates values of 24,590 euros for the resident population and 5,360 euros for the tourists. For the CA, the values found are 22,025 euros and 1,420 euros for residents and tourists, respectively. Hence, the WTP in the CA is more important than in the MPA. This may be explained by several reasons; including:

- The Langue de Barbarie MPA is very much a “paper MPA” where management measures are not very different from outside the area; the MPA is used for tourist purposes with the organisation of boat tours but the MPA does not support any conservation measures except this ecotourism activity; this lack of enforcement may have caused the population to lose their trust in the capacity

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47The total value is calculated by multiplying the marginal WTP by the total number of population.
to conserve marine ecosystems. Accordingly, the WTP is less important as regards activities in the MPA; and

- The south Saint Louis CA has experienced a major change in its geomorphological structure in 2003 with the opening of the breach in the sandbar between the Senegal River and the sea. This has caused major changes in the ecological balance of the estuary and disturbed the activities that depended on this balance (local subsistence fisheries, farming). As a result the population in the CA have realized the fragility of ecosystems balance and the importance of seeing these marine ecosystems protected, which could explain the higher WTP in the CA than in the MPA.

### Table 6-2: WTP of scenarios in the Cacheu MPA and its CA

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**Green**: best scenario for ecosystem conservation  
**Violet**: scenario which gained most WTP (if different from the green scenario)

For the Cacheu MPA, the WTP for the greenest scenario is 7.1 euros for the Cacheu MPA and 7.3 euros for the Cacine CA. In both sites, the greenest scenario was the most popular one. The estimated non-use values for the MPA are 118,000 euros and 96,300 euros for the CA. Three reasons were identified during field surveys to explain these results:
- Many residents of Rio Cacheu and especially the cities of Cacheu and Sao Domingo (the two main cities) consider the mangrove ecosystem in Cacheu as not subject to overexploitation or degradation. The estuary and mangrove are very much seen as inexhaustible sources of resources and the inhabitants do not see why they should pay to see these ecosystem conserved while they are still untapped;

- The inhabitants of remote villages in the area are more aware of the fragility of the mangrove ecosystem as (in some villages) they have already experienced land salinization and a lack of drinking water. These people are willing to contribute to the conservation of ecosystems and marine biodiversity, but they are in a minority compared to the populations of the largest cities of the MPA.

- Cacine has, until recently, been a hosting site for thousands of migrant fishermen from across the region (Senegal, Guinea and Sierra Leone). Rio Cacine has seen its fish stocks overexploited in the estuary, and its mangroves cut for the smoking of large volumes of small pelagics. As the migrants have now moved further south to Tristao MPA, the population in Cacine has acknowledged the need to conserve its marine ecosystems. This awareness of people in the CA can explain why the WTP results for ecosystem protection are higher in the CA than in the MPA.

Table 6-3: WTP of scenarios in the Urok MPA and its CA

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*Green*: best scenario for ecosystem conservation

*Violet*: scenario which gained most WTP (if different from the green scenario)
For the Urok MPA, the WTP for the greenest scenario is 7.1 euros and 7.5 euros for the CA. This is a minor difference in marginal WTP and, when comparing the WTP for other scenarios, I noticed that in the MPA the greenest scenario gained most interest, while this was not the case in the Galinas CA. Based on the total site population figures, the total WTP represents values for the MPA of 16,500 euros and 9,000 euros for the CA.

Table 6-4: WTP of scenarios in the Tristao MPA and its CA

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<th>Scenario</th>
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**Green**: best scenario for ecosystem conservation

**Violet**: scenario which gained most WTP (if different from the green scenario)

For the Tristao MPA, the WTP for the greenest scenario is 6.5 euros and 6.9 euros for the comparison area. This represents values for the MPA of 58,500 euros and 103,500 euros for the comparison area. The reason that may explain why WTP in Tristao is similar to that in the case of the Langue de Barbarie MPA is that the Tristao MPA is still under development and no management plan has been enforced yet. In addition, as the migrant fishing camp in Katchek heavily exploits the marine resource of the MPA, the local population is not ready or able to pay for this MPA to protect the marine ecosystems (as it has not proved to be effective in protecting coastal ecosystems from intense exploitation and the subsequent cutting of wood for smoking fish).
Table 6-5: WTP of scenarios in the Santa Luzia MPA and its CA

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<tr>
<th>Scenario</th>
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<th>B2</th>
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**Green**: best scenario for ecosystem conservation  
**Violet**: scenario which gained most WTP (if different from the green scenario)

For the Santa Luzia MPA, the WTP for the greenest scenario is 26.3 euros and 32 euros for the comparison area. This represents non-use values for the MPA of 26,300 euros and 49,600 euros for the CA. In this case, the WTP to conserve the marine ecosystems is again lower in the MPA than in the CA. Based on the feedback from respondents, this may also be caused by a loss of trust in the national authorities given their failure to enforce the management plan of the MPA and effectively protect the marine ecosystems in Santa Luzia.

6.4.3 Non-use values: key results

Some general results may be identified from the tables presented above. First, the marginal WTP for the greenest scenario calculated for the MPA is always below the WTP of the same scenario for the unprotected site comparison area. This trend reflects a greater willingness of the resident population (and tourists when present) to see MCEs protected more in the CA than in the MPA. The first explanation for this greater WTP, as respondents in the CA stated, is that they are inclined to develop one MPA in their area and are ready to support this (as is shown by a substantial WTP). Conversely, the population in the MPA (as they stated) already benefit from the MPA and do not see why they should pay for the protection of ecosystems when there are international NGOs and donors who are willing to pay for it.
This higher WTP in an unprotected site has already been observed in Emily et al. (2013), which investigated WTP for improved water services due to spring protection in Emuhaya District (Kenya). They observed a higher WTP in unprotected springs mainly because the water situation in their springs was worse and they had not paid anything before.

A second interesting result is that the greenest scenario is always among the three to be picked by respondents, although it includes the highest level of cost incurred. For 3 sites it is the first picked, for 5 sites the second picked and for 2 sites the third picked. This illustrates the importance given to the protection of ecosystems by the populations surveyed.

The non-use values of MPAs amount to nearly 250,000 euros. The most important value is for the Cacheu MPA (118,000 euros), followed by Tristao MPA (58,000 euros). The ecosystem that contributes most to the creation of non-use values is the mangrove ecosystem (109,000 euros) and then estuaries and channels (65,000 euros). Values are presented in Figure 6-4.
Figure 6-4: Breakdown of non-use values by ecosystem (euros)

The non-use values of the CAs amount to 491,000 euros, or about double the value in MPAs. The difference in non-use values between MPAs and CAs shows a surplus for CAs of 242,000 euros. The most important non-use values are found in Kanfarandé (223,000 euros) and Cacine (180,000 euros). The most valuable ecosystems are mangroves (195,000 euros) and estuaries and channels (147,000 euros).

6.5 Discussion

6.5.1.1 Improvement of the non-use valuation
The estimation of non-use values is a very difficult task that few researchers have addressed so far. It also remains a controversial task stemming from the question of the commensurability of non-use values (Martinez-Allier et al. 1998; Carson et al., 2001). This study has provided a first estimation of these non-use values for marine
ecosystems in West Africa, but there are a lot of improvements that are required to the method used and, more importantly, to the description of non-use values by attributes.

The choice of attributes is key to the estimation of non-use values. It relies greatly on a good understanding by the evaluator about the context and perception of the ecosystems by population (see section 6.3.3). This can be developed through extensive focus groups, face-to-face interviews, as well as discussions with local and national experts. The main difficulty here is to describe the existence and bequest values in precise terms, and then to try and find the best describers of these values that can be presented visually.

In my study, I have not had enough time to carry out extensive field missions prior to the survey for questionnaire filling-in. The focus groups and interviews were probably given insufficient attention. Yet, respondents were still able to describe non-use values through diversity of species for the most part. The species included those with religious or cultural meanings. The main difficulty during these discussions rested on the distinction between the uses and non-uses of the ecosystems: they seem concomitant and valuing them separately appears impossible to most of them. This may have created biases in the valuation of non-uses which also accounted for use values necessarily associated with the choice made by the respondents. To avoid such issues, further research should be carried out in the study sites in order to clearly define the attributes that best characterise non-use value.

6.5.1.2 Choice experiment method
The method proposed for these values consisted in estimating the willingness-to-pay for the "greenest" scenario. These values could be valued differently, by estimating the utility function and the marginal value for each of the non-monetary attributes (that represent the non-use value). This was developed in a case study in the marine reserve of Prêcheur (Martinique) (Binet et al., 2013b; Binet et al., 2014).

This model can be run with Limdep© statistical software. The various attributes (both non-monetary attributes and the cost attributes) are entered into the model, as well as the individual variables that could influence the behaviour of respondents (e.g. age, income category, job, distance from the coast, or dependency of their job on the environment). A utility function can be calculated that is composed of the attributes and the individual variables (a combination of one variable – e.g. age, income - and one level of realisation of this variable – e.g. from 18 to 30 y.o.). Then, the model enables the
coefficients of each of the variables to be calculated in order to define the indirect utility function for each population.

From the indirect utility function, it is then possible to derive the marginal value for each of the non-monetary attributes (the willingness-to-pay to see the attribute improved). The non-use value is the value associated with the simultaneous improvement of all non-monetary attributes. This is calculated by the difference between the utility of the improvement of all non-monetary attributes and the utility of the initial situation ('status quo') divided by the marginal utility of the cost attribute. The value is then considered for the total population considered.

An illustration of the calculation of non-use value for the ex ante assessment of the marine reserve of Prêcheur in Martinique is presented in Appendix 7. It was done using the multinomial logit model. This proposed method seems much more adequate for measuring the non-use values.
Chapter 7: Total Economic Values and conservation benefits

7.1 Introduction
This chapter aims to present the method and results of the TEV calculation in MPAs and their related CAs, as well as the results of estimated conservation benefits. The chapter also discusses two key aspects of the use of TEV and the subsequent calculation of conservation benefits: the importance of socio-cultural context and the issue of time and space in the valuation.

7.2 Calculation method adopted
The main approach for the calculation of conservation benefits in West African MPAs is presented in chapter 3. Importantly, the total values per site were calculated based on the reference surface areas of the MPAs. Equally, for each CA, the total value of use is calculated based on the same ecosystem surface areas of the MPA to which they are being compared.

7.3 Results
The economic valuation of ecosystems for the five MPA sites and their corresponding CA sites amounts to a total of 53.2 million euros for 2,087 km², with an average value of coastal ecosystems of 25,470 euros per km² per yr. The Total Economic Value (TEV) of the sample of 5 MPAs is estimated annually at around 35.4 million euros. This value is lower than a reference total value of 3.5 billion euros per year for aquatic ecosystems and about the same value for terrestrial ecosystems in total Guinea-Bissau, which is more than 36,000 km² (Martinez et al., 2007), moreover, is a minimum value because it was constrained to values that it was possible to express in monetary terms.\(^4\)

The TEV is broken down between indirect use values (associated with the ecological functions of MCEs) accounting for 81% of the TEV, the direct use value (19% of the TEV) and the non-use value (1% of the TEV). The indirect use values hence contribute most significantly to the creation of value in the MPA. Among the direct use values, the extractive uses (fishing, logging, and salt production) represent up to 96% of the total, while non-extractive activities (beach tourism and excursions) represent only 4% (275,000 euros per year).

\(^4\)The part of the value that was not calculated here corresponds to values that could not be monetized, due either to their nature (as for option values which need a valuation approach that was not possible here) and or to the complexity of method required for their calculation.
When comparing sites on the basis of the MPA reference surface area, the difference of TEV between MPAs and CAs displays a surplus of 265,000 euros for the MPAs. The indirect use values of MPA generate a surplus of 2.9 million euros for MPAs. This surplus can be explained by the better health status of ecosystems. However, the direct use values are 2.4 million euros lower in MPAs than in CAs. The non-use values are very low compared to use values, although they are significantly higher in CAs (0.2 million euros). The value differences between MPAs and CAs are shown in Figure 7-1.

Among the estimated values, water and waste treatment is the most important value representing 41% of the TEV in MPAs (14.5 million euros), followed by coastal protection with 17.4% (6.1 million euros). Commercial fisheries represent 4.8 million euros and 13.5% of TEV. Interestingly, commercial fisheries are the only direct use among the five largest contributors to the TEV in MPAs.
Table 7-1: Values ranking by uses

<table>
<thead>
<tr>
<th>No</th>
<th>Category</th>
<th>Details</th>
<th>Value in MPAs, all ecosystems included (euros)</th>
<th>TEV percentage</th>
<th>Cumulative percentage</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>Indirect use</td>
<td>Water and waste treatment</td>
<td>14,504,072</td>
<td>41.0</td>
<td>41.0</td>
</tr>
<tr>
<td>2</td>
<td>Indirect use</td>
<td>Coastal protection</td>
<td>6,158,310</td>
<td>17.4</td>
<td>58.5</td>
</tr>
<tr>
<td>3</td>
<td>Extractive direct use</td>
<td>Commercial fisheries</td>
<td>4,762,523</td>
<td>13.5</td>
<td>71.9</td>
</tr>
<tr>
<td>4</td>
<td>Indirect use</td>
<td>Carbon sequestration</td>
<td>4,027,313</td>
<td>11.4</td>
<td>83.2</td>
</tr>
<tr>
<td>5</td>
<td>Indirect use</td>
<td>Fisheries biomass production</td>
<td>3,795,179</td>
<td>10.7</td>
<td>94.0</td>
</tr>
<tr>
<td>6</td>
<td>Extractive direct use</td>
<td>Subsistence fisheries</td>
<td>966,700</td>
<td>2.7</td>
<td>96.7</td>
</tr>
<tr>
<td>7</td>
<td>Extractive direct use</td>
<td>Mangrove wood-cutting</td>
<td>464,500</td>
<td>1.3</td>
<td>98.0</td>
</tr>
<tr>
<td>8</td>
<td>Non-use</td>
<td>Improvement of ecosystems and their related functions</td>
<td>249,250</td>
<td>0.7</td>
<td>98.7</td>
</tr>
<tr>
<td>9</td>
<td>Non-extractive direct use</td>
<td>Ecotourism and observation of fauna and flora</td>
<td>183,300</td>
<td>0.5</td>
<td>99.2</td>
</tr>
<tr>
<td>1</td>
<td>Extractive direct use</td>
<td>Sport fisheries</td>
<td>157,500</td>
<td>0.4</td>
<td>99.7</td>
</tr>
<tr>
<td>1</td>
<td>Non-extractive direct use</td>
<td>Coastal tourism, diving and nautical activities</td>
<td>92,300</td>
<td>0.3</td>
<td>100.0</td>
</tr>
<tr>
<td>1</td>
<td>Extractive direct use</td>
<td>Salt production</td>
<td>17,600</td>
<td>0.0</td>
<td>100.0</td>
</tr>
</tbody>
</table>

In terms of their contribution to the formation of TEV, mangroves offer the most important benefit with 26 million euros (or nearly 74% of the TEV of MPAs), while estuaries and channels represent 4.8 million euros (13.6%) and mudflats amount to 2.3 million euros (6.4%). Figure 7-2 shows the distribution of TEV in MPAs by ecosystem.
Looking at the economic values of ecosystems per unit area, I find that the seagrass meadows have the largest unitary value with 92,000 and 90,000 euros per km² per year respectively for MPAs and CAs. The coral bottoms of the Santa Luzia MPA have the second largest value with 84,000 euros per km² per year, a value equivalent to the average unit value of beaches in CAs. The results are presented in Figure 7-3.

![Figure 7-2: Breakdown of TEV in MPA by ecosystem](image)

![Figure 7-3: Average unitary values in MPAs and CA by ecosystem (euros/km²/yr)](image)
These values are mostly lower than the values found in the literature. This is verified for mangroves for which the average total value of ecosystem services was found to be 515,000 euros per km² per yr (approx. ten times more than the value in our study). This is even truer for coastal systems for which an average value of 4.8 millions euros per km² per yr was found (Armstrong et al., 2010). A meta-analysis carried out on wetlands also resulted in a total average value of about 400,000 euros per km² per yr (Brander et al., 2006).

As a conclusion, while some important ecosystems within the region (namely seagrass, rocky bottoms, beaches) greatly contribute to the creation of economic value they are not considered at all in the management of coastal ecosystems (which focuses on the protection of mangroves and estuaries). The protection of coral bottoms and seagrass meadows (or at least a better understanding of these ecosystems as we know little of these ecosystems) seems essential for the protection of coastal and marine biodiversity in the region. I now look more closely at each MPA to see how these differences are distributed across large ecosystems, countries and sites. Also, an approach by MPA enables me to identify which MPAs are generating which benefits. The following sections present the results by MPA.

7.3.1 Langue de Barbarie MPA (Senegal)

The TEV of Langue de Barbarie MPA is evaluated at 384,000 euros per year. The TEV of the St. Louis CA, calculated based on the reference ecosystems surfaces of the MPA, is 388,000 euros. Based on these results, the Langue de Barbarie MPA generates a deficit of 3,700 euros.

The breakdown of value by ecosystems is presented in Figure 7-4. For the MPA, as for its CA, it is clear that estuaries, mangroves (despite their small size within both sites) and beaches are the three ecosystems that contribute most to the TEV. The seagrass is also very important for the MPA producing 48,500 euros for only 0.5 km² in surface area and 3% of the total area of the MPA. The TEV of the CA is very high thanks to the beaches value, which can be explained by important tourism in the CA (the beaches of Saint-Louis CA host several resorts).
7.3.2 Cacheu MPA (Guinea-Bissau)

The TEV of the Cacheu MPA is evaluated at 17.1 million euros per year. The mangrove ecosystem is by far the most important contributor to the creation of value in the MPA, with 15.2 million euros or 91% of the total. This dominance is largely due to the considerable coverage of the mangrove ecosystem in the MPA and the fact that most economic activities are associated with this ecosystem either directly (commercial and subsistence fishing, and logging) or indirectly (e.g. coastal protection, water treatment, carbon sequestration). Estuaries and channels are the second most important contributor, followed by mudflats, beaches and finally seagrass meadows. The distribution of these values is presented in Figure 7-5.

The Cacheu MPA generates a benefit of 235,000 euros compared to the CA. This benefit is attributed to the beaches (40%) and to mangroves (33%). For the other ecosystems, the values are similar between the MPA and CA.
7.3.3 Urok MPA (Guinea-Bissau)

The TEV of the Urok MPA is evaluated at 4 million euros per year. The mangrove ecosystem is the largest contributor to the TEV (77%), followed by estuaries (14%), mudflats (4%) and seagrass meadows (2%). The breakdown of these values by ecosystem is presented in Figure 7-6.

The Urok MPA generates a benefit of 704,000 euros compared to the CA. The benefit is attributed mainly to mangroves (667,000 euros) and rocky bottoms (24,000 euros).
The estimated benefit of the Urok MPA can be explained by the better health status of the ecosystems (especially mangrove and the predominance of *Rhizophora* and *Avicennia* in the total area of mangrove coverage) that contributes to higher indirect use values for the MPA than for the CA. The surplus generated by the better health status offsets the economic losses caused by the curbing of economic activities (including limiting mangrove logging, reduced exploitation by the commercial fisheries and sport fishing, or the ban on holiday tourism – as in the specific case of Urok).

### 7.3.4 Tristao MPA (Guinea)

The TEV of the Tristao MPA is evaluated at 153.6 million euros per year. The mangrove is again the greatest contributor to the creation of value (67%), followed by estuaries (18%) and mudflats (12%). The breakdown of these values ecosystem is presented in Figure 7-7.

The comparison of the Tristao MPA with its CA shows a deficit of 204,000 euros per year. This deficit is caused by the very high direct use values in the CA that is not offset by a better health status of the ecosystems in the MPA. Also, the non-use values are much higher in the CA.
7.3.5 Santa Luzia MPA (Cape Verde)

The TEV of the Santa Luzia MPA is evaluated annually at 1.31 million euros per year. All those ecosystems present in the MPA contribute almost equally to the creation of value: beaches (31%), coral bottoms (25%), rocky bottoms (24%) and seagrass meadows (20%). The breakdown of these values by ecosystem is presented in Figure 7-8.

The comparison between the MPA and its CA shows a large deficit for the MPA of 762,000 euros (58% of the total value of the MPA). This deficit is in large part caused by tourism activities concentrated on the beaches and rocky bottoms of the Sao Vicente CA (responsible for a deficit of 890,000 euros of direct use value in the MPA) not being fully offset by a surplus of indirect use values in the MPA. This comparison is, however, somewhat compromised since the two areas are very different: one is populated and the other not; one is easily accessible from Sao Vicente and the other not). As a result, the comparison could be improved by choosing another uninhabited island of the archipelago that is not protected.
7.4 Discussion of findings

7.4.1 A critique of the findings of research

The research developed in this study has enabled a comparison of the value of protected ecosystems (within a MPA) with unprotected areas in the vicinity of the studied MPA. A method that compares two different areas, one protected and the other unprotected, may however calculate other values than the sole benefits of conservation. Hence, simultaneous comparison enables avoid changes over time to be avoided that may not be caused by protection policy, but it certainly includes other factors in its valuation. These factors may include:

- differential economic activities: when the MPA and the CA have different activities that accrue to specific resources; this can influence the direct use values;

- external pressures on ecosystems: when ecosystems are exposed to different pressures of natural or human natures (e.g. upstream pollution, increased exposition to erosion, existence of industrial activities and subsequent pollution); this can influence the indirect use values;

- different populations: when the populations considered in the MPA and the CA are different in numbers, distribution in villages or cities. This can influence the results of direct use values and non-use values;
- different socio-cultural contexts: when traditions and the related uses of ecosystems for cultural or religious differ; this can highly influence the result for non-use values;

These factors have been insufficiently taken into account in my assessment of MPA benefits and deserve further investigation to be identified and evaluated. The valuation of conservation benefits should therefore mention the existence of such factors as part of the valuation study.

The approach and method adopted for this research have emphasized some shortcomings and points that deserve further discussion. This chapter therefore extends the research on the economic valuation of ecosystems by providing further insights into specific points that were judged as critical to the attainment of the objectives of the economic valuation of ecosystems. The chapter first discusses the shortcomings of the approach adopted as evidenced via the research and proposes some ways of improvement.

Also, the importance of socio-cultural context has been identified as one major issue during the research. It may largely interfere in the results of the economic valuation. This is discussed in a second sub-section.

Further, the question of time and space has been a recurrent concern throughout the research (e.g. the definition of boundaries, the population targeted, the time constraints of the project). The chapter therefore discusses time and geographical issues associated with the execution of valuation studies in a last sub-section.

7.4.2 Economic valuation and the socio-cultural context

Many scholars have focused their research on the interactions between people and nature. Across disciplines, they have developed various systems that all recognize the cultural differences as being very fundamental to the way people conceive of and relate to the environment (Brondizio et al., 2010). Descola (1996) proposes a three-tier analytical model to characterize implicit schemes of praxis (i.e. practical and applied knowledge) used by different societies to objectify their relationship to nature (modes of identification – animism, totemism, and naturalism; modes of interaction – based on reciprocity, predation, protection between species; modes of categorization – metaphoric similarities, analogy). Palsson (1996) describes three different paradigms for specific forms of people-nature interactions: orientalism, paternalism and communalism. Ellen (1996) proposes a model to interpret cultural variations on this relationship based on a comparative perspective to human cognitive imperatives: how
people perceive of things dependent on their senses, the context and value; which keys people use to distinguish self from others; and the different ways people see some inner force and essence in nature

Considering the great diversity in the people-nature relationships, the Judeo-Christian tradition that supports the utilitarian approach and the subsequent tool of economic valuation used for this research may not be compatible with all socio-cultural systems, the most remote being the indigenous understandings of people-nature relationship that acknowledge a continuum between human and non-human as part of a large chain of dependence and predator-prey interactions. As noted by Brondizio (p. 155), "economic values and processes of valuation, although grounded in a shared scientific methodology, are socially and culturally constructed, as are concepts such as ecosystems and biodiversity. Economic values are not objective facts nor do they reflect universal truths; instead they reflect the culturally constructed realities, worldviews, mindsets and belief systems of particular societies and/or sectors of society" (Brondizio et al., 2010; Wilk and Cliggett, 2006).

Taking stock of the importance of the socio-cultural context for the economic valuation, there has been several attempts to better integrate cultural views and perceptions in the valuation exercises. A report by the US Environmental Protection Agency (2009), for example, has proposed a range of multiple methods to integrate the various dimensions of ecosystems services. This report considers the possible use of not only economic methods, but also such alternative methods as measures of attitudes, preferences, and intentions; civic valuation; decision science approaches; ecosystem benefit indicators, biophysical ranking methods; and cost as a proxy for value (EPA, 2009). Not going that far in my study, however, I thought that socio-cultural information on the local populations should be further explored in complement to the economic valuation per se.

Also, some authors have clearly pointed out the methodological, practical and policy challenges associated with applying valuation techniques in developing countries, mostly referring to socio-cultural context issues. With regard to methodological issues for instance, low levels of literacy and education create barriers to valuing complex environmental goods and create difficulties in applying classical survey techniques such as questionnaires and interviews (Pascual et al., 2010; Christie et al., 2008). To avoid this, more participatory approaches to data collection may have better results
Considering the importance of such context, and the influence perceptions by populations may have on the TEV results, I decided to verify the understanding of the questionnaire by respondents \textit{a posteriori} and test their perception and knowledge of the ecosystems in order to: 1) validate the use of individual questionnaires in my research; and 2) grasp the cultural context that prevails in coastal communities and further describe the TEV results in the light of this perception. Thus, I have added questions pertaining to the perception and knowledge of respondents to their environment (ecosystems, ecosystem services, or threats to biodiversity).

The following proposes a discussion on the information collected about the knowledge of the environment in a first part, and the perception and attitudes towards change of ecosystems and its influence on the TEV results in a second part. This discussion is done with regards to the results of the economic valuation on the one hand and to the use for decision-making and management of biodiversity on the other hand.

\subsection*{7.4.2.1 Knowledge of the MCEs}

One section of the survey investigated the knowledge and perception of residents and tourists vis-à-vis the MCEs (see details of this section in the questionnaire in Appendix 4). Respondents were first asked to identify the main MCEs present in their region based on pictures presented to respondents. Pictures of all considered ecosystems were presented to respondents on a portfolio. Respondents were asked to point at the picture that related to the ecosystem the surveyor named.

The knowledge of ecosystems in this case is not the knowledge of the ecosystem itself (i.e. its specific fauna and flora, its uses, its dynamics and services) but rather the knowledge of the typology of ecosystems used by ecologists, the names associated with the different places of nature. This knowledge would therefore translate into an education about the environment. For this reason, it was expected that knowledge would be higher in MPAs (which could benefit from education from managers and local NGOs) than in CAs.

The results of this evaluation, synthesized in Figure 7-9, are very heterogeneous between countries (e.g. intermediate level or higher in Guinea, but average or bad in Senegal). As a reminder, the size of sample for each site is a population of 250 respondents.
In contrast, at the country level the results are relatively consistent between MPAs and CAs. The proximity of each MPA and its CA and the mobility of population between the two sites is certainly one of the reasons explaining these results in Senegal, Guinea Bissau and Guinea. Also, the behaviour of surveyors may have influenced the results: the way surveyors presented the results or influenced the respondents may have led to different results from one country to another.

Significantly, knowledge is slightly better in all MPAs than in their CAs, except for Santa Luzia. This better knowledge may be the result of awareness campaigns conducted in the MPA. It would, however, be necessary to undertake a further study to prove or disprove this result, especially as there is a causal link between a higher level of knowledge of MCEs and the establishment of MPAs. In Cape Verde, the result is very surprising because the villages where the survey took place benefit from a community centre for marine environment and fisheries (“casa do pescador”). The staff in that centre organise meetings about marine ecosystems management and awareness campaigns. So it is not surprising that the respondents from Santa Luzia village centres have an awareness of the marine environment (as pictured below in Figure 7-10).

49The surveyors for all surveys were recruited in the country and so each country has had different surveyors. One specific protocol that could have been implemented would be where surveyors from Guinea-Bissau would have asked questions in Tristao along with Guinean surveyors already present in Tristao. Unfortunately for logistical reasons (difficulties crossing the border between Guinea-Bissau and Guinea) this was not possible.
Other questions in the survey measured knowledge on the ecological services and resilience associated with MCEs (for example, related to the coastal protection offered by mangrove or its poor ability to regenerate after wood-cutting). These questions went beyond knowledge about typology of ecosystems and tested the knowledge of ecological functions and related services to the population\textsuperscript{50}. The results of this test are presented in Figure 7-11.

\textsuperscript{50} This included right or wrong test on sentences like: “Fish living offshore do not depend on coastal ecosystems for their survival” or “mangroves, seagrass and mudflats are important nursery sites for various species of fish and crustaceans”
Likewise, although the results are rather heterogeneous between countries, they are quite similar between the MPAs and their CAs. Except for Tristao MPA, all MPAs have a higher level of right answers, which translates into a better understanding of MCEs functioning (again, this may be explained by better awareness campaigns and communications about the importance of marine ecosystems for services provision).

7.4.2.2 Perception about threats and pressures to the ecosystems

The questionnaire also collected the perceptions of local people about their environment’s health status. Overall, respondents were rather concerned, even worried, about the future of MCEs. In Senegal and to a lesser extent in the Bijagos and Guinea, the majority of people consider that MCEs could really disappear from their region one day (Figure 7-12). The awareness of threats to the MCEs seems less present in Guinea Bissau mainland and especially in Cape Verde, where less than half of the respondents did not expect to see the MCEs disappear in the future.
Figure 7-12: Breakdown of answers to the question: “do you think MCEs could one day totally disappear?”

For this disappearance issue, the existence of an MPA (or not) does not seem crucial in the perception of people with regard to MCEs and their related health status. Thus, in 3 out of 4 MPAs, respondents have shown less concern for the disappearance of MCEs than in CAs. The main factor for this seems to be the effectiveness of the MPA and the existence of education and awareness campaigns on ecosystem protection. In Urok, the Tiniguena NGO has aimed to deliver education to the local population settled on the islands. It is the only MPA where there is a local NGO involved in education in all the studied MPAs. All other MPAs have local authorities that are responsible for MPA management but poorly involved in education. In Urok, the results are significant: an extra 15% of the population thinks that the MCEs could disappear one day (as opposed to findings in the Galinas CA).

In the Langue de Barbarie and its CA, the danger of degradation of MCEs is important for almost all surveyed populations. This may be thanks to the existence of the MPA (or its proximity to the CA), but it is more likely that a major ecological change (the opening of the breach in the Sandbar between the sea and the Senegal River) has made the local population understand the fragility of, and risks associated with, the protection of MCEs.
Surprisingly, despite these differences on the perceived risk of disappearance, a general consensus prevails in all sites on the question as to whether this disappearance would cause any trouble to the respondent: more than 4 out of 5 people consider ecosystem change would be a problem for them.

A further question investigated the perceived causes of degradation of MCEs. The results are presented in Figure 7-13.

![Figure 7-13: Breakdown of answers to the question: “what are, according to you, the main causes of degradation of MCEs in the region?”](image)

The surveyed population has largely identified human factors as the main cause of MCE degradation. These responses reflect recognition of the impact the population has on the ecosystems. The results for this question clearly favour the MPAs (except for Santa Luzia), with the population in MPAs having a greater perception of the human impact on MCEs.

The perception of the various impacts of human nature was further investigated. For each site, the level of importance of degradation due to human nature was tested. The level of importance ranged from not important, not very important, moderately important, to very important. The results are shown in Table 7-2.
Table 7-2: Perception of importance of human factors by the local population

<table>
<thead>
<tr>
<th>Site / Human degradation factors</th>
<th>Deforestation and mangrove wood cutting</th>
<th>Pollution (e.g. sewage, waste, release of chemicals)</th>
<th>Destructive fishing practices and overfishing</th>
<th>Overuse by tourists (e.g. sport fishing boat tours)</th>
<th>House building on the shore</th>
<th>Industrial exploitation (e.g. oil, gas)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Langue de Barbarie MPA</td>
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<td>Santa Luzia MPA</td>
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<td>Sao Vicente CA</td>
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N.B: the colour indicates the principal perception for the considered factor, ranging from green (not important), to yellow (quite important), to orange (important), to red (very important).

The results show that the factors considered to have the greatest impact on the MCEs surveyed are:

1) Destructive fishing and overfishing (very important for 5 sites out of 10);

2) Pollution was recognized as very important for 3 sites out of 10; and

3) Deforestation was seen as very important for 2 sites out of 10).

Overuse by tourists is considered to have a low impact (moderate importance for 2 sites out of 10), while house building on the shore and industrial development are considered to have the least impact (moderately important in the area of Kanfarandé).

It should be noted that the factors identified as the most destructive are those for which the impact is directly visible (wood cutting, over fishing, or pollution). Conversely,
factors that have a more indirect influence (building, heavy presence of tourists, or industrial activities) are considered by respondents as having a minor impact. These results indicate that despite the populations’ awareness of the threats to the MCEs, the lack of environmental education perhaps ensures that indirect impacts of certain practices that seem a priori relatively undestructive are overlooked. However, these activities, such as tourism industrial exploitation, and intense building on the shore are also absent in most studied sites. The surveyed population therefore had no tangible experience of the impacts of such practices that, unlike fishing, deforestation or pollution, had a visible impact on their environment.

7.4.2.3  Key findings about the socio-cultural context and influence on the TEV

The first conclusion after these rather positive results is that respondents, both in MPA and CA sites, were really responsive to my questions and demonstrated a good understanding of the content of the questionnaire. This therefore confirms that the methods applied using questionnaires is adapted to the socio-cultural context in West Africa.

The populations in the MPAs have shown better knowledge of the marine and coastal ecosystems typology (i.e. recognizing various habitats and naming each ecosystem without using vernacular terms). Further, their knowledge of the functions, goods and services these ecosystems provide are higher in MPA sites. Interestingly, this knowledge may not be the result of training by managers and NGOs but rather the result of traditional ecological knowledge.

Along with better knowledge, the perception of threats to the ecosystems is higher in MPAs. Further, inhabitants in MPAs are more conscious about their strong impact on the ecosystems than in CA sites.

These results, although expected, are all the more surprising as the WTP estimated is lower in MPA sites than in CA sites. As detailed in the earlier chapter, however, this may be explained by a “laissez-faire” attitude of the MPA inhabitants who believe that international donors can finance the protection of the ecosystems, while this is clearly not the case in CA sites. Paradoxically then, while MPA inhabitants are highly conscious that marine ecosystems should be protected, they are not ready to contribute to this protection. This result may demonstrate that WTP for protection does not fully reflect the non-use values for these ecosystems. This example shows that while non-use values associated with MCEs may exist, the local population is not contributing to the protection of these ecosystems.
In order to get improved results in non-use valuation, an alternative option could have been to ask as part of a choice experiment the following question (ahead of the presentation of scenarios): “if national and international donors refused to help protect these ecosystems, would you be willing to pay for the protection of this marine and coastal ecosystem?”

The additional information collected on the socio-cultural context thus greatly supports the work on the non-use values. It provides the necessary background material to analyse the data collected. The information on the perception of local populations also provides useful material for MPA design and management. It shows that the behaviour of populations can change over time. People gain knowledge about their environment, its functioning and the human and natural pressures that apply to it. For this reason, awareness campaigns are an important element for the implementation of an MPA. However, the gap observed between the ecosystem perception and the low willingness to pay for ecosystem protection may prevent effective MPA implementation as the local population is not contributing to the management of the MPA although they are keen to see the ecosystem services maintained. Based on this information, two management options are possible:

1) One option is the top-down implementation and management of the MPA, where the MPA is managed by national and local authorities. This is likely to ensure a strong management framework to the MPA. Participative management (through village committees for the MPA management) can be set up afterwards in order to increase social acceptability to the MPA.

2) Another option is the implementation of a communitarian MPA where support from international donors and NGOs is absent and management is the local population’s duty; this option is likely to prevent the local population from ignoring their responsibility to protect the MCEs. However, this option is only possible when the local population is generally willing to see the MCEs protected (as was the case in Urok where the local population witnessed a drastic decline in shellfish population that made them ask for support for the development of a communitarian MPA).

## 7.4.3 Spatial and temporal context in economic valuation

### 7.4.3.1 The question of time

The results as presented in chapter 3 are estimated on a per annum basis and calculated for the year 2013. This kind of evaluation put aside some important aspects
associated with time though. First, the comparison of MPA and CA sites is for the year 2013 only; it is a snapshot of the current situation and does not consider the past activities and the current trends in degradation (or improvement) of ecosystem health statuses.

Second, the economic value estimated does not reflect the degree of sustainability of the practices undertaken. It values all activities that provide direct or indirect economic assets (or more intangible value in the case of non-use valuation) but does not consider the permanence of these activities over time. For instance, the fisheries service in Tristao has generated a very high value but this service was started only a few years ago and will likely go on for only a few years before stocks will be depleted and fishermen forced to move to another fishing ground. This practice is therefore unsustainable, although its economic value in 2013 is very high.

To better integrate sustainability into the economic valuation, several methods can be applied. First, for the fisheries provisioning service, the economic value could be estimated with regards to the Maximum Sustainable Yield level of stocks. If beyond the MSY level, the value of fisheries should be reduced to introduce a level of sustainability. Unfortunately, the MSY levels are not known with precision at these sites and, except from the observation that stock are being overfished and their size diminishes, it is difficult to set a MSY level without precise stock assessment data. Instead, this study has introduced fisheries biomass production as an indirect use value. This value aims to reflect the sustainability of fisheries: it counterbalances the value obtained with fisheries use value. But the biomass production levels are based on literature references and not necessarily on the local biomass production observed. Hence, further research should be carried out in the studied sites (or even on the marine and coastal ecosystems of West Africa for which little information is available) to help refine the fisheries biomass production level.

Second, the economic value could be estimated over a certain period of time (from 10 to 25 years) through the net present value (NPV) method. This NPV is the sum of present values over a time series. Each present value is discounted using a discount rate (see section 2.4.1 for details about discount rate and the excellent chapter on discounting, ethics and options for maintaining biodiversity and ecosystem integrity by

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51 Maximum Sustainable Yield (or MSY) is the level of fishing that maximizes the volume of catch without jeopardizing the production of the stock. It is associated with the notion of sustainable harvest as beyond the MSY level the fishing effort increases while the volume of catches diminishes (because of the depletion of the stock).
Gowdy in the TEEB main publication – Gowdy et al, 2010). However, such valuation is in my opinion very imprecise and does not necessarily translate in reality because of its high speculative aspect. In addition, the NPV obtained may be very high (amounting millions or billions of euros) and therefore only useful for advocacy in order to highlight the critical importance of ecosystem protection.

Third, the economic valuation could have been carried out over a longer period of time, in order to better measure the impact of MPA implementation or management. With the same measurement protocol, the economic valuation on the same sites would have been much more precise and useful to decision-makers. A recent study has proved that the economic benefits of MPAs (mainly thanks to increased fishing in adjacent waters and development of tourism) can be observed in as little as 5 years (Sala et al., 2013). However, I doubt the same time period would lead to MPA benefits in West Africa, because of the four following reasons:

- The economic benefits observed in the study by Sala and colleagues relates mostly to sites with high tourism potential (or with great opportunities for tourism development) which is not the case in most sites in West Africa.
- Most MPAs have difficulties enforcing their management plan (when they have adopted one) and preventing illegal fishing in the MPA waters. This is likely to limit the spill-over effect (dispersal of larvae and adult beyond the limits of the MPA) and so will limit benefits for fisheries in adjacent waters.
- The MPAs considered in the sole study are marine reserves and mostly no-take areas, which is not the case in my West African MPAs; this difference in activities within the MPA boundaries substantially modifies the observed ecological benefits on the adjacent waters.
- Most MPA economic benefits considered in my study relate to the indirect benefits of good health status of ecosystems (such as coastal protection, water treatment) which requires more time to show benefits than the development of tourism activities.

For these reasons, I estimate that the minimum time period to measure the economic benefits of MPAs in West Africa is about 10 to 15 years from the time when the MPA is implemented and its management plan effectively enforced. This time period is incompatible with normal research project length, which makes such a protocol difficult to implement.
As a consequence of these discussion points, this study should be considered as a snapshot in time study and caveats with regards to sustainability and trends in exploitation and related ecological statuses of ecosystems should be considered. One should also consider the practical barriers (in terms of research length, costs, or interests from decision-makers) to accurately ascertain an economic valuation that takes account of time and sustainability criteria.

### 7.4.3.2 The question of space

Another key factor that may influence the economic valuation is space and, more precisely, the spatial scale to which the economic valuation applies. The role of spatial scale in valuation studies has been poorly investigated (Kerkhof et al., 2010). One part of the only studies that addressed this question investigated the sensitivity of respondents to the physical dimension of the valued item (Brander et al. for instance showed that reef recreationists are sensitive to the scope of the area they visit – Brander et al., 2007). More interestingly, some other studies distinguish spatial scales at which ecosystem services are being supplied and demanded by stakeholders (Hein et al. 2006; Mander et al., 2007; Martin-Lopez et al. 2009). This will be discussed below in a first sub-section. Another issue is the scale used for the estimation of non-use valuation and the geographical answer to the question “who are the non-users?” This will be discussed in a second sub-section.

A last issue discussed here is the scaling of economic valuation that could be envisaged for a regional economic valuation of ecosystems in West Africa, based on the research undertaken on the sampling of MPAs and using the transfer of benefits.

#### 7.4.3.2.1 Various spatial scales of ecosystem services and subsequent values

Hence, different types of ecosystem are valued differently as the spatial scale of the analysis varies. The ecosystem services generated at a particular ecological level may be enjoyed by stakeholders from a wide range of institutional scales. Similarly, stakeholders from a single institutional scale may receive services generated at a wide range of ecosystem scales (Hein et al., 2006; Martin-Lopez et al., 2007). As a consequence, some values are meant at the local scale of the ecosystem. This is the case with most provisioning services (e.g. subsistence fisheries, tourism, mangrove woodcutting) and some regulating services (coastal protection, fisheries biomass production) that deliver local values. Other values have more regional beneficiaries such as commercial fisheries (that benefit the markets in the neighbourhood or further within the landlocked countries for instance), the tourism activity for operators located not far from the ecosystems, the water treatment that benefit the regional waters. Some
other values have global beneficiaries. This is the case with pharmaceutical uses that may lead to the development of medicines all over the world, or carbon sequestration that benefits the composition of the atmosphere globally.

These differences in values at a spatial scale may cause variations in the estimation of a TEV according to the different calculation techniques used. For direct market pricing and cost-based methods, the spatial scale does not influence the final result, as long as you are able to estimate the total production in the area considered and you only consider producers' turnover (not the added-value created along the value chain once the product has left the area considered; for instance the turnover of fishmongers on the remote markets that sell fish caught in my area of study).

For non-market methods, the spatial scale plays a very significant role because of the population surveyed about their revealed or stated preferences. Martin-Lopez (2007) has demonstrated that a multi-scale approach should be applied instead of a global scale for travel cost methods used to estimate cultural services.

7.4.3.2.2 Spatial scale and the non-use valuation

Beyond the non-market methods, the spatial scale has a specific importance for non-use valuation. Thus, the population that support use values is relatively easy to qualify. But non-users are much more difficult to demarcate. While it is quite easy to define the users of ecosystems, it is much more difficult to answer the question “who are the non-users of my ecosystem?” For well-known large ecosystems (e.g. the Great Barrier Reef, the Amazonian Forest), the people that bear non-use values for these ecosystems are likely to be considered at an international scale. In this case, it is reasonable to apply the non-use valuation to the World population. For less well-known ecosystems such as the West African ecoregions and their subsequent marine and coastal ecosystems, the definition of non-users is much more difficult. In my research, I decided (for practical and financial reasons mostly) to consider the two population of local residents and visitors as non-users. I omitted all non-users that may have non-use values for these sites but have not been considered in the total population.

For non-use valuation, I suggest adapting the spatial scale to the scale of the institution that will use the economic valuation. or, in the absence of identified beneficiary of the valuation study, the scale of the main authority that manages the ecosystem. In my study, for instance, I maintained a local scale for the non-use valuation because the beneficiaries were local to national decision-makers in the first instance. In addition to the scope of the valuation, the question of available means is to be considered since a
large-scale valuation may be costly and requires a large sampling of population to be surveyed.

An extreme case for the non-use valuation is when the ecosystems are uninhabited and unvisited. This is the case of small remote islands such as a lot of unknown atolls in the Pacific (e.g. Clipperton, the Tuamotu Archipelago) or subantarctic islands (South Georgia, Kerguelen for instance). In this case, the reference population cannot be local. Rather, the population considered for the valuation is likely to be the population from the main administrative authority to manage these territories: the French population in the case of Clipperton or Kerguelen, or the UK population in the case of South Georgia, for instance.

This issue of spatial scale for the non-use valuation should however be further investigated while it has not been treated by the scientific literature so far.

7.4.3.2.3 Scaling up economic valuation

The question of scaling up valuation studies was first introduced with the emergence of international valuation exercises such as the one carried out by Costanza on the global ecosystems (Costanza et al., 1997). Some methodological issues have been raised following these exercises on the caveats associated with scaling up local studies to regional or global scales. These caveats are mostly to do with the benefit transfer when applied to very different sites or at a much larger scale (see section 2.3.2.4).

First, as Kerkhof et al. (2010, p. 131) states: “environmental public goods are different from private commodities in the sense that they are collectively consumed and indivisible. At low spatial scales, the differences between environmental public goods and private commodities are rather small. For example, all individuals in a neighbourhood may enjoy the scenic beauty and the fresh air of an urban park. The number of people in a neighbourhood is relatively small. At higher spatial scales, however, the analogy between private goods and environmental public goods is more and more blurred, as the number of individuals who benefit from the environmental public good increases”. There are therefore limits to the analogies we can draw between the economic values of nonmarket environmental goods and those associated with private goods (Bockstael et al. 2000).

Second, respondents of surveys to measure stated preference methods may be less acquainted with changes in environmental public goods at high spatial scale (Kerkhoff et al., 2010). It is hence very difficult to grasp all of the natural assets considered in large scale studies through a short interview. This incompatibility between contingent
valuation and high scale of valuation is the main limitation to the use of such method on large area. It also highlights the difficulties of non-use valuation at high scale.

Third, the ecosystem services considered may be considerably heterogeneous at higher scale. This is caused by differences of main characteristics of climate, water circulation, effects of coastal erosion and pollution intensity. As a result, the ecological functions quantified in one site may not be homogeneous for the whole area considered, causing uncertainties in scaling up the data collected.

7.5 Conclusion of the chapter
This chapter has presented the results of the economic valuation in the five MPA sites. It has described the calculation techniques used for each component of the TEV and detailed the results in each site. A final section has provided a synthesis of the results for all sites, and then by component of the TEV (by ecosystem). This chapter has provided some insights into the findings which have been discussed. For example, I have acknowledged the biases in the valuation and the problems caused by the lack of data available, along with other limitations. Second, I have highlighted the extreme importance of considering the socio-cultural context as part of the valuation studies, and even more so for non-market valuation. Third, the discussion section has raised the question of time and space scales within the valuation studies. These are very important question to be considered according to the valuation objectives and final use for decision-making. The next chapter extends this discussion by considering the perception of local populations and the results of the survey as regards the knowledge of these populations about the coastal ecosystems and the need for their protection.
8 Chapter 8: public policy considerations

The previous chapter has shown that economic valuation of ecosystems can be seen as a cultural projection that imposes a way of thinking and a form of interaction with nature. However, as Arrow (1982) and Sen (1973) have demonstrated, valuation can serve as a tool for self-reflection which helps people rethink their relationship with nature and increases knowledge about the consequences of their consumer choices and behaviour. Economic valuation therefore has the (p. 174) “potential to serve as a tool of awareness and as a feedback mechanism for a society that has distanced itself from the resources it uses and from the impacts of its uses on distant ecosystems and people” (Brondizio, et al., 2010). And the researcher plays a central role in undertaking this economic valuation.

In this chapter, I explore the importance of economic valuation from a public policy perspective. I first provide an overview of how economic valuation is used for decision-making today. I provide an overview of an original typology for the economics of ecosystems (applied to the specific example of coral reef ecosystems). Then I discuss the various roles envisaged for economic valuation to influence decision-making and how the valuation is effectively implemented for decision-making purposes.

In a second section, I discuss the added-value of my method to value the net benefits of conservation based on the TEV approach with regards to a better use of valuation in decision-making. Furthermore, I propose some specific tools that could feed into the decision-making process. These specific tools are presented through the example of the West African MPA and using the results from my study.

8.1 Economic valuation for decision-making: a review

8.1.1 The three approaches of the economics of ecosystems

A literature review on the economics of ecosystems and their uses for decision-making has enabled me to categorize the economics of ecosystems into three different economic approaches: the economics of welfare, the economics of degradation and the economics of protection. As an illustration of this categorization, an overview of these three approaches is provide below, applied to coral reefs, mangroves and seagrass meadows ecosystems, for which the literature on such issue is the most extended52.

52 This section is extracted from text prepared by the author for publication in the Journal of Environmental Management released in early 2013 (Laurans, Pascal, Binet et al., 2013)
The economics of degradation concentrates on the assessment of impacts of human activities on coral reefs. This approach was developed first at the end of the 1980s through three seminal papers: Hodgson and Dixon (1988), McAllistair (1988) and Hundloe et al. (1987). This category of economic valuation consists of comparing private benefits and social costs associated with human activities that impact coral reefs, so as to demonstrate the negative impact of those activities for society, when external (thus often hidden) costs are taken into account. “External costs” are costs borne by an economic agent, that are created by the activity of another agent, who does not consider and thus integrate them in their reasoning (Meade, 1979). For instance, when people practice blast fishing, they tend to deplete the available stock of fishes, thus affecting the wealth and well-being of other fishermen or of businesses that make a living from a healthy and scenic coral reef (Pet-Soede et al., 1999). Cyanide fishing (Mous et al., 2000), coral mining (Berg et al., 1998; Ohman and Cesar, 2000; Cesar and Chong, 2004) and tourist overuse (van Beukering and Cesar, 2004) are other reported and evaluated typical external costs in coral reefs regions. Another set of studies pertains to this category, in that they investigate the costs of degradation from anthropogenic threats to the coral reefs at the global scale: impacts of climate change and coral bleaching (Cesar and Chong, 2004; Westmacott et al., 2000), ocean acidification (Brander et al., 2009) or of algae blooms (van Beukering and Cesar, 2004). These studies largely advocate for bans on destructive practices, or for strengthening preventive policies by assessing the cost of policy inaction.

Some researchers have also quantified the costs of policy inaction of destructive practices, which is the total economic and social cost that would result from not passing or enforcing adequate regulations, thus ensuring depletion of natural resources and of related economic activities, as well as loss of natural capital. For example, in Indonesia the cost of not enforcing blast fishing regulation during the 1990s is estimated at US$3.8 billion (Pet-Soede et al., 1999). Economics of degradation analyses are thus intended to justify conservation measures, explicitly or implicitly.

The economics of welfare stems from the recognition of dependence of human beings on the provision of coral reef services and the contribution of coral reefs to coastal and national economies (Costanza et al., 1997; MEA, 2005; Haines-Young and Potschin, 2010). Studies with this perspective aim to provide an overall value of the coral reefs and generally frame valuations in terms of Total Economic Value (TEV, see below), which allows the identification of economic agents and sectors that are associated with the components of the TEV. High estimates of TEVs are then used to make the case for
considering coral reef conservation in the national decision-making process. Hence, the TEV of Martinique’s coral reefs were estimated to be almost US$100 million a year (Failler et al., 2010), US$360 million a year for Hawaii (van Beukering & Cesar, 2004) and US$14,300 per km² in Samoa (Spurgeon, 2004). “Economics of welfare” analyses are thus intended to advocate generally for a better inclusion of coral reefs in the choices of stakeholders, when their behaviour is likely to influence the reef’s condition.

The economics of protection and management of natural resources involves the valuation of benefits from marine biodiversity conservation and management. Studies have valued the net benefits of conservation to society by assessing its return on investment (i.e. the series of gains from conservation policy or project minus the series of costs of conservation, divided by the latter to obtain the ROI ratio), or its net revenues (revenues generated by conservation minus its costs). For instance, the coral reefs and mangroves of Olango Island in Philippines generate annual revenues of US$1.53 to 2.54 million, whereas the costs of conserving this environment are estimated to be less than US$100,000 a year, a more than ten-fold difference (White et al., 2000). This provides a strong message in support of the conservation and management of such ecosystems. Other studies value the preference of users for ecosystems in a good ecological state, which is most often carried out through non-market valuation methods, such as contingent valuation (Spash, 2000). Contingent valuation is a means to simulate this absent market by eliciting, through surveys, individuals’ willingness to pay for the preservation of the given services, or willingness to accept their loss. Another category of economic “valuation for conservation” focuses on the costs and benefits of specific conservation measures, such as the establishment of marine protected areas (Dixon et al., 1995; Subade, 2007). Finally, some authors have also been interested in the restoration of critical habitats. This approach aims to compare the costs and benefits of restoring degraded ecosystems (Spurgeon and Lindahl, 2000) or rehabilitating and creating habitats (Ibid). “Economics of conservation” analyses are thus intended to assess the economic opportunity created by protection measures, from a general social perspective. They are the mirror image of the “economics of degradation”.

8.1.2 The actual use of economic valuation for decision-making
The economic valuation of ecosystems is recognized as one useful (necessary for some authors) tool for decision-making. It is thought to influence policy for coping with the accelerating degradation of ecosystem services and biodiversity (NRC, 2005). It is seen by many as a prerequisite for better management decisions (e.g. Randall, 1988; Daily et
al., 2009). Contrary to those assertions symbolized by the general diagnosis ‘we don’t protect what we don’t value’ made by Myers and Reichert (1997), there are about the same proportion of researchers who claim that valuation is neither necessary nor sufficient for conservation (Heal, 2000) or for coherent and consistent choices about the environment (Vatn and Bromley, 1994). These oppositions are connected to the conceptual opposition between environmental economists and ecological economists that was developed in the Chapter 2.

To make progress about this debate, authors have stressed the importance of understanding if and how the economic valuation exercises were used or expected to be used (Fisher et al., 2008; Gowan et al., 2006; Navrud - in OECD, 2002; Pearce and Seccombe-Hett, 2000; Liu et al., 2010). In order to judge the use and influence of the economic valuation on decision-making, a team of researchers led by Laurans have reviewed the literature in order to highlight the use of such economic valuation of ecosystems in decision-making (Laurans et al., 2013). It shows that the literature gives little attention to this issue and rarely reports cases where the economic valuation of ecosystems has been effectively put to actual use. Interestingly, at the same time such use is frequently referred to as being the goal and justification of the economic valuation implementation.

Laurans et al. (2013) have defined a typology for the economic valuation of ecosystems uses that include: decisive valuation; ‘technical’ valuation; and informative valuation. Not being far from the typology of approaches defined in section above, this typology is yet more policy-oriented. The first one is the decisive valuation when the valuation is meant to inform a specific decision. This exercise comes generally ex ante and is included as part of a general cost-benefit analysis (CBA). This type includes three sub-categories:

- The valuation for trade-offs: in this case the valuation aims to integrate concerns within a CBA. The valuation should (p. 212) “enable the policymaker to optimize social well-being by making choices that balance out preference criteria” (Laurans et al., 2013);

- The participative valuation where valuation is considered as a ‘negotiation language’ (Henry, 1984; 1989; Laurans et al., 2013); in this case valuation is used as a basis for discussion between various stakeholders; and
- The valuation as a criterion for environmental management: when valuation can help prioritizing conservation expenses; valuation is used as a management tool.

Second, the ‘technical’ valuation is used for the design of an instrument. This type of valuation occurs after the choice or the design of the project has been made in order to adjust the economic instrument that will implement the decision. It involves two main categories:

- The valuation for establishing levels of damage compensation: when the valuation provides guidance for administrative or regulatory decisions in court or for large project approval; and
- The valuation for price-setting: when the valuation is used to determine the payment vehicle for environmental management (e.g. entrance fees, taxes);

Third, informative valuation consists of an indirect use of the valuation for decision-making. It plays, as the OECD states, (p. 212) “an important role in educating decision-makers about biodiversity benefits” (OECD, 2001; Laurans et al., 2013). This type includes three categories:

- The valuation for awareness-raising: when the valuation is a vector to ensure that ecosystem services considerations are integrated into public and private choices;
- The valuation for justification and support: when the valuation is used to promote a given course of action; it is different from the valuation for trade-offs where valuation inform on the optima choice; and
- The valuation for producing 'accounting indicators': when the valuation is used to monitor the state of the natural capital and integrate this into the framework of decision-making.

Scholars have browsed 313 articles from *Ecological Economics* which make reference to the economic valuation of ecosystems. The result is striking: out of these articles, only eight articles (2%) describe how the valuation has played an important role in decision-making! For most references, the use of the economic valuation is expected to be used for informative purposes only. Whilst the authors have expressed some biases in their review, the results still show that the economic valuation is not often used by decision-makers. Laurans and his colleagues have then proposed further avenues for improvement of the economic valuation of ecosystems through: i) the creation of a
specific field of research; ii) the refining of valuation techniques; and iii) the change of context of use.

8.2 Valuing net benefits of conservation in protected areas: a new area for policy-oriented research?

Taking stock of the observation of poor use of economic valuation exercises for decision-making and propositions for enhancing this use in future decision-making (section 8.1.2), I have proposed in this thesis a new scope for the economic valuation. Thus, the economic valuation of the benefits of conservation suggests switching from an ecosystem-oriented valuation towards the valuation of a management process. This change hence brings new opportunity for uses of the economic valuation on the monitoring and evaluation of biodiversity conservation.

While this new valuation offers new opportunities of uses, it can still be used for the economic valuation of ecosystems. Hence, the fact that I first value the ecosystems before comparing it to the same ecosystems in a comparison area still provides the information on the value of the ecosystems. This means that the work can be used for all uses as described in an earlier section (i.e. decisive, technical and informative valuation). I anticipate that the research presented in this study may therefore increase the potential use of economic valuation when applied to the specific case of protected areas.

In the following, I envisage some developments to the research carried out here and potential applications of the economic valuation exercise as an approach to support decision-makers at local, national and international scale. In particular, I see this approach as a way to support:

- Management based on the comparison of benefits brought by the MPA and the costs incurred by its management (sub-section 1);
- Advocacy and management based on the recognition of indirect use values beyond the direct use values for MPA managers and local decision-makers (sub-section 2);
- Advocacy for action on biodiversity conservation through the definition of the costs of policy inaction for national policymakers and international donors (sub-section 3); and
- Sustainable financing through the application of payments for ecosystem services for MPA managers, as well as national decision-makers (when conservation is state-funded) and international donors (sub-section 4)
8.2.1 Comparing economic benefits of MPAs and management costs

One of the most common tools derived from economic valuation is the costs-benefits analysis which was presented in chapter 2. This analysis compares the benefits generated by the MPAs (when compared with CAs) with the management costs incurred in the MPA\textsuperscript{53}. In the case where MPA value is less than in the CA, there is no need to compare their value with management costs: this deficit sends a clear message that the MPA is somehow inefficient because it is a “paper MPA” that has no management plan and no measures enforced. An alternative solution could be that the MPA has been created recently (and its management plan well enforced) but the poor health status resulting from degradation of ecosystems has not been compensated by the protection offered by the MPA yet.

The fact that the MPA was designated in this place highlights the ecological importance of the ecosystems there (for fisheries biomass production mostly) so the MPAs are more inclined to be overexploited than other neighbouring areas for their biological importance.

For those MPAs that generate benefits when compared to CAs, the comparison between benefits brought by the protection and the management costs\textsuperscript{54} can be meaningful.

When applied to the case study on the sample of five MPAs in West Africa, the comparison shows interesting results that are presented in Table 8-1.

Table 8-1: Net benefits of MPA protection

<table>
<thead>
<tr>
<th>MPA</th>
<th>Benefits of protection</th>
<th>Management costs</th>
<th>Net benefits</th>
</tr>
</thead>
<tbody>
<tr>
<td>Langue de Barbarie MPA</td>
<td>Negative</td>
<td></td>
<td>Negative</td>
</tr>
<tr>
<td>Cacheu MPA</td>
<td>213,000 euros per year</td>
<td>About 220,000 euros per year</td>
<td>-7,000 euros per year</td>
</tr>
<tr>
<td>Urok MPA</td>
<td>704,000 euros per year</td>
<td>About 150,000 euros per year</td>
<td>+554,000 euros per year</td>
</tr>
<tr>
<td>Tristao MPA</td>
<td>Negative</td>
<td></td>
<td>Negative</td>
</tr>
<tr>
<td>Santa Luzia MPA</td>
<td>Negative</td>
<td></td>
<td>Negative</td>
</tr>
</tbody>
</table>

\textsuperscript{53} MPA management costs are the costs incurred every year for the functioning of the MPA; they mostly include salaries and maintenance of equipment. They are different from the transaction costs incurred for the development of the MPA (feasibility study, infrastructure development, major investment in management capabilities, etc).

\textsuperscript{54} The management costs were extracted from the toolkit for sustainable financing of MPA project led by IUCN in West Africa: https://cmsdata.iucn.org/downloads/boite_a_outils_durable_financiere_rampao_2.pdf
The only MPA to provide net benefits is the Urok MPA. This result is not surprising as Urok is managed by a very proactive NGO (Tiniguena) which is located within the Urok archipelago and has based its management on a participative approach. Thus, management is coordinated by the village council which decides on the measures to be implemented and enforced. This increases social acceptability of the MPA and ensures further compliance by the local population with management measures.

The Cacheu MPA is almost at break-even. It is run by the IBAP (Biodiversity and Protected Area Institute) located in Bissau and with an office in Cacheu composed of fewer than 10 persons (including one MPA manager). The Cacheu MPA management process is more top-down than in Urok and enforcement is made difficult by the poor acceptance of management measures by the local populations, the large surface covered by the MPA, and the poor human and technical resources available for enforcement of management measures. However, there are various initiatives carried out, such as annual replanting of mangrove campaigns that contribute to the provision of benefits in the MPA. In total, the MPA costs exceed benefits by 7,000 euros per year; meaning that the MPA could easily deliver net benefits by improving its management and enforcement slightly.

The comparison of benefits with management costs can be a good indicator to check MPA management efficiency and allows international donors (as they are the main source of funding for MPAs in West Africa) to evaluate their support to the MPA. This comparison can include non-market values that enable stakeholders to have a good view of MPA effectiveness. This approach is much more appropriate to the socioeconomic context of West Africa than in cost-benefit analysis that only considers market values. In West Africa, tourism is less developed and does not compensate for the limitation of extractive uses enforced by the MPA management. In Urok, tourism is not even permitted and, as a result, a cost-benefit analysis that only looked at market values would show an important loss of value to the economic development of the MPA. Through my method, however, I have shown that Urok is the MPA that provide the higher benefits thanks to the improved health status of its ecosystems. Through my method again, I have shown that “paper MPAs” that do not enforce limitations of destructive uses, bear net losses of value when considering not only the high market value but also the degraded ecosystems. A classic cost-benefit analysis (different from
the method used here) would in this case show benefits to the MPA, whereas it is not delivering any biodiversity benefit.

Importantly, the net benefits calculation here does not consider possible changes of practices developed by users following the creation of the MPA (opportunity costs) and the creation of the MPA and the subsequent investment required to establish and develop the MPA (transaction costs). Rather, the net benefit evaluation only considers MPAs that are established and running. Such evaluation should be carried out over a longer period which was not possible in my case.

### 8.2.2 Indirect use values as a support to advocacy and management

The indirect use values represent nearly 81% of the TEV in MPAs. This important contribution to the total value reflects the leading role played by support and regulating services in creating economic and social value in MPAs: fisheries biomass production, water and waste treatment, carbon sequestration and coastal protection. This is the first time in West Africa that such non-market values have been estimated in monetary terms and compared to market values (both extractive and non-extractive activities) for the MPAs of the region. The result shows that: the indirect use value of marine and coastal ecosystems is more than 4 times greater than the direct use values of commercial activities. The mangrove ecosystem comes first amongst the studied ecosystems, the indirect use values of these ecosystems being 24.5 million euros.

The recognition of indirect use values may aid lobbying for biodiversity conservation by environmental NGOs and MPA managers. It shows that there are monetary values that are associated with the ecological functions of ecosystems and that benefit local and international populations. These values can be much higher than the tangible direct use values (and lost if the ecosystems are not protected).

The indirect use values from support and regulating services should also be considered in coastal management and the development of MPAs in particular. These values are driven by the good health status of ecosystems in MPAs. For this reason, the management of MPAs should seek to improve this health status in order to maximize the indirect use values. This priority should be considered within the definition of the management plan and objectives of each MPA.

Consideration of indirect use values in advocacy for ecosystem protection and better management of MPA can have several outcomes:
- It helps to give focus to the overexploitation of ecosystems by extractive uses (such as logging, fishing) and recognizes the importance of limiting these uses in economic terms;
- It can help to prohibit the destructive practices that are damaging the ecosystems (such as deforestation for rice planting and destructive fishing practices);
- It can help promote the restoration of important coastal ecosystems (e.g. replanting propagules in mangrove ecosystems);
- It offers suggestions for further research into the under-researched ecosystems in the region: seagrass meadows from Senegal to Guinea, the coral bottoms and beaches in Cape Verde)

Finally, the consideration of indirect use values promotes ecosystem-based management of MPAs, instead of species-driven management. Currently, most MPAs in West Africa have management objectives associated with the protection of emblematic species (i.e. manatees, migrating birds, hippos, dolphins, turtles). This management approach, however, does not recognize the importance of ecosystems and their contribution to the provision of services to the local population. Ecosystem-based MPA management is likely to better reconnect environmental objectives with local development. It also indirectly contributes to the development of better habitats for those emblematic species, which is today the primary objective of West African MPAs.

The better integration of indirect use values – or indirect ecosystem services when using an ecosystem services approach – can substantially help decision-making and local MPA management. It has barely been documented in scientific literature, except by a recent study of Rees et al. (e.g. Rees et al., 2012).

8.2.3 Economic losses and the cost of policy inaction
The cost of policy inaction (COPI) is one concept developed recently for biodiversity conservation within the TEEB project (see details in chapter 2). The COPI approach was applied to the case of not meeting the 2010 target to halt biodiversity loss. A similar approach can be applied here, based on the results of the research for the costs incurred by not protecting the MCEs in West Africa. The evaluation of costs is based on the loss of ecosystem surfaces in the region due mainly to two processes: coastal erosion on the one hand and deforestation of the mangrove ecosystem on the other hand. Coastal erosion is partly due to policy inaction with regards to stopping the
destruction of mangroves and seagrass that provides coastal protection. It is also caused by the lack of actions limiting house building on the beaches and the use of sand for construction. MPAs could possibly reduce such threats to coastal ecosystems and thus reduce erosion. However, such measures are not likely to stop coastal erosion because of climate change and its consequences. As regards mangrove deforestation, MPAs may contribute significantly to reducing the pressure by humans for wood-cutting, though natural factors (climate change, storms, etc.) may still contribute to the reduction of mangrove ecosystems.

The following section details the economic losses caused by these two factors and provides estimates of the cost of policy inaction in the sub-region. There are surely other processes that cause ecosystem surface areas to decrease (and, more importantly, degradation of ecosystem health status), but these are not considered here. They could, however, be evaluated in a more in-depth study of the costs of policy inaction in West Africa.

8.2.3.1 Coastal erosion
The available literature in the region on coastal erosion shows that it affects all sandy beaches between Mauritania and Guinea-Bissau at varying speeds, generally less than or equal to 2.5 metres per year (Faye, 2010). This value excludes the sandy spits that are characterized by a higher rate of change, gaining or losing a dozen metres to several hundred metres per year (Faye, 2010).

The same study estimated the total length of coastline and its proportion of beaches, cliffs and mudflats. The values obtained are approximately:

- For the upwelling large marine ecosystem in Mauritania and Senegal: 813 km of beaches and 825 km of mudflats (including sebkhas\(^{55}\) that may be associated with mudflats) and
- For the estuaries and mangroves large ecosystem from southern Senegal to Guinea-Bissau: 960 km of beaches.

Then, it is possible to estimate the annual loss of surface area for each ecosystem based on the annual retreat of the coastline for each ecosystem. Using the average values

\(^{55}\text{Sebkhas means "salt flats" in Arabic. They are located in arid areas above the limit of upper tide. They are generally created between crests of dunes by flooding that makes small ponds highly concentrated in salt.}\)
estimated for each ecosystem in the two largest marine ecosystems\textsuperscript{56}, it is possible to calculate the loss of value associated with coastal erosion. For the upwelling large marine ecosystem, the economic loss caused by coastal erosion is about 45,000 euros for the estuaries and 144,000 euros for the mangrove large marine ecosystem from Senegal to Guinea-Bissau

Finally, the annual loss of value caused by the reduction of beach in the region between Mauritania and Guinea-Bissau is 190,000 euros. Extrapolating these values to include Guinea and Sierra Leone (a linear range of 1.5 times) the estimated annual economic loss caused by coastal erosion in the sub-region is approximately 410,000 euros. This value is, however, very conservative because it ignores the forecasts for sea level rise and the degradation of ecosystems that helps to control the erosion (mangrove, seagrass). Also, while urban areas are not considered here, it is anticipated that coastal erosion in these areas (the peninsulas of Dakar and Conakry in particular) could be well above the costs estimated here.

As a result of the calculation of very conservative values and the fact that policy action against coastal erosion may not fully stop a phenomenon that is also influenced by climate change, the calculated value seems a good estimate of the costs of policy inaction against coastal erosion.

\textbf{8.2.3.2 Mangrove deforestation}

A study of spatial distribution of ecosystems conducted by the United Nations Environment Programme on the mangrove area in 19 countries of the West African region shows that mangrove area dropped 25\% between 1980 and 2006 (UNEP, 2007). According to the same study, the mangroves from Mauritania to Sierra Leone saw their surface area reduced from approximately 11,000 km\textsuperscript{2} in 1997 to just 8,000 km\textsuperscript{2} in 2006, a decrease of about 40\%. Assuming an average rate of mangrove loss of 25\% over 20 years (conservative estimate), this would result in an additional loss of 2000 km\textsuperscript{2} by 2026 compared to 2006. This is equivalent to an annual decline in mangrove area of 100 km\textsuperscript{2} per year. This study has established the average value of mangrove to be about 50,000 euros per km\textsuperscript{2} per year (55,000 euros/km\textsuperscript{2}/yr for MPAs and 49,800 euros/km\textsuperscript{2}/yr for unprotected ecosystems – which is the most common

\textsuperscript{56} Based on the results of the economic valuation carried out, we consider the following average values: 20,000 euros/km\textsuperscript{2}/yr for the beaches and 2,000 euros/km\textsuperscript{2}/yr for the mudflats in the upwelling large marine ecosystem and 60,000 euros /km\textsuperscript{2}/yr for the beaches of the estuaries and mangrove large marine ecosystem. These have been calculated as an average of values found in the three ecoregions and are, as mentioned before, conservative values based on the minimum aggregate of use and non-use values.
situation). The unit value for coastal protection is deducted from this value in order to avoid double-counting. As a result, the value of mangroves average 40,000 euros per km² and the cost of mangrove deforestation can be estimated to be of the order of 4 million euros per year for West Africa.

8.2.3.3 Cost of policy inaction
The economic losses associated with the two considered pressures on ecosystems in the region can be valued at a minimum of 4.4 million euros. If a part of the degradation is due to the pressure of natural factors (including climate change - rising sea levels), another part is due to political inaction: 1) a lack of political considerations for biodiversity protection at regional and national level; 2) a policy of laissez-faire that characterizes the observed open-access to all coastal and marine resources; and 3) a lack of interest in the knowledge of ecosystem roles and related ecological and economic functions.

As a consequence, the estimated economic loss should be understood as the cost to government for failing to control exploitation and degradation of marine and coastal resources. A political commitment for the protection of marine and coastal ecosystems, through a network of MPA strengthening, would reduce erosion and deforestation and thus reduce the costs incurred. This would also help to build resilience of the coastal ecosystems, and increase adaptation to climate change, which would otherwise cause additional economic losses.

Compared to the cost of policy inaction, the implementation of mangrove management measures seems worthwhile. This can include measures to limit intensive cutting in the most sensitive zones (for fish smoking, as in Tristao), mangrove replanting programmes, the development of alternative systems for the production of salt (solar ovens for the production of salt and the abandonment of wood ovens). All of these measures could be implemented at little cost and generate significant economic benefits. The implementation of measures to limit erosion in the most affected areas could now also be considered in relation to the economic losses of beaches that disappear each year but also with respect to the risks for the coastal populations of losing their homes, and being forced to migrate and possibly lose their jobs and sources of livelihood and food security.

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57 There are various examples of erosion that caused displacement of population and loss of jobs: in Plaignment Island in Sierra Leone where migrant fishers were forced to move away from their village because the latter disappeared because of erosion. The subject of environmental
8.2.4 Economic valuation as a support for sustainable financing of MPAs
The economic valuation of ecosystems within MPAs is a prerequisite for planning their long-term financing or for identifying new sources of funding (Emerton, 1999). The economic evaluation and its development on the valuation of net benefits of conservation, helps the sustainable financing of MPAs by guiding the choice of development of sources of revenues for the MPAs. It also helps the construction of the financial strategy and related business plan. The TEV of ecosystems in the MPA highlights the ecosystems and the uses that are the major contributors to the creation of economic value. This information can therefore orientate the choice for MPA financing source. For instance, if the carbon sequestration value for mangroves proves to be important (as is the case in Cacheu MPA with 1.2 million euros), the MPA manager can seek to develop a blue carbon project for mangrove carbon capture payment from international donors through the establishment of a REDD+ mechanism. In Santa Luzia, the tourism-related activities represent nearly a quarter of the TEV of the MPA. It is therefore best for this MPA to consider a fee for tourists on excursions or diving to access the site. This tax would fund the costs associated with the management of the MPA. In most MPAs, the gross added value of the fishery is very important. Hence the implementation of licenses that grant access to certain fishing areas in the MPA could also fund part of the management of the area.

The evaluation of non-use values through choice experiment methods also provides information on the willingness-to-pay of individuals to see the ecosystems of the region protected. This can justify the implementation of a national tax for this kind of protection and more large-scale sources of funding for MPA networks. The choice experiment method can be used to set the amount of the tax for tourists for example, or the level fishing licenses for commercial fishing.

Furthermore, the calculation of the net benefits of the conservation brought by MPAs can serve as a support to develop such innovative financing mechanisms as Payments for Ecosystem Services (PES). The most widely acknowledged definition of PES was provided by Wunder (2005). He defined it as (p. 2) “a voluntary transaction by which a well-defined environmental service (or a land use likely to secure that service) is being ‘bought’ by a (minimum one) service buyer from a (minimum one) service provider and

migration of fishers on the coast of West Africa was covered in a paper published by the author in Maritime Studies Journal available online at
http://www.maritimestudiesjournal.com/content/11/1/1 (Binet et al., 2012b)

58 See briefing note on REDD+ mechanism and other carbon offset programmes in Appendix 6.
if and only if the service provider secures service provision.” The core principle underlying payment for ecosystem services is that “external ecosystem services beneficiaries make direct, contractual and conditional payments to local landholders and users in return for adopting practices that secure ecosystem conservation and restoration” (ibid). The concept of payment for ecosystem services has generated much interest as the centre of more direct biodiversity conservation approaches or avoidance of degradation (Hardner and Rice 2002; Niesten and Rice 2004; Scherr et al., 2004; Ferraro and Kiss 2002; Wunder, 2005). The implementation of such a concept has been applied to carbon sequestration in forests or soil, provision of habitats for endangered species, protection of landscapes (Ghazoul et al., 2009), or watershed protection (Landell-Mills and Porras, 2002; Gomez-Baggethun et al., 2010). Payments for ecosystem services are largely supported by both ecologists and investors, and it is seen as a promising way of reintegrating conservation within the economy. The research undertaken here can be of interest to inform the development of PES, whereas authors disagree with the use of economic valuation of ecosystems for the design of market instruments such as PES or compensation measures (Karsenty and Ezzine de Blas, 2014; Guingand et al., 2014). However, when the object measured by the valuation exercise is the benefits brought by ecosystem conservation and not the ecosystem services as such, the economic valuation can be useful. Hence PES is supposed to remunerate the excess services brought by the actions of the “service provider” and that is precisely what has been measured within this thesis.

As part of this research, the author has investigated the fisheries agreement between Mauritania and the European Union and the financial compensation for fishing paid by the EU (Binet et al., 2013a)\(^5\). One part of this compensation has been dedicated to finance the National Park of Banc d’Arguin. One could see this contribution to the conservation of Banc d’Arguin National Park’s ecosystems as payment for the biomass production of commercial species exploited by its fleet, which could be defined as a PES.

In this specific case, the valuation of net benefits brought by the conservation within the National Park is of particular interest because it can suggest a value to the payment (compared to the value brought by conservation and the subsequent increase in fisheries biomass productivity for species of EU interest) as well as the negotiation for

\(^5\)This research was carried out in Mauritania and led to the preparation of an article submitted to and accepted for publication by Global Environmental Change. The draft paper can be consulted upon request.
the financing of the Park. It can also justify the dedication of one part of the compensation to the Park. As a consequence, the method developed in this research can be of great interest to develop innovative conservation financing mechanisms.

8.3 Conclusion of the chapter
This chapter has dealt with the public policy objectives of the economic valuation exercise. It has presented the various approaches to the economics from a decision-making perspective, from the ‘economics of degradation’ to the ‘economics of welfare’ and the ‘economics of protection’. It has detailed the various uses of the valuation studies for policy-making going further. It has identified some extensions to my research in preparing policy tools that integrate valuation studies results, for management purposes (comparison of benefits of conservation and costs of management an indirect use values-driven management), for advocacy purposes (the evaluation of the costs of policy inaction, the consideration of indirect use values), and for the development of financing mechanisms (identification of park entrance fees, payments for ecosystem services).
9 Conclusions

The two research questions asked as part of this study included the ways to measure the value of marine conservation and how these could help decision-making in the field of conservation. It stemmed from the recognition that few studies of the economic valuation of ecosystems had aimed to value the benefits of conservation, beyond the value of ecosystems themselves. I have aimed to address these questions through the development of a specific method for the valuation of conservation and the application to a case study in a selection of 5 Marine Protected Areas in West Africa. I have also aimed to explore the application of such valuation for decision-making uses (from management of protected areas to advocacy or development of financing mechanisms).

This concluding chapter first gathers the main conclusions of my study, both factual and conceptual. It then identifies the contribution to knowledge that this research has provided. Last, it gives an agenda for further research.

9.1 Key conclusions to the study

The valuation exercise proposed in this study stems from the Total Economic Value concept and the available tools for economic valuation of ecosystems (from direct market pricing to the stated and revealed preference methods). The valuation study applied in five West African MPAs led to the estimation of the Total Economic Value of the marine and coastal ecosystems in these MPAs. Overall, this first evaluation of the ecosystems proves that market uses (both extractive such as fisheries and wood cutting, and non-extractive such as tourism) are not the only activities that create value along the coast. Commercial fisheries are the third most important value, after water and waste treatment and coastal protection, and before carbon sequestration and fisheries biomass production. The results thus highlights the predominance of indirect use values on the direct use values, which are the only values that have been assessed in the past valuation studies. Most famous ecosystems of West Africa (mangroves, estuaries) are not the only valuable ecosystems: seagrass meadows, beaches, mudflats are less-known ecosystems, but very much valuable when considered as unit values (values per km² are most often larger for these ecosystems than for mangroves and estuaries). Also, they show that the health status of ecosystems have a critical importance on the delivery of the indirect uses and, subsequently, their values. Non-use values also substantially contribute to the creation of value, highlighting one part of the TEV that is most often overlooked and very rarely considered in economic valuation studies worldwide. Through the evaluation of the TEV of ecosystems, these results
provide a first element on the valuation of the benefits brought by conservation. Also, this evaluation contributes to better knowledge of the ecosystems in the MPAs.

Furthermore, based on the theoretical background of the TEV, a method was developed to value, not the ecosystems themselves, but the benefits brought by the protection of these ecosystems. This method is based on the comparison of a protected ecosystem (among seven West African marine ecosystems – estuaries and channels, seagrass meadows, mangroves, beaches, mudflats, rocky bottoms, coral bottoms) with an equivalent unprotected ecosystem nearby that presents similar (as much as possible) ecological and socioeconomic characteristics to the unprotected ecosystem. The estimation of unitary values of ecosystems for protected and unprotected sites enables a comparison of the value of the ecosystems within the MPA with the value of the same surface area of ecosystems in an unprotected site. This comparison leads to the calculation of the net benefits of the protection by the MPA. I have found that the few MPAs that have enforced their management plan and reduced the threats to the ecosystems have improved the health status of their ecosystems, thus creating net benefits to the MPA, when compared to unprotected comparison site. For MPAs that have no management plan (‘paper MPA’), the comparison shows a deficit of value for MPAs when compared to unprotected sites. These evaluations provide a key conclusion about the necessity to enforce conservation measures in order to expect benefits to the protection policy, when compared to unprotected sites.

Along with the calculation of values and benefits, the study has unveiled key information and data on the various direct uses and associated practices. It has also provided a review of available ecological and socioeconomic information on the MPA and, if not available, on the West African ecoregions. Further, this study has outlined the knowledge about functions and services of ecosystems by the local and visiting populations, the perception of health status and the pressures that threatens the ecosystems. These pieces of information have provided the necessary socio-cultural context to my study. My valuation study can thus be viewed as an integrated, multi-actor assessment tool that brings together knowledge from different disciplines – ecology, biology, economics and social sciences – and expresses it in a monetary form that is intelligible.

As I demonstrated in the fifth chapter, assigning value to biodiversity on the one hand, and to biodiversity conservation on the other hand are the first steps towards the development of decision-making support in the field of conservation. Economic
valuation can therefore have one of the three specific approaches: the economics of degradation; the economics of welfare; and the economics of protection. My study has aimed to address this last approach and further the objectives pursued include: MPA management; advocacy for conservation; and the development of sustainable financing mechanisms. In the same chapter, I have aimed to develop these objectives through a specific conceptual approach that includes: a cost-benefit analysis of conservation in MPA, the consideration of indirect use values as guidance for management; the evaluation of the costs of policy inaction, the development of financing mechanisms.

9.2 Contribution to knowledge
As I presented in Chapter 2, the economic valuation of ecosystems has rapidly progressed over the past decades in developing new methods. However, it seems to have reached a methodological cul-de-sac where the TEV has been criticised for its limitations and its failure to progress towards more accurate, cost-effective, policy-oriented research. Even the core principle of valuation is questioned, since studies tend to show that the more humans exploit an ecosystem, the more its economic value increases, boosted by direct use values. The research presented in this study has aimed to build-up on the economic valuation background and advance the knowledge in this field. This is detailed by the four following paragraphs.

First, the research has not only focused on direct and indirect use values, but also on non-use values, which are most often overlooked although recognised as a substantial part to the TEV (since 1967 and Krutilla’s work: more than 45 years ago). This is, to my knowledge, among the first attempts to estimate non-use values in African coastal and marine ecosystems, and among the first to use choice experiment worldwide (most non-use valuation studies use contingent valuation method). The method used to estimate non-use values through choice experiment has also evolved during the time of research and an improved method is proposed in Appendix 7. It has also identified new and emerging issues worthy of investigation in discussing the question of spatial and time scale to the non-use values, as well as the importance of the socio-cultural context.

Second, a review of the literature on valuation studies clearly highlights the confusion of valuation of ecosystems and valuation of conservation. It is a common mistake found

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60 This method has also been subject to publications and oral papers (Binet et al., 2013a; Binet et al., 2014).
in the literature that the value of ecosystems is similar to the value of conservation (Pagiola et al., 2004). This confusion between the value of ecosystems and the value of conservation demonstrates a clear misunderstanding of the object valued. On the one hand, the ecosystems are the object of the valuation, while, on the other hand, it is the protection policy that is valued. In one case, it is the absolute value that is measured (that computes all initial assets and the results of successive policies enforced in this area). In the other case, it is one specific policy that is measured. While interconnected, these values are distinct. This was confirmed by Pagiola et al. (2004, p. 23): "by using the entire flow of benefits as a yardstick for policy decisions, we are implicitly assuming that doing nothing would result in the complete and instantaneous loss of all ecosystem services, and that conversely conservation would result in the complete and instantaneous halt of all degradation processes. Neither assumption is realistic.”

Through this study, I have aimed to delimit the boundaries of each of the ecosystem and conservation values. My case study also illustrates how each can be used for decision-making purposes.

Third, the valuation of conservation benefits using a TEV approach remained, to my knowledge again, essentially theoretical (see section 2.4). This research has designed and applied new field instruments in a constrained context of research. Hence the valuation process had to fit the short period allowed for the project and could only be deployed over a three years window, which is inconsistent with the necessary time to measure the benefits of biodiversity conservation (most likely 10 to 15 years). The method proposed to compare protected and unprotected sites simultaneously was quite innovative in that it addressed these constraints and the objective of the research. From the literature review I carried out, such simultaneous comparison for conservation valuation purposes has never been applied in the field of economic valuation.

Fourth, the research was particularly of interest to the West African biodiversity experts, MPA managers and national decision-makers not only for the economic valuation results but for the synthesis the research provided. It is hence among the few studies that gathers within the same document ecological information (on ecoregions, key ecosystems, ecosystem functions and health statuses of ecosystems), geographical information (calculation of ecosystem surface areas in MPAs and proportion of each ecosystem within MPAs), socioeconomic information (economic activities per ecosystem, income and production), cultural information (perception and knowledge of ecosystems and threats to ecosystem maintenance). Hence, the management plans of
MPAs can gather such information. Also, the work by Cormier-Salem (1995) has provided a meaningful transdisciplinary review of knowledge on the mangrove ecosystems in West Africa. But, apart from these works, such integrated research that combines transdisciplinary research is still lacking.

9.3 Agenda for further research

Since the Millennium Ecosystem Assessment (2005), experts in biodiversity have aimed to develop a normative tool for the identification of benefits of ecosystems to human. Later, with the TEEB initiative, environmental economists aimed to follow-up this approach and build-up a normative tool for the economic valuation of ecosystems and its use in decision-making. However, this normative objective proved more difficult to design than expected. The first barrier was the limitations of available methods to value ecosystems (see Table 2-2) and their ability to supply accurate data. On large scales, values are often astronomically high: consequently, they are hard to compare with economic reality or to integrate in a national accounting system. The second barrier is the importance of context that prevents valuation studies from being generalised at a broader level or being transferable to other ecological and socioeconomic contexts.

Thus, TEEB has switched its objective from the preparation of a guide to value all ecosystems of the world to the development of multiple national and regional, sectoral-specific, ecosystem-specific studies (many national TEEB studies, TEEB for agriculture and food, TEEB for water and wetlands, TEEB for oceans). Also, the expectations of beneficiaries of these valuation studies are wide-ranging, which have forced the TEEB authors to diversify their outcomes to better fit the various targeted beneficiaries (TEEB for national and international policy-makers, TEEB for regional and local policy-makers, TEEB for citizens, TEEB for business and enterprises) and with various sectoral or biomes approaches). This normative objective was abandoned toward an umbrella initiative that encompasses many studies and TEEB is now valued for being a meta-data platform on the economic valuation.

61 An interesting outcome of this change was illustrated during the preparation of the TEEB flagship publication (TEEB, 2010), in which the author took part (on specific biome - coastal systems, mangroves, coral reefs, polar and mountain ecosystems). The review of all valuation studies by biome (e.g. coastal wetlands, forests), which was considered central to the publication, was eventually put in annexes, its content shortened and conclusions revised. Wisely, the research team considered the results of this review as so diversified that a generalisation was very difficult and the transfer of results very uncertain.
The next challenge for economic valuation lies in overcoming this services-based approach (or uses-based approach) – so constraining in many ways – and developing an approach based directly on ecosystem functions and their interactions. This calls for an inventory of knowledge from the many disciplines involved in the economic valuation, one that establishes connections between disciplines. Further research on the economic valuation of biodiversity should aim to take into account the high complexity and incomplete knowledge that apply to biodiversity and ecosystem management. As noted by Ostrom and Parks in McGinnis (1999, p. 284), “the more social scientists preach the need for simple solutions to complex problems, the more harm we can potentially cause in the world”. Economic valuation should follow that route, in order not to be too exposed to critics of over-simplification for modelling purposes.

More specifically, more research on the evaluation methods are required, which include the following: i) further development to the choice experiment method; ii) more standardisation of the benefit transfer method; iii) more accurate evaluation of non-use values; iv) a better consideration for time and space in the valuation studies; iv) more proficient methods for the valuation of conservation.

Beyond questions of methods, further work could also be undertaken on how to better integrate valuations into practical decision-making, making them more relevant and useful for policy-makers. This includes the development of communication tools which are able to transfer the messages from the economic valuation in order to be understood by policy-makers. This also includes the design of policy-oriented economic valuation exercises that assess ex ante the policy objectives to be pursued. For instance, an economic valuation for advocacy purpose should not be similar to a valuation for management purposes. If the economic valuation field is to escape that methodological cul-de-sac, it should seek the solution in better reconnection with the political public sphere and fine-tune its approach to better address decision-making needs.
10 References


Albaret J-J. et Simier M. (2006) Suivi biologique des peuplements de poissons d’une aire protégée en zone de mangrove : le bolon de Bamboung (Sine Saloum, Sénégal), Rapport final, IRD


Asia Forest Network (2002). *Participatory rural appraisals for community forest management: tools and techniques*. Asia Forest Network, Santa Barbara, CA.


Burke, L., Selig, E & M. Spalding (2002). Reefs at Risk in Southeast Asia. World Resources Institute, Washington, DC.


Cormier-Salem, 2011. Personal communication.


Wells S, Ravilious C, & E. Corcoran (2006). *In the front line; shoreline protection and other ecosystem services from mangroves and coral reefs*. UNEP.


11 Appendices

11.1 Appendix 1: MPAs and their related comparison areas
The following sections provide geographical information on the selected MPAs (red boxes) and their related comparison sites (yellow boxes).

Langue de Barbarie MPA (Senegal)

NB: This figure does not include the breach that opened in 2003 on the sandbar (this breach is located in the middle of the CA and has had a substantial influence on both the MPA and CA).
Urok MPA and comparison area (Guinea-Bissau)
Rio Cacheu MPA and comparison site (Guinea-Bissau)
Tristao MPA and comparison site

Santa Luzia MPA and comparison site
11.2 Appendix 2: Geographical distribution of ecosystems in MPA and CA

Senegal

Figure 11-1: Ecosystems of the Langue de Barbarie MPA and its CA Saint-Louis/« secteur de la Brêche »
Figure 11-214: Ecosystems of the Rio Cacheu MPA and its CA Rio Cacine
Figure 11-3: Ecosystems of the Urok MPA and Galinas Island CA
Guinea

Figure 11-4: Ecosystems of the Tristao MPA and Rio Nunez CA
Figure 11-5: Ecosystems of the Santa Luzia MPA and its CA, the western part of Sao Vicente Island
11.3 Appendix 3: Ethical Review and letter of approval

FORM UPR16

Research Ethics Review Checklist

Please complete and return the form to Research Section, Quality Management Division, Academic Registry, University House, with your thesis, prior to examination.

<table>
<thead>
<tr>
<th>Postgraduate Research Student (PGRS) Inform</th>
<th>Student ID</th>
<th>672682</th>
</tr>
</thead>
<tbody>
<tr>
<td>Student Name</td>
<td>Thomas BINET</td>
<td></td>
</tr>
<tr>
<td>Department:</td>
<td>Economics</td>
<td></td>
</tr>
<tr>
<td>First Supervisor</td>
<td>Pr. Andy Thorpe</td>
<td></td>
</tr>
<tr>
<td>Start Date:</td>
<td>10/2012</td>
<td></td>
</tr>
<tr>
<td>(or progression date for Prof Doc students)</td>
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<table>
<thead>
<tr>
<th>Study Mode and Route</th>
<th>Part-time</th>
<th>MPhil</th>
<th>Integrated Doctorate</th>
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<td>❑</td>
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<tr>
<td>Full-time</td>
<td>❑</td>
<td>❑</td>
<td>❑</td>
</tr>
</tbody>
</table>

| Title of Thesis | VALUING NET BENEFITS OF BIODIVERSITY CONSERVATION IN WEST AFRICAN MARINE PROTECTED AREAS: A CASE FOR BETTER DECISION-MAKING |
If you are unsure about any of the following, please contact the local representative on your Faculty Ethics Committee for advice. Please note that it is your responsibility to follow the University’s Ethics Policy and any relevant University, academic or professional guidelines in the conduct of your study.

Although the Ethics Committee may have given your study a favourable opinion, the final responsibility for the ethical conduct of this work lies with the researcher(s).

11.3.1

**UKRIO Finished Research Checklist:**

(If you would like to know more about the checklist, please see your Faculty or Departmental Ethics Committee rep or see the online version of the full checklist at: [http://www.ukrio.org/what-we-do/code-of-practice-for-research/](http://www.ukrio.org/what-we-do/code-of-practice-for-research/))

<p>| | |</p>
<table>
<thead>
<tr>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td>a)</td>
<td>Have all of your research and findings been reported accurately, honestly and within a reasonable time frame?</td>
</tr>
<tr>
<td>b)</td>
<td>Have all contributions to knowledge been acknowledged</td>
</tr>
<tr>
<td>c)</td>
<td>Have you complied with all agreements relating to intellectual property, publication and authorship?</td>
</tr>
<tr>
<td>d)</td>
<td>Has your research data been retained in a secure and accessible form and will it remain so for the required duration?</td>
</tr>
<tr>
<td>e)</td>
<td>Does your research comply with all legal, ethical, and contractual requirements?</td>
</tr>
</tbody>
</table>
**Student Statement:**

I have considered the ethical dimensions of the above named research project, and have successfully obtained the necessary ethical approval(s)

<table>
<thead>
<tr>
<th>Ethical review number(s) from Faculty Ethics Committee (or from NRES/SCREC):</th>
<th>E231</th>
</tr>
</thead>
<tbody>
<tr>
<td>Signed:</td>
<td>Date: 25/09/2013</td>
</tr>
</tbody>
</table>

If you have *not* submitted your work for ethical review, and/or you have answered ‘No’ to one or more of questions a) to e), please explain why this is so:

| N/A |
|---|---|
| Signed: | Date: 23/09/2013 |
1. What are the objectives of the research project?

The objectives of the project were to carry out an economic valuation of marine ecosystems in a sample of marine protected areas in West Africa. The research included: interviews with national experts about socioeconomic data on economic activities in surveyed sites; interviews with local population with the help of a questionnaire.

2. Does the research involve NHS patients, resources or staff? YES / NO (please circle).

3. Do you intend to collect primary data from human subjects or data that are identifiable with individuals? (This includes, for example, questionnaires and interviews.) YES / NO (please circle).

4. What is the purpose of the primary data in the dissertation / research project?

The purpose of the primary data collection in research project was to be able to value marine ecosystems in studied sites. Primary data consisted in information on economic activities carried out in studied sites on the one hand, and answers to questionnaires by local populations on the other hand.

5. What is/are the survey population(s)?

The population surveyed was the local population who lives in the studied sites in four countries: Senegal, Guinea, Guinea-Bissau, Cape Verde.
6. How big is the *sample* for each of the survey populations and how was this sample arrived at?

The surveyed population by site was 250 persons.

7. How will respondents be *selected and recruited*?

The respondents were selected on a random basis. The sampling method was to select all persons above 15 years old in one house every three successive house in streets of the various villages within the studied site. Since no precise socioeconomic information were available by village in these sites, this selection method was the best available. It is recognized as creating the most representative samples of populations in data-poor situations. The selection also considered surveying at different hours during the day. Survey also considered seasonal migration of population in and out the site in order to be representative (e.g. seasonal migrant fishers).

The survey was carried out in all streets of the village in all villages of the selected site. If the population was above one third of the sampling size, then survey was carried out on one street every three street, but still in all villages (except for the most remote small villages which were difficult to reach).

For each person selected for survey, a short introduction to the questionnaire was made (content, objective, anonymous feature, etc.). Then the surveyor asked the respondent if he/she was willing to participate. If answer was yes, then the survey could be carried out.

8. What steps are proposed to ensure that the requirements of *informed consent* will be met for those taking part in the research? If an Information Sheet for participants is to be used, please attach it to this form. If not, please explain how you will be able to demonstrate that informed consent has been gained from participants.

The introduction to the question consisted in a presentation of the questionnaire, its objective, precision about anonymous character of the questionnaire. The surveyor then asked the person interviewed whether he/she is voluntary to take part to this survey. No written consent were judged necessary in this survey since the survey was totally anonymous. Also, since survey was carried out over a large proportion of the total population in some places, a written consent could have jeopardized the representativeness of sample because of high proportion of refusal to answer the questionnaire.

9. How will *data* be *collected* from each of the sample groups?

The data collection was managed by Thomas Binet. Data were collected by Thomas Binet assisted with two to four research assistants recruited in national universities. Research assistants were specifically trained for this survey. They were given instructions for selection of respondents. During the survey, Thomas Binet ensured that the surveyed population was representative of the total population of the site for specific criteria including sex ratio and estimated populations of villages.

10. How will *data* be *stored* and what will happen to the data at the end of the research?
The data were gathered by Thomas Binet and stored in a database in excel format on his personal computer and not circulated. Data are accessed via a password on document. They will be stored until all publications are completed. There have been backups of database carried out. These will be deleted once all publications are completed.

11. How will confidentiality be assured for respondents?

Confidentiality of the respondent was ensured by face-to-face interviews with surveyors. The surveyors were recruited in capital cities of the country and attention was born in case surveyors had families or any other relationships within the studied sites.

12. What steps are proposed to safeguard the anonymity of the respondents?

The questionnaire never asked respondents to provide information that would enable to identify them: there was not any question about name or address or telephone number, etc.

13. Are there any risks (physical or other, including reputational) to respondents that may result from taking part in this research? ☐ / NO (please circle).

14. Are there any risks (physical or other, including reputational) to the researcher or to the University that may result from conducting this research? YES / ☐ (please circle).

There were physical risks to the researcher during the field missions. In order to prepare for these, risks assessments were prepared for each of field mission in every country the researcher visited. Risk assessment preparation helped the researcher to be prepared to any risk associated to field survey in the countries of study (be it tropical diseases, political problems, any kind of insecurity, protection against sun, etc.). About surveyors, they were recruited based on their experience of field mission, in order to limit the risks associated to their presence on the field.

15. Will any data be obtained from a company or other organisation. YES ☐ (please circle) For example, information provided by an employer or its employees.

16. What steps are proposed to ensure that the requirements of informed consent will be met for that organisation? How will confidentiality be assured for the organisation?

The information obtained from company was collected during face-to-face semi-directive interviews with managers or company representatives. Data collection only focused on economic information without any question that could enable to identify the company (company name, registered number, director’s name, etc.). Questions focused on general economic information (production means, average volume of production, details about labour forces, etc.)
17. Does the organisation have its own ethics procedure relating to the research you intend to carry out? YES / NO (please circle).

18. Will the proposed research involve any of the following (please put a √ next to ‘yes’ or ‘no’; consult your supervisor if you are unsure):

- Vulnerable groups (e.g. children)? YES √ NO  
- Particularly sensitive topics? YES √ NO  
- Access to respondents via ‘gatekeepers’? YES √ NO  
- Use of deception? YES √ NO  
- Access to confidential personal data? YES √ NO  
- Psychological stress, anxiety etc.? YES √ NO  
- Intrusive interventions? YES √ NO  

19. Are there any other ethical issues that may arise from the proposed research?

No
INTRODUCTION (PRESENTED BY SURVEYOR)

Presentation of research, objective of research, definition of words given in questionnaire (ecosystems, biodiversity), and delimitation of site/region considered in the questionnaire

Explanations about content of questionnaire and that answers to questionnaire are anonymous

GENERAL INFORMATION

Q1 Are you resident or visitor to the site?
Resident: ☐ Visitor: ☐

Q2 (Resident) what is your usual place of residence?
Village: ______

Q3 (Resident) How long have you been in the region?
Number of years: __

Q4 (Visitor) What is your country of origin?
Europe: ☐ (country: )
Sub-region: ☐ (country: )
Other: ☐ (country: )

Q5 (Visitor) How long will you stay in the region?
Number of days: ___

Q6 (Visitor) How often have you visited the region?
Number of stays: ___

(Visitor) Independently from familial reasons, what are the reasons why you stay in the region?

<table>
<thead>
<tr>
<th>Reasons</th>
<th>Not at all</th>
<th>A little</th>
<th>Somewhat</th>
<th>A lot</th>
<th>Very much</th>
</tr>
</thead>
<tbody>
<tr>
<td>Tropical weather</td>
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<td>☐</td>
<td>☐</td>
<td>☑</td>
<td>☑</td>
</tr>
<tr>
<td>Sceneries, fauna and flora</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☑</td>
<td>☑</td>
</tr>
<tr>
<td>Culture</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☑</td>
<td>☑</td>
</tr>
<tr>
<td>Beaches</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
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<td>☑</td>
</tr>
<tr>
<td>Marine life</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☑</td>
<td>☑</td>
</tr>
<tr>
<td><em>Art de vivre</em></td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☑</td>
<td>☑</td>
</tr>
<tr>
<td>Cost of living</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☑</td>
<td>☑</td>
</tr>
<tr>
<td>Other (precise: )</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☑</td>
<td>☑</td>
</tr>
</tbody>
</table>
GENERAL KNOWLEDGE ABOUT COASTAL AND MARINE ECOSYSTEMS OF WEST AFRICA

Q7 The ecosystems of West Africa have been introduced to you during introduction to this questionnaire. Could you now try and identify these on the following pictures?

Picture 1  Picture 2  Picture 3

Picture 4  Picture 5  Picture 6

Picture 7

Picture number: Estuary and channel:

Mudflats:

Beaches:

Mangroves:

Seagrass meadows:

Rocky bottoms:

Coral bottoms:

Q8 Do you know any cultural or religious traditions associated to the marine ecosystems in the region?
Yes: ☐ No: ☐

Q9 (If answer «Yes» to Q8, What are they?)
Free answer:

Q10 (If answer «Yes» to Q8 For which ecosystems?)
Free answer:

Q11 (If answer «Yes» to Q8 are they alive and still practiced?)
Yes: ☐ No: ☐

Q12 According to you, are the following sentences right or wrong?

295
The ecosystems of the region are living

Mangroves, beaches and seagrass meadows protect our coasts (from waves, storms and floods)

Mangroves do not suffer from wood cutting

Marine ecosystems have a strong resilience (capacity to recover from important damages)

Mangroves take part to water treatment

Mangroves, seagrass and mudflats are important nursery sites for various species of fish and crustaceans

Seagrass meadows are places where one can find only marine grass

Fish living offshore do not depend on coastal ecosystems for their survival

Q13 According to you, what are the most damaging factors to marine and coastal ecosystems?
Natural factors: □ Human factors: □

Q14 Among human factors, what do you think are the most damaging to marine ecosystems in the region?

<table>
<thead>
<tr>
<th>Factor/degree of importance</th>
<th>Not important</th>
<th>Slightly important</th>
<th>Important</th>
<th>Very important</th>
</tr>
</thead>
<tbody>
<tr>
<td>Mangrove wood-cutting</td>
<td>□</td>
<td>□</td>
<td>□</td>
<td>□</td>
</tr>
<tr>
<td>Pollution (industrial and domestic.)</td>
<td>□</td>
<td>□</td>
<td>□</td>
<td>□</td>
</tr>
<tr>
<td>Destructive fishing practices and overfishing</td>
<td>□</td>
<td>□</td>
<td>□</td>
<td>□</td>
</tr>
<tr>
<td>Tourism over-frequentation (sport fishing, boat tours, etc.)</td>
<td>□</td>
<td>□</td>
<td>□</td>
<td>□</td>
</tr>
<tr>
<td>House building on the shore, on the beach</td>
<td>□</td>
<td>□</td>
<td>□</td>
<td>□</td>
</tr>
<tr>
<td>Industrial exploitation (dredging, mining for oil and gas)</td>
<td>□</td>
<td>□</td>
<td>□</td>
<td>□</td>
</tr>
<tr>
<td>Other: precise...</td>
<td>□</td>
<td>□</td>
<td>□</td>
<td>□</td>
</tr>
</tbody>
</table>
**PROTECTION OF MARINE AND COASTAL ECOSYSTEMS**

Q15 Do you think that estuaries and channels, mangroves, seagrass, beaches, mudflats, rocky and coral bottoms are in danger?

Yes: ☐  No: ☐

Q16 For what reasons and to what extent?

<table>
<thead>
<tr>
<th>Reasons/importance</th>
<th>Not at all</th>
<th>Slightly important</th>
<th>Somewhat important</th>
<th>Important</th>
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</tr>
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<tbody>
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<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
</tr>
<tr>
<td>Natural factors</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
</tr>
<tr>
<td>Lack of knowledge on the importance of marine life to local populations</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
</tr>
<tr>
<td>Lack of environmental concern among local population</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
</tr>
<tr>
<td>Lack of regulatory framework</td>
<td>☐</td>
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<td>☐</td>
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<tr>
<td>Lack of political support</td>
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</tr>
<tr>
<td>Other (precise: )</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
</tr>
</tbody>
</table>

Q17 Do you think marine ecosystems of the region could one day totally disappear?

Yes: ☐  No: ☐

Q18 (If «yes» to Q17) Would their disappearance be a problem to you?

Yes: ☐  No: ☐

Q19 Before answering this questionnaire, were you aware of the situation of ecosystem in the region?

Yes: ☐  No: ☐

Q20 Have you heard of the protection of ecosystems carried out in the region?

Yes: ☐  No: ☐

Q21 By which media have you heard about it?

<table>
<thead>
<tr>
<th>Media</th>
<th>Yes</th>
<th>No</th>
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<tbody>
<tr>
<td>TV</td>
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</tr>
<tr>
<td>Radio</td>
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</table>

<table>
<thead>
<tr>
<th>Media</th>
</tr>
</thead>
<tbody>
<tr>
<td>Meetings</td>
</tr>
<tr>
<td>Word of mouth</td>
</tr>
</tbody>
</table>
MANAGEMENT OPTIONS

We would now like to invite you to select the scenario that you prefer with regards to ecosystem management, among the 9 following scenarios. Only one choice is possible

Please refer to the presentation booklet

Scenario number:

Q22 Which criteria have you first considered in priority to make your choice? Please classify according to degree of importance, 1 being the most important, 4 the least important:

<table>
<thead>
<tr>
<th>Criteria</th>
<th>1</th>
<th>2</th>
<th>3</th>
<th>4</th>
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<tr>
<td>Terrestrial activities</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Marine activities</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Biodiversity</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Cost</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

If you chose a scenario with a cost of zero, what is the reason for such choice?
Free answer:
Q23  For each of the following terrestrial and marine activities, what is according to you their impact on ecosystem health status? From 1 to 5 for each activity, 1= no impact and 5= strong impact  
NB : please refer to the presentation booklet

<table>
<thead>
<tr>
<th>Terrestrial activity</th>
<th>N (1-5)</th>
<th>Marine activity</th>
<th>N (1-5)</th>
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</thead>
<tbody>
<tr>
<td>Wood-cutting</td>
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<td>Commercial fishing</td>
<td></td>
</tr>
<tr>
<td>Pollution</td>
<td></td>
<td>Subsistence fishing</td>
<td></td>
</tr>
<tr>
<td>Coastal construction</td>
<td></td>
<td>Sport fishing and other touristic activities at sea</td>
<td></td>
</tr>
<tr>
<td>Tourism activities</td>
<td></td>
<td>Industrial exploitation</td>
<td></td>
</tr>
</tbody>
</table>

Q24  What do you think of the participation of local and national institutions for the management of marine ecosystems in the region?  
Enough: ☐  Not enough: ☐  
Do not know: ☐  Other (precise : ) : ☐

Q25  Do you think that residents and visitors should be more involved in the management of marine ecosystems in the region?  
Yes: ☐  No: ☐  Do not know: ☐

DIRECT USES

Q26  What are your uses (and frequency) of estuaries and channels of the region?

<table>
<thead>
<tr>
<th>Use / frequency</th>
<th>Never</th>
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<th>Once a month</th>
<th>Once a week</th>
<th>Almost every day</th>
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</thead>
<tbody>
<tr>
<td>Subsistence fishing</td>
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<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
</tr>
<tr>
<td>Commercial fishing</td>
<td>☐</td>
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<td>☐</td>
</tr>
<tr>
<td>Sport fishing</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
</tr>
<tr>
<td>Swimming and bathing</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
</tr>
<tr>
<td>Boat tours, ecotourism</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
</tr>
<tr>
<td>Nautical activities</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
</tr>
<tr>
<td>Medical use</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
</tr>
<tr>
<td>Other (precise : )</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
</tr>
</tbody>
</table>

Q27  What are your uses (and frequency) of mangroves of the region?
<table>
<thead>
<tr>
<th>Use / frequency</th>
<th>Never</th>
<th>Once a year</th>
<th>Once a month</th>
<th>Once a week</th>
<th>Almost every day</th>
</tr>
</thead>
<tbody>
<tr>
<td>Subsistence fishing</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
</tr>
<tr>
<td>Commercial fishing</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
</tr>
<tr>
<td>Sport fishing</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
</tr>
<tr>
<td>Boat tour</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
</tr>
<tr>
<td>Observation of fauna and flora</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
</tr>
<tr>
<td>Wood cutting</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
</tr>
<tr>
<td>Hunting</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
</tr>
<tr>
<td>Medical use</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
</tr>
<tr>
<td>Other (precise : )</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
</tr>
</tbody>
</table>

Q28 What are your uses (and frequency) of seagrass meadows of the region?

<table>
<thead>
<tr>
<th>Use / frequency</th>
<th>Never</th>
<th>Once a year</th>
<th>Once a month</th>
<th>Once a week</th>
<th>Almost every day</th>
</tr>
</thead>
<tbody>
<tr>
<td>Subsistence fishing</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
</tr>
<tr>
<td>Commercial fishing</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
</tr>
<tr>
<td>Sport fishing</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
</tr>
<tr>
<td>Spearfishing</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
</tr>
<tr>
<td>Diving</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
</tr>
<tr>
<td>Swimming and bathing</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
</tr>
<tr>
<td>Boat tours, ecotourism</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
</tr>
<tr>
<td>Observation of fauna and flora</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
</tr>
<tr>
<td>Medical use</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
</tr>
<tr>
<td>Other (precise : )</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
</tr>
</tbody>
</table>
Q29  What are your uses (and frequency) of beaches of the region?

<table>
<thead>
<tr>
<th>Use / frequency</th>
<th>Never</th>
<th>Once a year</th>
<th>Once a month</th>
<th>Once a week</th>
<th>Almost every day</th>
</tr>
</thead>
<tbody>
<tr>
<td>Subsistence fishing</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Commercial fishing</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Sport fishing</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Spearfishing</td>
<td></td>
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<td></td>
<td></td>
</tr>
<tr>
<td>Diving</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Swimming and bathing</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Boat tours, ecotourism</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Medical use</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Other (precise: )</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

Q30  What are your uses (and frequency) of mudflats of the region?

<table>
<thead>
<tr>
<th>Use / frequency</th>
<th>Never</th>
<th>Once a year</th>
<th>Once a month</th>
<th>Once a week</th>
<th>Almost every day</th>
</tr>
</thead>
<tbody>
<tr>
<td>Subsistence fishing</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Commercial fishing</td>
<td></td>
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<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Sport fishing</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Boat tours, ecotourism</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Observation of fauna and flora</td>
<td></td>
<td></td>
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<td></td>
<td></td>
</tr>
<tr>
<td>Medical use</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Other (precise: )</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>
Q31 What are your uses (and frequency) of rocky bottoms of the region?

<table>
<thead>
<tr>
<th>Use / frequency</th>
<th>Never</th>
<th>Once a year</th>
<th>Once a month</th>
<th>Once a week</th>
<th>Almost every day</th>
</tr>
</thead>
<tbody>
<tr>
<td>Subsistence fishing</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
</tr>
<tr>
<td>Commercial fishing</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
</tr>
<tr>
<td>Sport fishing</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
</tr>
<tr>
<td>Spearfishing</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
</tr>
<tr>
<td>Diving</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
</tr>
<tr>
<td>Swimming and bathing</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
</tr>
<tr>
<td>Other (precise : )</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
</tr>
</tbody>
</table>

Q32 What are your uses (and frequency) of coral bottoms of the region?

<table>
<thead>
<tr>
<th>Use / frequency</th>
<th>Never</th>
<th>Once a year</th>
<th>Once a month</th>
<th>Once a week</th>
<th>Almost every day</th>
</tr>
</thead>
<tbody>
<tr>
<td>Subsistence fishing</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
</tr>
<tr>
<td>Commercial fishing</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
</tr>
<tr>
<td>Sport fishing</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
</tr>
<tr>
<td>Spearfishing</td>
<td>☐</td>
<td>☐</td>
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<td>☐</td>
<td>☐</td>
</tr>
<tr>
<td>Diving</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
</tr>
<tr>
<td>Swimming and bathing</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
</tr>
<tr>
<td>Boat tours, ecotourism</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
</tr>
<tr>
<td>Medical use</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
</tr>
<tr>
<td>Other (precise : )</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
<td>☐</td>
</tr>
</tbody>
</table>
Q33  (Visitor) How much have you spent for activities associated to marine ecosystems during your stay in the region (boat tours, entrance to a park, sport fishing, fauna and flora observation, etc.)?

<table>
<thead>
<tr>
<th>Choice</th>
</tr>
</thead>
<tbody>
<tr>
<td>Less than 10000 CFA</td>
</tr>
<tr>
<td>Between 10000 and 50000 CFA</td>
</tr>
<tr>
<td>Between 50000 and 10000 CFA</td>
</tr>
<tr>
<td>Between 100000 and 200000 CFA</td>
</tr>
<tr>
<td>More than 200000 CFA</td>
</tr>
<tr>
<td>Do not know</td>
</tr>
</tbody>
</table>

SOCIOECONOMIC INFORMATIONS

Q34  How old are you?
From 15 to less than 25 y.o.  □  From 50 to less than 65 y.o.  □
From 25 to less than 35 y.o.  □  From 65 y.o. and more  □
From 35 to less than 50 y.o.  □

Q35  Gender
Female: □  Male: □

Q36  How many persons are there in your household, including yourself?
Total number of persons: __  Number of children less than 18 y.o.: __

Q37  What is main activity?
Fisher  □
Farmer  □
Breeder  □
Civil servant  □
Worker  □
Salesman, craftsman (independent)  □
Liberal (accountant, lawyer, etc.)  □
Unemployed  □
Q38  How old were you when you stopped going to school?
Never been to school
Before 10 y.o.
12 y.o.
14 y.o.
16 y.o.
18 y.o.
More than 18 y.o.

Q39  How much do you need to live per month?
Between 0 and 3000 CFA
Between 3000 and 7000 CFA
Between 7000 and 10000 CFA
Between 10000 and 15000 CFA
Between 15000 and 30000 CFA
Between 30000 CFA and 50000 CFA
Between 50000 CFA and 100000 CFA
More than 100000 CFA
11.5 Appendix 5: Survey presentation portfolio

Illustrations: Thierry Caroff

Questionnaire introduction

The network of West African Marine Protected Areas (RAMPAO) is a regional initiative that aims to protect and value marine and coastal ecosystems of the West African region. Its priority is to develop marine protected areas able to protect these ecosystems and provide benefits to local population through this protection.

The following questionnaire is part of a study that seeks to inform local and national decision-makers about the social and economical value of marine and coastal ecosystems that compose the West African marine protected areas (MPA). It also aims to define the best management options for the future in terms of local benefits and sustainable exploitation of ecosystems. It examines the knowledge and perception of inhabitants of coastal communities.

No specific knowledge about the ecosystems is needed to respond to the questionnaire. It is not necessary to have a specific use of coastal ecosystems to answer. The most important thing here is to provide answers that really reflect your thoughts.

Also, this questionnaire is totally anonymous and confidential. Data will be used for this study only and stored safely. It will not be communicated to any other institutions.
Brief presentation of marine and coastal ecosystems

**Estuaries and channels**

**Seagrass meadows**

**Mangrove**

**Beach**

**Coral bottoms**

**Rocky bottoms**

**Mudflats**
Scenarios presentation (for residents and working visitors)

Although they are protected by an MPA, the marine and coastal ecosystems are currently in a difficult situation. They face a decrease of fish stocks, a destruction of critical habitats (such as mangroves) and various pollutions. Considering this, management measures are being designed in order to better protect ecosystems and ensure their good health. Let’s consider here several scenarios of management of the ecosystems of the region which would guarantee such good health of ecosystems and their maintenance for future generations. These scenarios are defined by 4 attributes:

1- Terrestrial activities which impact ecosystems:
Terrestrial activities may be characterized by 4 sub-activities: mangrove wood cutting along the coastline, pollution (domestic and industrial), construction on the shore; intensive tourism. The three possible scenarios are:
- 20% decrease of current terrestrial activities that impact ecosystems through their ban or limitation;
- Maintain terrestrial activities along the shoreline at their current level (status quo); and
- 20% increase of current terrestrial activities through the economic development and uses of the coastline.

2- Marine activities which impact ecosystems:
Marine activities that impact ecosystems can be divided into 4 sub-activities: 1) commercial fisheries, 2) subsistence and small-scale fisheries, 3) recreational fishing and other touristic activities (bathing, boat tours, diving, etc.) and 5) industrial exploitation (oil, mining, dredging, etc.). The three possible scenarios are: Maintain current level of limitation of marine activities that impact ecosystems;
- Ban on most destructive practices: commercial fisheries and industrial exploitation;
- Ban on all activities that impact ecosystems: all fisheries, all touristic activities, and all industrial exploitation.

3- Marine biodiversity:
The marine biodiversity is characterized by its diversity of species (fish, shellfish, vegetal, algae, etc.); richness and abundance. The three possible scenarios are:
- 20% decrease of biodiversity level: diversity and abundance of marine species (fish, shellfish, sea plants, etc.)
- Maintain current level of biodiversity (status quo)
- 20% of biodiversity level: diversity and abundance

4- Cost:
The cost is the cost associated with the realization of the scenario proposed. This cost is for each household and could be implemented by a local tax to support local activities for marine and coastal ecosystems. The three different level of tax are:
- 0CFA/household/year
- 2000 CFA/household/year
- 5000 CFA/household/year

Pictures have been prepared in order to provide you with a better understanding of the proposed scenarios for each of the 4 attributes:
The objective of this part of the questionnaire is to ask you about your preferred option for the future of the marine and coastal ecosystems. You will be asked to choose your preferred scenario from a list of 9. Your choice should be done as much as possible independently from the current uses you have of the ecosystems. Rather, you should consider the perception you have from these ecosystems and how much you would like to see them maintained now and for future generations. Here is presented an example scenario:

<table>
<thead>
<tr>
<th>Terrestrial activities that impact ecosystems</th>
<th>20% decrease</th>
<th>Maintain current level</th>
<th>20% increase</th>
<th>No regulation</th>
<th>Ban on industrial use</th>
<th>Ban on all activities</th>
</tr>
</thead>
<tbody>
<tr>
<td>Mangrove wood cutting</td>
<td><img src="https://example.com/mangrove.png" alt="Image" /></td>
<td><img src="https://example.com/keep.png" alt="Image" /></td>
<td><img src="https://example.com/increase.png" alt="Image" /></td>
<td><img src="https://example.com/no.png" alt="Image" /></td>
<td><img src="https://example.com/ban.png" alt="Image" /></td>
<td><img src="https://example.com/ban.png" alt="Image" /></td>
</tr>
<tr>
<td>Pollutions (domestic and industrial)</td>
<td><img src="https://example.com/pollution.png" alt="Image" /></td>
<td><img src="https://example.com/keep.png" alt="Image" /></td>
<td><img src="https://example.com/ban.png" alt="Image" /></td>
<td><img src="https://example.com/no.png" alt="Image" /></td>
<td><img src="https://example.com/ban.png" alt="Image" /></td>
<td><img src="https://example.com/ban.png" alt="Image" /></td>
</tr>
<tr>
<td>Construction on the shore</td>
<td><img src="https://example.com/construction.png" alt="Image" /></td>
<td><img src="https://example.com/keep.png" alt="Image" /></td>
<td><img src="https://example.com/ban.png" alt="Image" /></td>
<td><img src="https://example.com/no.png" alt="Image" /></td>
<td><img src="https://example.com/ban.png" alt="Image" /></td>
<td><img src="https://example.com/ban.png" alt="Image" /></td>
</tr>
<tr>
<td>Touristic development</td>
<td><img src="https://example.com/tourism.png" alt="Image" /></td>
<td><img src="https://example.com/keep.png" alt="Image" /></td>
<td><img src="https://example.com/ban.png" alt="Image" /></td>
<td><img src="https://example.com/no.png" alt="Image" /></td>
<td><img src="https://example.com/ban.png" alt="Image" /></td>
<td><img src="https://example.com/ban.png" alt="Image" /></td>
</tr>
</tbody>
</table>

This scenario represents a decrease of 20% of the terrestrial activities that impact ecosystems, a ban on industrial activities; an increase of 20% of biodiversity level for a cost of 0 CFA/household/year.
**Scenario choice**

The choice of scenario depends upon your preference for the future of marine and coastal ecosystems in the area, independently from the use you have of these ecosystems. **NB: look at all the scenarios before choosing one. This is only about your individual preference; there is no true or false answer in the following scenarios.**

<table>
<thead>
<tr>
<th>No.</th>
<th>Activities terrestres: augmentation 20%</th>
<th>Activities maritimes: régles sur toutes activités</th>
<th>Biodiversité: augmentation 20%</th>
<th>Coût</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td><img src="image1.png" alt="Diagram 1" /></td>
<td><img src="image2.png" alt="Diagram 2" /></td>
<td><img src="image3.png" alt="Diagram 3" /></td>
<td>0 CFA/m²/m²</td>
</tr>
<tr>
<td>2</td>
<td><img src="image4.png" alt="Diagram 4" /></td>
<td><img src="image5.png" alt="Diagram 5" /></td>
<td><img src="image6.png" alt="Diagram 6" /></td>
<td>2000 CFA/m²/m²</td>
</tr>
<tr>
<td>3</td>
<td><img src="image7.png" alt="Diagram 7" /></td>
<td><img src="image8.png" alt="Diagram 8" /></td>
<td><img src="image9.png" alt="Diagram 9" /></td>
<td>2000 CFA/m²/m²</td>
</tr>
<tr>
<td>4</td>
<td><img src="image10.png" alt="Diagram 10" /></td>
<td><img src="image11.png" alt="Diagram 11" /></td>
<td><img src="image12.png" alt="Diagram 12" /></td>
<td>0 CFA/m²/m²</td>
</tr>
<tr>
<td>5</td>
<td><img src="image13.png" alt="Diagram 13" /></td>
<td><img src="image14.png" alt="Diagram 14" /></td>
<td><img src="image15.png" alt="Diagram 15" /></td>
<td>5000 CFA/m²/m²</td>
</tr>
<tr>
<td>6</td>
<td><img src="image16.png" alt="Diagram 16" /></td>
<td><img src="image17.png" alt="Diagram 17" /></td>
<td><img src="image18.png" alt="Diagram 18" /></td>
<td>5000 CFA/m²/m²</td>
</tr>
<tr>
<td>7</td>
<td><img src="image19.png" alt="Diagram 19" /></td>
<td><img src="image20.png" alt="Diagram 20" /></td>
<td><img src="image21.png" alt="Diagram 21" /></td>
<td>0 CFA/m²/m²</td>
</tr>
<tr>
<td>8</td>
<td><img src="image22.png" alt="Diagram 22" /></td>
<td><img src="image23.png" alt="Diagram 23" /></td>
<td><img src="image24.png" alt="Diagram 24" /></td>
<td>5000 CFA/m²/m²</td>
</tr>
<tr>
<td>9</td>
<td><img src="image25.png" alt="Diagram 25" /></td>
<td><img src="image26.png" alt="Diagram 26" /></td>
<td><img src="image27.png" alt="Diagram 27" /></td>
<td>2000 CFA/m²/m²</td>
</tr>
</tbody>
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**11.6 Appendix 6: blue carbon opportunities in MPA**

This appendix is extracted from a briefing note and keynote speech prepared for a workshop held in Montenegro in 2013 on the opportunities for blue carbon development in Mediterranean MPAs.
What is blue carbon?

Blue carbon represents the carbon dioxide that is captured and stored in marine and coastal ecosystems. These ecosystems include mangrove, saltmarshes and seagrass meadows. They play a role of "carbon sinks" and thus greatly contribute to atmospheric carbon rate mitigation.

Why are Marine Protected Areas key sites for blue carbon project implementation?

Marine protected areas (MPAs) are privileged areas for implementing conservation of coastal ecosystems that enhance carbon capture. They already gather the required human resources and technical capacities to effectively implement the necessary activities of blue carbon projects: monitoring; mapping, protection measures implementation; evaluation; etc.

In most cases, MPAs also benefit from better scientific knowledge about their ecosystems. Blue carbon projects can therefore build on a pre-existing wealth of understanding. MPAs also have management measures in place that ease the implementation of blue carbon project measures, thus diminishing the costs associated with the blue carbon project: 1) transaction costs associated with the costs incurred to develop the project and implement the measures and all monitoring and evaluation activities; 2) opportunity costs associated with the reduction of practices that impact ecosystems (such as trawling, anchoring etc).

MPAs, by running blue carbon projects, could in exchange get a sustainable source of income to finance day-to-day management and investment.

What are the current opportunities for blue carbon market mechanisms worldwide?

There are two main mechanisms to pay for avoided emissions or enhancement of carbon stocks: 1) the compliance market which is based on the United Nations Framework Convention on Climate Change (UNFCCC); and 2) the voluntary markets.

From the compliance market first, the UNFCCC (and the Kyoto Protocol in 1997) have gathered over 190 countries to agree on the reduction of their carbon emissions. The Kyoto Protocol enabled countries to trade emission rights to meet their targets more easily. It has created several mechanisms to ease this trading. First, the Clean Development Mechanism (CDM) enables developing countries to voluntarily undertake greenhouse gases (GHG) reduction projects and generate carbon credits that could be marketed to developed countries. This created the first global market for carbon. In this context, blue carbon has unfortunately not been covered. At the time of the renewal of the Kyoto Protocol (under the Cancun Agreement), it is still uncertain whether blue carbon will be taken forward because of the lack of political support for the Agreement.

Another important mechanism of the UNFCCC system is the "Reducing Emissions from Deforestation and forest Degradation conserving forest carbon stocks, sustainably managing forests and enhancing forest carbon stocks" (REDD+). Mangroves have recently been included as part of this and a few projects have developed that get carbon credits for mangrove restoration and avoidance of degradation. However, REDD+ apply to the general term of 'forests' which mangroves fall outside of, as do other coastal ecosystems such as seagrass, saltmarshes, wetlands, etc. One additional constraint to blue carbon through REDD+ mechanism is the fact that REDD+ only considers carbon...
stored above ground (and not underground) for which mangrove may provide substantial storage, but not seagrass which store most of the carbon in the soil.

As part of the compliance markets again, blue carbon projects could be developed under the CDM afforestation/reforestation (A/R) projects. Methodologies for small-scale and large-scale projects have been validated for mangroves but not for the other coastal ecosystems considered within blue carbon. Also, these A/R projects only consider restoration activities which are much easier for mangroves than for seagrass.

One good prospect for blue carbon ecosystems that are not recognised as forests (e.g. seagrass meadows) is the development of blue carbon projects as nationally appropriate mitigation actions (NAMAs). The NAMAs correspond to the key actions to be implemented for carbon mitigation by developing countries only. NAMAs for blue carbon can serve for implementation of demonstration projects. Funds for implementation of NAMAs could be accessed (or further mobilized) through a number of multilateral and bilateral initiatives currently providing fast-start finance.

As an extension to the rather narrow boundaries of compliance markets, voluntary markets have developed. These markets target companies or individuals that wish to compensate their GHG emissions on a voluntary basis. The voluntary market is small (less than 3% of the value of the regulated markets), but is more open to blue carbon. Blue carbon has thus had several projects developed through organizations like the Verified Carbon Standard (VCS) or the American Climate registry (ACR) certify carbon mitigation projects and the issuing of carbon credits in the market. The VCS is the most advanced for developing coastal carbon systems. It has launched the requirements for crediting wetland conservation projects. In doing so, VCS has sought to expand its scope to mangroves, coastal wetlands and possibly seagrass ecosystems as well. Other standards include Climate Community and Biodiversity Standard the Carbon Fix Standard, Plan Vivo Systems and Standards. The latter standard is in the process of certifying one mangrove project based on community-led conservation and plantation projects in Kenya.

Apart from standardized markets, the Ocean Foundation has developed the independent seagrass carbon compensation scheme (Seagrass grow!)62. The Ocean Foundation therefore supports two seagrass conservation projects and enables companies and individuals to compensate their carbon emission online. This initiative is the first that applies to seagrass, where the methodology is not standardized and carbon volumes stored annually (and the corresponding actions that allow for an increase or maintaining this storage) not yet transparent.

In summation, though scientific evidence exists to support the carbon sequestration benefits of coastal ecosystems, blue carbon sinks have largely fallen outside of international and national climate change policies. A major priority should therefore be to support scientific research to better analyse the quantity of emissions captured by blue carbon sinks and therefore provide arguments to include them into the accounting framework. This can be developed through demonstration projects as part of the NAMAs

62www.oceanfdn.org
at a country level, or through voluntary markets if the country does not bear sufficient support to develop blue carbon. These represent good avenues for practical, science-based methodologies and tools for further inclusion of blue carbon within the UNFCCC framework. However, the number of current blue carbon projects in the voluntary markets is still limited and there is an urgent need to develop project in ecosystems that are active carbon sinks such as the Mediterranean seagrass.

What would be the expected benefits from such seagrass blue carbon in Mediterranean MPAs?

The MPA implementing seagrass blue carbon projects would first benefit from the sale of carbon credits, which could be substantial. Depending on the surface covered by the project and the price of the carbon dioxide, credit sales could generate profits to the MPA in order to finance its management. Recent estimates of seagrass in the Med could represent a value of 6 to 23 euros per m² per year (considering a CO2 price of 15 euros per ton), which is 9 to 35 times more than for a m² of tropical forest (Laffoley and Grimsditch, 2009; MacCord and Mateo, 2010).

Such a project could also provide scientific benefits in increasing the knowledge of critical habitats (such as Posidonia ecosystems) in the MPA and improve the management measures that apply to these.

In addition, the avoided degradation measures on seagrass ecosystems would provide important indirect benefits to other services that seagrass provide: habitat and nursery sites to fisheries species, ecotourism, increased erosion control, enhanced water and waste treatment, etc.
11.7 Appendix 7: Calculation of the non-use values in the Prêcheur reserve (Binet et al., 2012a; Binet et al., 2012b)

For the Prêcheur marine reserve in Martinique, considering visitor population, the indirect utility function defined by multinomial logit model was:

\[ V_{ij} = 0.5962Z_{\text{pays}} + 0.2101Z_{\text{mar}} - 0.0175Z_{\text{ct}} + 2.0144Z_{\text{env1}} - 0.0926Z_{\text{env2}} + 0.9332Z_{\text{con1}} + 0.5591Z_{\text{con2}} + 2.4501Z_{\text{pre1}} + 3.34Z_{\text{pre2}} - 0.1285Z_{\text{nuit1}} + 0.0016Z_{\text{nuit2}} \]

With: \( Z_{\text{pays}}, Z_{\text{mar}} \) and \( Z_{\text{ct}} \) being the three variables of the attributes 'beauty of coastal sceneries', 'richness of submarine life', and 'costs' respectively; \( Z_{\text{env}}, Z_{\text{con}}, Z_{\text{pre}} \) and \( Z_{\text{nuit}} \) being various individual variables at different levels of realisation. The only presented variables are those that have been found as significant.

The coefficient estimates for the three attributes represent the marginal utilities associated with the upgrade of each of the attributes from a degraded level to a good level. In other words, they represent the relative preference for progress from one level of attribute to a higher level of the attribute. In this case, visitors have a preference (shown by a higher coefficient than the second non-monetary attribute) for the first attribute 'pays' which corresponds to 'beauty of coastal sceneries' attribute. The cost attribute negatively influence the utility function, which is to be expected: the higher the cost, the lower the willingness-to-pay (WTP) for better health status of the ecosystems. However, a coefficient close to zero indicates that the cost attribute poorly influence the WTP of respondents.

The coefficient calculated for individual variables gives more details on the choice of respondents: it shows how the variable considered influences the WTP of respondents, at the level of realisation considered. For instance, the coefficient of \( Z_{\text{env1}} \) is substantial, showing that people who have a job related to the environment are more inclined to pay than people with jobs that are disconnected from the environment \( Z_{\text{env2}} \).

It is then possible to derivate the marginal values for each of the non-monetary attributes through the marginal utility of cost attribute (Rolfe et al., 2000). The marginal value of the improvement of one of the non-monetary attribute is expressed by WTP:

\[ WTP_{\text{pays}} = -\frac{\beta_{\text{pays}}}{\beta_{\text{ct}}} \]
With $\beta_{pays}$ being the coefficient of the attribute ‘beauty of the sceneries’ and $\beta_{ct}$ the coefficient of cost attribute.

In this case, the marginal value for the improvement of the ‘beauty of coastal sceneries’ amounted to 23 euros for residents and 34 euros for visitors. The marginal value for the second non-monetary attribute is calculated with the same method.

Then, the non-use value is the result of the simultaneous improvement of the two non-monetary attributes. It is possible to calculate the simultaneous change of two attributes through the marginal utility of cost attribute (Hanley et al., 1998). The change considered in this case (improvement) was defined from a ‘status quo’ situation which was well described, which enables us to consider the economic value as absolute.

The total value associated with the realisation of simultaneous improvement of the two non-monetary attributes is the difference between the utility for the improvement of the two non-monetary attributes ($V_1$) and the utility of the ‘status quo’ situation ($V_0$) divided by the marginal utility of monetary attribute ($\beta_{ct}$). It is expressed by the formula:

$$CAP_{total} = \frac{V_0 - V_1}{\beta_{ct}}$$

As a consequence, the total value associated to the joint improvement of the two non-monetary attributes amounts to 70 euros for residents and 46 euros for visitors.

Based on the total population of Martinique, the non-use value of the residents in Martinique for the Prêcheur coastal ecosystems amounts to more than 28 million euros per year. For visitors, this value is 27 million euros.

The survey has enabled us to divide this total value in to existing and bequest values.