Exploring the Theoretical Construct of Marine Payments for Ecosystem Services

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A Tool for Marine and Coastal Conservation

Lessons Learned from Terrestrial Payments for Ecosystems Services

Universiteit Utrecht

UNIVERSITÄT LEIPZIG

Joint Degree MSc in Sustainable Development
(Track: Resource Management)

Master Thesis
(GEO4-2321)

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Date: 13.03.2012

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Acronyms

ACE	Annual Catch Entitlement
BNMP	Bonaire National Marine Park
CDM	Clean Development Mechanism
CES	Compensation for Environmental Services
CIS	Co-Investment in Stewardship
CO_2	Carbon Dioxide
COP	Conference of the Parties
COS	Compensation for Opportunities Skipped
CPR	Common Pool Resources
EEZ	Exclusive Economic Zones
ES	Ecosystem Services
FAO	Food and Agriculture Organization
FCPF	The Forest Carbon Partnership Facility
FIP	Forest Investment Program
GDP	Gross Domestic Product
GHG	Greenhouse Gas
ITQ	Individual Transferable Quotas
MEA	Millennium Ecosystem Assessment
MPA	Marine Protected Area
MPES	Marine Payments for Ecosystem Services
NGO	Non-governmental Organization
NRM	Natural Resource Management
NZ	New Zealand
PES	Payments for Ecosystem Services
PSA	Pago por Servicios Ambientales
QMS	Quota Management System
REDD	Reduced Emissions from Deforestation and Forest Degradation
STINAPA	Stichting Nationale Parken (National Parks Foundation)
TAC	Total allowable catch
UNDP	United Nations Development Programme

UNEP	United Nations Environment Programme
UNFCCC	United Nations Framework Convention on Climate Change
UN-REDD	The United Nations Collaborative Programme on Reduced Emissions from Deforestation and Forest Degradation
VOC	Volatile organic compounds

Summary

Human well-being depends to a great extend on ecosystem services (ES). However, the current rates of abstraction and use of these goods and services in order to sustain and increase human well-being give rise to trade-offs in consequence of their overexploitation and unsustainable use. Often the aftermath are high social and economical costs. Also marine and coastal ecosystem services add up to a high percentage of the globally provided services for our well-being. However, increasing fishing activities, industrial and agricultural pollution, coastal development and other anthropogenic influences are putting high pressure on these ecosystems. Additionally, compared to terrestrial conservation and protection, marine conservation lacks behind and several studies showed how marine protected areas fail in achieving their objectives of marine conservation. Due to the ineffectiveness of common top-down and command and control approaches to solve the above mentioned trade-offs, market-based mechanisms receive more and more attention. Payments for Ecosystem Services (PES) is one of these mechanisms. These payment schemes are believed to have mutual benefits including the continuous and sustainable provision of ES while taking other social and economical aspects into account. This is especially true for natural resource management practices on the terrestrial scale.

For the marine and coastal environment, however, similar payment schemes have received little attention until now. Taking into account that various studies already document and give information on the values of fisheries, tourism, carbon sequestration, and coastal protection, it is likely that MPES hold a great potential for marine and coastal conservation. Considering the potential MPES have for marine and coastal conservation, the objective of this research project is to show what the lessons learnt are from payments for ecosystem services and to what extend these insights can be used for the successful implementation of MPES. Hence, the steering research question is: *What can we learn from terrestrial payments for ecosystem services for the successful implementation of Marine Payments for Ecosystem Service ?*

From the literature several success conditions and design principles for terrestrial PES could be derived and then divided into four different categories (1) institutional context, (2) biophysical context, (3) social context, and (4) economic context. These aspects are the foundation for the developed preliminary assessment framework. The framework focuses on design principles of MPES interventions which contribute to their likelihood of success on implementation. By applying the framework to three different MPES examples (of which two case studies have been identified as success stories, namely Individual Transferable Quotas and Marine Protected Areas; while the other example - mangroves as blue carbon sinks which has not been implemented yet due to various reasons) its applicability and feasibility was tested.

Due to the broad design of the framework it allows the analysis of success conditions of different types of MPES schemes. Keeping some limitations of the framework in mind its application indicates that the terrestrial versions of PES deliver important information about success conditions, key factors and design principles for potential MPES schemes. Furthermore, through the application of the framework the presence of different design principles which influence the potential success of a payment scheme could be identified and weighted. The creation of a stewardship behaviour of the resource manager/land-owner turned out to be of great importance in order to achieve sustainable land- and resource-use practices. Inclusion and active participation of stakeholders are other important criteria. Also, special attention should be paid to the institutional conditions and their interplay. Good and effective local institutions are essential for achieving sustainable management practices for natural ecosystems. For this reason, especially tenure, property and use-rights have to be considered and further investigated in order to reach equitable and sustainable solutions which include local communities, their social implications, economic development and marine and coastal conservation.

Keywords: Payments for Ecosystem Services, Marine Payments for Ecosystem Services, Individual Transferable Quotas, Marine Protected Areas, Blue Carbon Sinks, success conditions and design principles for PES

Acknowledgement

Writing and finishing this paper has only been possible with the input and support from several people. Special thanks go to my supervisor Dr. Frank van Laerhoven who had to put up with and follow my ever changing ideas and other obstacles. Nevertheless, I could always rely on the provision of his expertise, insights and suggestions.

Also, the support of my dear friends in seemingly endless library-sessions, discussions and uncountable coffee-breaks help me finishing this paper.

Last but not least I would like to thank my family for their unconditional support not only in the final moment of my studies but during my whole education.

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1 Introduction

Ecosystems provide various types and kinds of Ecosystem Services (ES) e.g. food production, air and water purification, which are essential for sustaining human well-being. Unsustainable use and wasteful handling of natural resources and ecosystems create negative externalities, which often bear high social costs. Fortunately, scientists', politicians' and public awareness has been rising for the last few years and the issue of the currently unsustainable development has been acknowledged to a certain extent (Pagiola et al., 2004; Daily & Matson, 2008). Nevertheless, top-down approaches implemented by environmental policy- and decision-makers, as well as command and control regulations that were instituted in order to deal with the above mentioned problems, turned out to be not as successful as assumed (Echavarria *et al*., 2003). Hence, in order to decrease or prevent such negative environmental externalities created by pollution and ecosystem degradation a shift towards economic and market-based instruments could be observed. A different and more recent approach, which is based on financial incentive measures is the generation of positive environmental externalities (Mayrand & Paquin, 2004). One of these recent approaches are the payments for ecosystem services (PES) which

> *"support positive environmental externalities through the transfer of financial resources from beneficiaries of certain environmental services to those who provide these services or are fiduciaries of environmental resources"* (Mayrand & Paquin, 2004: p.1).

The participants of the Regional Forum on Payment Schemes for Environmental Services in Watersheds agreed on the definition of PES schemes as "flexible, direct and promising compensation mechanisms by which service providers are paid by service users" (FAO, 2004). These compensation mechanisms establish a market for a given environmental good or service. Hence, PES schemes are supposed to be more effective and cost-efficient than common command and control mechanisms due to their setup and potential of integrating externalities. The most popular PES schemes are based on creating positive externalities from the services of watersheds, biodiversity and carbon sequestration (Mayrand & Paquin, 2004).

In comparison to terrestrial conservation which received a lot of attention and momentum through potential markets for ecosystem services and payment schemes for ecosystem services, marine conservation lacks significantly behind. However, depending on the situation e.g. the ecosystem service and other socio- economic factors, Marine Payments for Ecosystem Services (MPES) might be a useful tool for marine and coastal conservation measures. The preconditions like the type of resources, available institutions and secure property rights etc. are substantially different for MPES then for terrestrial PES schemes. However, it might still be of importance to look at the success conditions of PES schemes in order to see what can be learned and used for the successful implementation of MPES.

1.1 Marine Payments for Ecosystem Services

MPES is a relatively new concept and currently there is only little knowledge about such payments for ecosystem services and few examples exist. Nevertheless, various studies already document and give information on the values of fisheries, tourism, carbon sequestration and coastal protection. Ingram and Wilkie (2009, p.2) suggest that

> "*the lack of Marine Payments for ecosystem Services schemes may be due to a lack of analysis regarding proper mechanisms for the trade and exchange of these services, the complex nature of marine and coastal property rights, and the lack of globally fungible services in the marine and coastal environment such as storm protection*"

Hence, only a few actual programmes exist. Most of them are related to fisheries, marine protected areas and biodiversity conservation. Several ES can be identified as potential assets for MPES schemes:

- beach maintenance and production
- marine and coastal carbon storage and sequestration
- fish nursery habitats
- marine species, habitat, and biodiversity conservation
- marine species bioprospecting (Forest Trends and The Katoomba Group, 2010)

There are various tools and mechanisms which can be used for coastal and marine conservation. However, some of the top-down approaches are not as successful and efficient as they were intended to be (Daily & Matsen, 2008). In relation to terrestrial ecosystem conservation and protection, PES schemes have in certain situations proven to be useful and effective by creating positive externalities (Fischer et al., 2010).

1.2 Research Objective and Issue

Accepting the indicators for success of terrestrial PES schemes to solve the problems of degradation of ecosystems and natural resources, implementation prerequisites and key factors for success might be useful and applicable to the marine and coastal version of payments for ecosystem services. Considering the potential MPES have for marine and coastal conservation, the objective of this research project is to show what the lessons learnt are from payments for ecosystem services and to what extend these insights can be used for the successful implementation of MPES. For this reason the first part of the research project will focus on marine and coastal ecosystems and on the terrestrial version of payments for ecosystem services. Based on that research a preliminary assessment framework will be developed and used to indicate success and failure of existing MPES examples, which is derived from the success conditions and key factors of terrestrial PES interventions.

Derived from the research objective the following research question will be used to steer the research project:

> ***RQ-1: What can we learn from terrestrial payments for ecosystem services (PES) for the successful implementation of Marine Payments for Ecosystem Service (MPES)?***

In addition the following sub-question will be answered:

> *RQ-2: How can Marine Payments for Ecosystem Services (MPES) contribute to successful coastal and marine conservation?*

This sub-question focus on the elaboration on the different types of MPES.

> *RQ-3: What is the applicability and feasibility of the developed preliminary assessment framework?*

The applicability of the developed framework will be tested by using it on already existing successful and less or unsuccessful MPES interventions.

1.3 Research Methodology

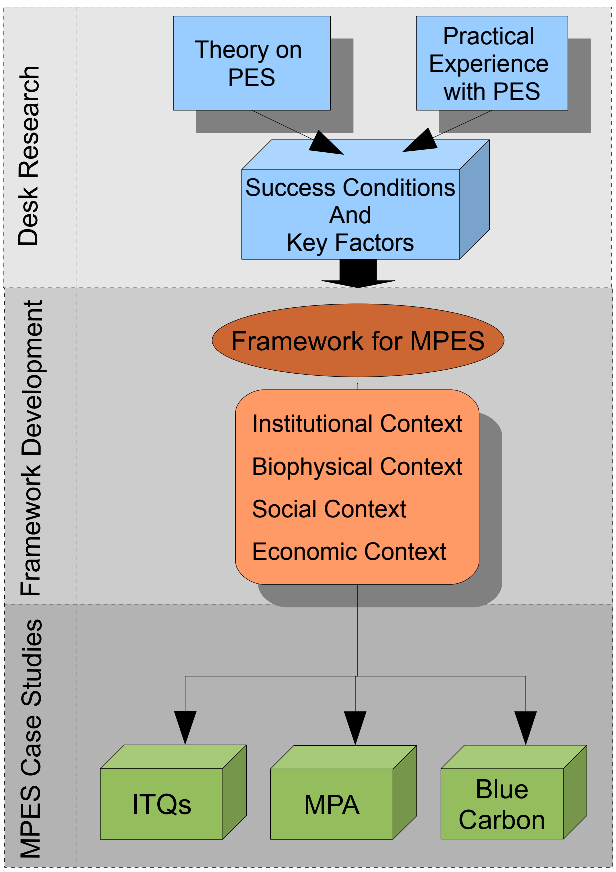

Figure 1: Research Framework

The first part of the study project is based on literature reviews and uses the citation index in order to identify experts and key authors on the subject of PES. Firstly, the study will look at Ecosystem Services in general before narrowing the research down to coastal and marine ecosystem services. Then the concept and theoretical background of terrestrial PES will be described before focusing on MPES. Chapter 5 illustrates the success conditions derived from an extensive literature review which are valid for PES. From these success conditions several independent variables can be derived for the development of a preliminary assessment framework. This framework can be used to assess and explain the success or non-success, the efficiency and feasibility of different MPES schemes. This mainly deductive research and analysis will be completed with the application of the preliminary assessment framework to several MPES case studies. The first case study will be from New Zealand and will report on the implementation of an individual transferable quota system for a more sustainable management of New Zealand fisheries. A second example will deal with mangroves and their missed potential as blue carbon sinks for carbon sequestration and storage programmes. Regardless of the high potential of mangroves to capture and store CO_2 and the provisioning of other essential ES these ecosystems are still degrading at an alarming rate and little payment schemes have emerged until now. Hence, the developed framework will look at the circumstances and conditions of mangroves and related ES in order to investigate the crucial shortcomings, potential corrections and solutions. The third case study investigates the success and non-success of MPAs which are the most applied marine and coastal protection mechanisms world wide, in combination with multiple use purposes including tourism. Figure 1 illustrates the research framework and shows the research methodology from desk research over the development of the framework to the application of the framework to the three MPES case studies.

The evaluation of the case studies will help to determine the feasibility and applicability of the developed preliminary assessment framework. It is therefore possible to identify and evaluate the presence of the summarised key factors and different success conditions and their particular influence on the overall success of each program.

The selection of the used case studies was based on practical issues such as the available information and on published literature and secondly by the actual implementation of these schemes.

1.4 Outline of the Research

Chapter 2 gives an introduction to ecosystems, their valuation and their importance for human well-being. It continuous with an introduction to the various marine and coastal ecosystems, the services they produce and their importance. Furthermore, chapter 2 gives a short insight into the current state of marine and coastal protection and conservation. Chapter 3 introduces the theoretical concept of PES including the different types, payments and their specific characteristics. After elaborating on the background of marine and coastal ES, chapter 4 focuses on the theoretical construct of marine payments for ecosystem services in general. Chapter 5 summarises and elaborates on the experiences, success conditions and lessons learned from terrestrial PES schemes which will be used for the creation of the preliminary assessment framework.. At the end of chapter 5 the framework will be presented before chapter 6 introduces the different case studies of Individual Transferable Quotas, mangroves as blue carbon sinks and Marine Protected Areas. For each case study the general concept will be described first. Then a short introduction of the specific case follows. Each subchapter concludes with an elaboration on the specific context as an MPES scheme more in detail before applying the developed framework. After the detailed description and assessment of the three case studies a discussion in chapter 7 follows including a detailed typology of the used case studies before concluding with chapter 8.

2 Ecosystem Services

2.1 Ecosystem Services and Human Well-being

Ecosystem services are the basis for and directly linked to human well-being and are defined as "[...] *the benefits people obtain from ecosystems*" (MEA, 2005; p.26). ES can be categorised into

> "*provisioning services such as food, fuel and fibres; regulating services including climate regulation and disease control; supporting services like soil formation and nutrient cycling; and cultural services which are nonmaterial benefits such as recreational and spiritual services"* (MEA, 2003; p.3, BOX 1.).

The growing demand for such ES and natural resources are more and more leading to various trade-offs. Fish as a food resource are one of the most important protein sources around the world but especially in developing countries. Not only are fisheries the source of food but also play an important role for the integrity of ecosystems and related provisioning of ecosystem services (Holmlund & Hammer, 1999). According to the World Resource Institute (1996) almost 70% of the most important marine fisheries have been overfished in the year 1995. These fishing practices put high put high pressure on fisheries and marine and coastal ecosystems and constrain the services they provide (Holmlund & Hammer, 1999). The Millennium Ecosystem Assessment (MEA, 2005) investigated the current use and degradation of ecosystem services and found that approximately 60% are being used in an unsustainable way and are subject to degradation. These ecosystem services include fresh water, capture fisheries, air and water purification, and the regulation of regional and local climate, natural hazards, and pests (MEA, 2005). The proceeding degradation and even losses of ecosystem services bear high economical and social costs which, however, are difficult to estimate but are believed to intensify in the next years. According to the MEA (2005) the above mentioned trade-offs and the related costs are shifting from one group of people to another or are passed on to future generations. ES are usually undervalued or have no economical value at all (Costanze et al., 1987). Hence, the degradation of natural resources and ecosystems is more severe due to different forms of market failures e.g. external effects and missing property rights (Engel, 2008,p.664; Tietenberg, 2006). The provisioning of ecosystem services are directly linked to human well-being and their relation and strength of linkages are illustrated in figure2.

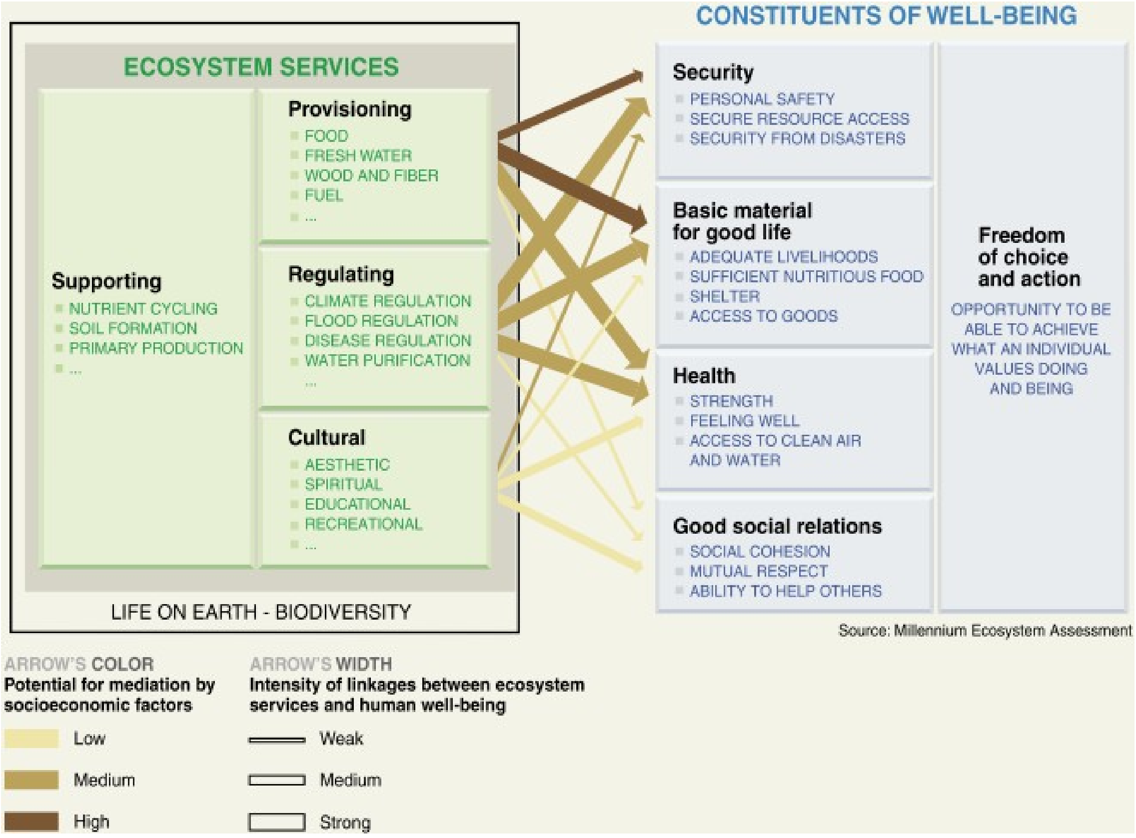

Figure 2: Strength of linkages between ecosystem services and constituents of human well-being

Source: Millennium Ecosystem Assessment, 2005

In order to counteract these trends of unsustainable development and natural recourse degradation environmental policy- and decision-makers used top-down, and command and control regulations. Nevertheless, most of such measures turned out to be not as successful as assumed and hoped (Echavarria et al., 2003; Sommerville, 2009; Swallow et al., 2009; Wunder, 2005).

Negative externalities as well as decreasing provisioning of essential ecosystem services can be linked to the unsustainable use and management of ecosystems. The lack of and difficulties in the economical valuation of ES contribute to this problem (de Groot et al., 2002). With regards to the economic valuation of ecosystem services Hawkins (2003) distinguished between different categories which make up the total economic value of ES. First of all it can be distinguished between (1) '*use value*' and (2) '*non-use value*'. The use values can be further classified and divided into different categories (Hawkins, 2003):

Direct use values are derived from the direct and physical use of ES such as fish, food, wood, medicines, recreation and tourism. Furthermore, the direct use value can be further divided into consumptive e.g. fish, fuel wood and non-consumptive use such as recreation and tourism.

Indirect use values arise from the supporting and maintaining humans and their well-being through services such as flood control and carbon sequestration.

It can be distinguished between three different kinds of non-use values:

Option values are linked to the possible benefits provided by ecosystems in the future which might not be used currently. An example is the potential of deriving new medicines through the maintenance of biodiversity.

Bequest value is the value and satisfaction of passing on current environmental and ecosystem benefits to future generations.

Existence value is derived from the knowledge of the mere existence of intact ecosystems even though no direct value can or will be derived from them.

The above described different values make up the total economic value of ecosystem services. Their relation and categorisation is displayed in figure3 (Hawkins, 2003).

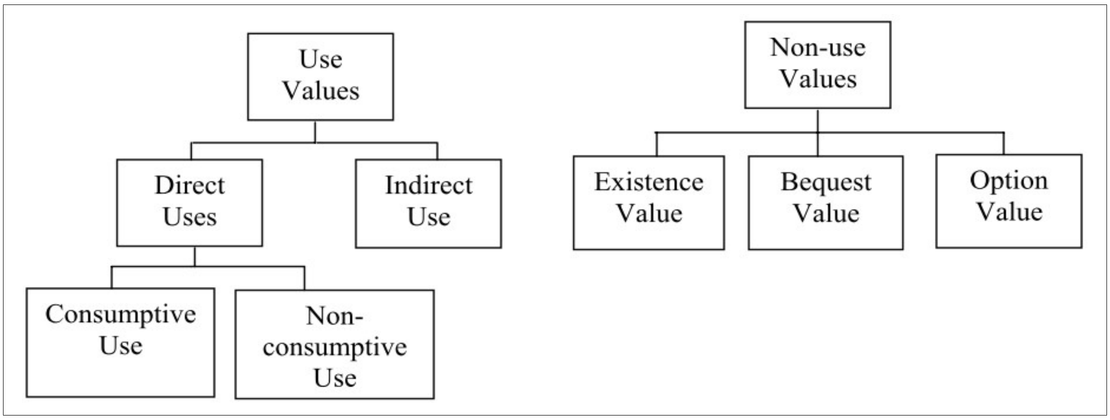

Figure 3: Types of values

Source: Hawkins, 2003: p.13

2.2 Marine and Coastal Ecosystems

The MEA (2005) uses the widely used and acknowledged classification of different marine systems brought forward by Longhurst and colleagues which divides marine systems into four biomes (Longhurst et al. 1995; Longhurst 1998): the coastal boundary zone, trade-winds, westerlies, and polar zone. The most productive zone, yielding around 90% of marine fisheries, is the coastal boundary zone, which is the area surrounding the continents. The trade-winds, westerlies, and polar zones are less productive in comparison and are usually being fished for large pelagic fish. The annual average fisheries landings from coastal and marine systems combined added up to 82.4 million tons per year in 1991 and 2000. Nevertheless, due to reasons of overfishing an overall declining trend in catches could be observed even though fishing efforts increased.

It is difficult to generalise the various marine ecosystems because of their diversity. However, all of them are ecologically important at the global scale and are essential for human well-being. Sherman (1991) speaks of large marine ecosystems and defines them as large marine systems that support and provide several services including "*climate regulation, the freshwater cycle, food provisioning, biodiversity maintenance, energy, and cultural services, including recreation and tourism*" (MEA, 2005, p.480). Additionally, marine systems play an important role in the economic sector. According to the FAO (2002) in the year 2000, captured fish alone added up to a monetary value of approximately $81 billion.

Furthermore, marine tourism along the coasts contributed with $161 billion in 1995, aquaculture with $57 billion in 2000 and the offshore oil and gas industry added up to $132 billion also in 1995 (FAO 2002). An estimated 15 million fishers are working within the marine capture fisheries sector, whereas the majority, about 90% of these fishers work in small-scale and artisan fisheries (FAO, 2002). Caught fish is an essential source of animal proteins to more than one billion people worldwide, especially in the developing countries. Due to the worldwide growing population, and hence a growing demand for food, migration of people to coastal areas and other changes in consumption lead to an ever increasing demand for food and other services from the oceans and seas (Agardy 2010). As a consequence the increased demand for food is causing trade-offs, namely heavy reliance and dependence on various marine ecosystem services but proceeding and increasing degradation of the same ecosystems (MEA, 2005). Over the last 50 years fishing practices had and still have the greatest impact on marine ecosystems. Apart from the decline and even collapses of stocks of high-value species, overfishing can also be observed in the shift of catches from higher trophic levels to the more abundant lower trophic levels by the industrial fleets. Depth and distance from the coasts are becoming more and more irrelevant due to the rapid development of high-tech fishing vessels and fishing industrialisation. This also puts pressure on deep-ocean fauna.

Apart from the already high pressure on marine ecosystems through overfishing, other developments such as the oil and gas industry affect the health, resilience and productivity of marine and coastal ecosystems). Examples for aforementioned energy sources are wind energy, mining for gold, diamonds and tin, and ocean dumping. (Walser & Neumann, 2008)

The coastal boundary zone biome (10.5% of the world ocean) includes the coastal system (as defined by the MEA) which reaches from 0-50 meters, the outer shelves (50-200 meters), as well as most of the continental slopes (200-1000 meters).

Due to its close location to the coast, this biome has been subject to fisheries the longest. From a global perspective most of the caught marine fish is taken from the coastal boundary biome and hence it is subject to a high degree of ecosystem degradation and exploitation. Due to the low water depth this biome is exposed to practices of bottom-trawling on a regular bases, which has wide spread impacts on the seabed while at the same time 'bycatch' (catching unwanted fish species or size) is a common appearance. Bottom-trawling can have immense impacts on ecosystems and leads to fundamental changes in them as can be observed in the North Sea after a century of trawling (Malakoff, 1998).

Other destructive fishing methods including dynamite fishing and the use of cyanide usually applied and used by small-scale fishers put high pressure on coastal habitats like coral reefs, soft corals and sponge beds (Cesar et al. 2003). These unsustainable fishing practices cause severe damages especially to coral reefs and their ability to recover. The MEA (2005) states that a high percentage of the unsustainable fishing practices and the decline in fish stocks are due to the development in fishing technology including boats and gear.

2.3 Marine and Coastal Ecosystem Services

Ecosystem Services are essential and support all kind of life on Earth. Loss of biodiversity will result in a decrease of ecosystem functioning and the provisioning of ES. These services come as food or material or are physical and chemical processes which interact with other process in the environment (Duarte, 2000). Ecosystem services can be categorised as: provisioning, regulating, cultural, and supporting services (de Groot et al., 2002, MEA 2005; Beaumont 2007, Naber et al., 2008; Forest Trends, 2010). Below is a brief description of each of these services:

(a) "*Provisioning services are defined as those that result in products obtained from ecosystems (in some cases referred to as production services)*" (Naber et al., 2008: p.8). Examples of provisioning services from marine and coastal ecosystem services are food, fisheries, aquaculture, fiber and timbers (for constructions), fuel, genetic and pharmaceutical products.

(b) "*Regulating services are defined as those that regulate ecosystem processes*" (Naber et al., 2008: p.8). Examples are biological regulation, freshwater storage and retention, atmospheric and climate regulation, carbon sequestration, waste processing, shoreline stabilization, flood/storm protection, and erosion control.

(c) "*Cultural services are the non-material benefits people obtain from ecosystems*" (Naber et al., 2008: p.9). The most prominent examples of these services are cultural and amenity, recreational, tourism, aesthetics, and education and research (de Groot et al., 2002, Beaumont 2007).

(d) "*Supporting services are those services that are necessary for the production of all other ecosystem services, but do not yield direct benefits to humans*" (Naber et al., 2008: p.10). Examples of supporting services include resilience and resistance (life support), nutrient cycling and fertility, biologically mediated habitats (de Groot et al., 2002, Beaumont 2007, Naber et al., 2008).

The most prominent and for this study most important ES derived from marine and coastal ecosystems are:

Fisheries: Several habitats, like coral reefs, mangroves and seagrass beds are important fish nurseries (Duarte, 2000).

Coastal protection and shoreline stabilization: Coral reefs, mangroves, beds of seagrass are natural barriers and protect coastal areas and their inhabitants from storm surges, flooding and even tsunamis.

Biodiversity: Intact and healthy ecosystems, especially coral reefs, mangroves, kelp forests and seagrass beds are the source of high biodiversity. A high biodiversity as a positive relation to the provision of Ecosystem Services (Duarte, 2000).

Carbon sequestration: Each marine and coastal ecosystem has an important role in the carbon cycle. However, especially mangroves hold a great potential for carbon sequestration and storage.

Tourism: Recreational and aesthetic values are also important. Healthy and beautiful coral reefs attract many tourists for recreational purposes, snorkelling and scuba diving.

The above described ES are essential for human well-being and for the functioning of ecosystems. However, they are subject to increased degradation and unsustainable management practices. The following section gives a short introduction into the current protection and conservation approaches (Duarte, 2000).

2.4 Protection of Marine and Coastal Ecosystems

Attempts for global marine protection emerged as a consequence to the ever increasing pressure on ecosystems and the degradation of marine and coastal environments and the depletion and collapse of fish stocks (Gary et al., 1998; Turner et al., 1999). These attempts for natural resource protection and conservation should ideally be based on scientific data which indicate the best practices and the handling of these resources (Daw & Gray, 2005). However, the transformation of scientific data and their translation into sound and practical policies is often not given due to high costs caused by extensive data collection in order to make representative predictions on stock assessments

(Pitchford et al., 2007). It is also difficult due to other various influences like social, economic and political factors. Daw and Gray (2005; p.189) point out that "[…] *such a pattern in fisheries science and policy, where the lack of effective management has contributed to a crisis in world fisheries*" (Daw & Gray, 2005). As a consequence, many essential fish stocks are already considered over-fished at a global level (Pitchford et al., 2007). Roberts & Polunin (1991) differentiate between two different types of classical fishery management:

1. management by catch which focuses on gear restrictions or the implementation of catch quotas, and
2. management of effects which imposes limitations of fishermen or vessels, seasonal restrictions, or temporary closing off areas for fishing activities.

Nevertheless, these traditional or classical fishery management approaches are not always effective (Roberts & Polunin, 1991). Another more recent tool to counteract these trends of marine and coastal ecosystem degradation and loss of fisheries are Marine Protected Areas (MPA) or marine reserves that have received a lot of attention during the last years and. These are strongly advocated by managers as well as biologists (Gary et al., 1998).

Several conventions, protection and conservation programmes starting with the Geneva Conventions on the Law of the Sea in 1958 and others such as the Ramsar Convention (1971—covering Wetlands of International Importance), the UNESCO World Heritage Convention (1972) and the UNEP Regional Seas Programme (1974) and later the World Conservation Union (IUCN) brought forward the development and establishment of Marine Protected Areas with the conservation of coral reefs, mangroves, and salt-water plants as the primary driver (Thorpe, 2011). During the 80s the mere conservation purposes of MPAs were transformed into more complex management schemes that included other goals beside conservation e.g. tourism/eco-tourism. Thorpe et al. (2011) state that the boost in the establishment of MPAs came with the introduction and ratification of the UN Convention on the Law of the Sea (UNCLOS) in 1994 and with it the possibility to establish MPAs outside national territorial (3 nautical miles) waters.

Wood (2011) argues that global marine protection targets which are usually percentage-based, are inadequate, over-ambitious and unattainable as well as ecologically irrelevant, "*particularly because they are rarely sufficient to ensure persistence of populations*" (Wood, 2001: p.525). Hence, they are frequently discredited and ignored. An example for the slow process of such conservation targets is the recent Conference of the Parties to the Convention on Biological Diversity in Nagoya, Japan, in 2010 which adopted hardly any additional or changed conservation goals after the last conference in 2006. Additionally, Wood (2011) states that despite the many efforts, conventions and declarations since the United Nations Conference on the Human Environment in Stockholm in 1972 no progress has been made and degradation of the marine environment and the pressure on ecosystems has even intensified.

MPAs are a fundamental part of global marine protection targets[1] and shall be created and 'managed effectively' in 'representative networks'. Even though, these targets sound rather specific, which according to Wood (2011) is an essential precondition for their achievement, they still lack clear and comprehensive definitions. The degree of protection in MPAs varies from "*complete exclusion of human presence to complex multiple use and zoning regulations*" (Wood, 2001: p.528). Hence, they have different objectives and definitions to start with. A general definition of an MPA is provided by The World Conservation Union (IUCN:

> "*Any area of intertidal or subtidal terrain, together with its overlying water and associated flora, fauna, historical and cultural features, which has been reserved by law or other effective means to protect part or all of the enclosed environment*" (Kelleher, 1999: p.XVII).

Though, Tognelli and colleagues (2009) point out that the establishment of most MPAs is not based on scientific relevance or oceanographic and biological features but are rather based on opportunities. Nevertheless, even less then 0.5% of the world seas are protected within MPAs. Apart from MPAs there also exist offshore exclusive economic zones (EEZs) which are subject to a sort of natural resource management approach based on international fishing agreements or national laws and regulations (Spergel & Moye, 2004). Marine protection and management require intensive financial resources. Balford and colleagues (2003) state in their study that the annual costs for a global network of MPAs covering about 30% of the worlds seas would add up to approximately $7 to $19 billion.

With regards to marine and coastal conservation and protection the establishment of Marine Protected Areas (MPA) is the most famous and widely used approach (Thorpe et al., 2011). The primary objective of MPAs is usually biodiversity conservation. However, other objectives have also been on the agenda of MPAs, such as the sustained provision of ecosystem services, cultural and spiritual values, and providing opportunities and space for research and education (National Research Council, 2001; Leslie, 2005). Unfortunately, MPAs often lack financial resources, effectiveness or only exist on the paper (as so called "paper-parks") due to various economical and social reasons. As Thur (2010, p.63) puts it:

> *"Marine protected areas have proliferated globally in the past three decades. However, inadequate funding often prevents these management regimes from fulfilling their missions. Managers have become increasingly aware that successful protection of marine ecosystems is dependent not only upon an understanding of their biological and physical processes, but also their associated social and economic aspects."*

The current state and still increasing degradation of marine and coastal ecosystems and the lack of successful conservation and protection measures indicates the need for more attention and innovative approaches in this field. Especially regarding the high costs for protection and conservation a clear link to the high economical benefits derived from

1 As demanded by the 2002 World Summit on Sustainable Development and the 2003 World Parks Congress

such ecosystems can be recognised. The implementations of no-take zones, MPAs and other top-down regulations are not always successful in achieving their conservation goals. Hence, looking at economic incentives combined with environmental protection objectives could contribute to a higher degree of conservation of these valuable ecosystems. The next chapter will therefore look at the terrestrial version of payments for ecosystem services in order to explore their potential and success conditions for the possible use in Marine Payments for Ecosystem Services.

3 Payments for Ecosystems Services

Payments for ecosystem services or also called payments for environmental services, here used as synonyms[2], is a natural resource management tool (NRM) and seeks to avoid the above mentioned market failures like negative externalities by rewarding e.g. through monetary payments the provision of one or more desired ES. In this sense PES

> *"support positive environmental externalities through the transfer of financial resources from beneficiaries of certain environmental services to those who provide these services or are fiduciaries of environmental resources"* (Mayrand & Paquin, 2004; p.1).

The theoretical construct of PES is well defined by Wunder (2005) and states that a PES is:

1. *a voluntary transaction where*
2. *a well-defined ecosystem service (ES) (or a land-use likely to secure that service)*
3. *is being 'bought' by a (minimum one) ES buyer*
4. *from a (minimum one) ES provider*
5. *if and only if the ES provider secures ES provision (conditionality).* (Wunder, 2005; p.3)

From now on, the notation '*five theoretical PES principles*' will refer to the above described characteristics by Wunder (2005). However, as also Engel et al. (2008) point out not all as PES defined programs fit this narrow and restrictive definition. In fact, many PES programs could only be defined as 'PES-like' schemes since they do not fullfill the '*five theoretical PES principles*' (Engel et al., 2005; Noorwijk & Leimona, 2010; Wunder, 2005). Even Wunder and his colleagues (2008) couldn't always agree where the line between PES and non-PES programs is. Hence, and especially with regards to Marine Payments for Ecosystem Services it would be essential to establish and agree on certain characteristics of PES schemes. Sommerville and colleagues (2009) agreed with Wunders '*five theoretical PES principles*' (2005) but stressed that the focus should be shifted towards the two main characteristics of PES, which differentiate and make such schemes unique. In their opinion the primary criteria of PES are: "*(1)to transfer positive incentives to environmental service providers that are (2) conditional on the provision of the service"* while the considerations for successful implementation depends on "*(1) additionality and (2) varying institutional contexts*" (Sommerville et al., 2009, p.2). According to Sommerville and colleagues, (2009) Wunder's theoretical PES criteria three and four are implicit within the 'P' of PES and are subject to conditionality and should be considered within the '*institutional context*'. The '*well defined ES*' is said to be implicit under the criteria of conditionality whereas the first theoretical principle - '*voluntary transaction*' - falls within the '*institutional context*'. It is here were the different scholars disagree: Sommerville and colleagues (2009) do not agree that it is crucial to all PES that they are voluntary for the participants. The

2 More on the debate about the terms see Sommerville et al. 2009 and Wunder 2005

participation of land managers is not always voluntarily, especially in cases where land-use changes are illegal (Sommerville et al., 2009). Nevertheless, the subject of voluntary PES intervention should be part of, and discussed within the institutional context. Especially with the focus on institutional context and on the slightly varying concepts of MPES, throughout this paper Sommervilles and colleagues (2009) definition will be used, which however does not in any case discredit Wunder's (2005) early definition but rather focuses on different aspects of PES schemes. Wunder's (2005) definition is sometimes too narrow and already caused a lot of dispute as well as confusion between the notions of PES schemes and 'PES-like' schemes. The new definition offers enough flexibility while staying close to the original methodology and idea of PES.

The participants of the Regional Forum on Payment Schemes for Environmental Services in Watersheds attributed several qualities to PES schemes and describe them as "*flexible, direct and promising compensation mechanisms by which service providers are paid by service users*" (FAO, 2004).

The concept behind a PES scheme is that the manager (farmer, logger or the manager of a protected area) of an ecosystem, from now on called ES providers/stewards, is compensated for his/her land-use changes which otherwise would generate little or no personal benefits e.g. forest conservation (Engels et al., 2008). An alternative land-use to forest conservation could be agricultural production on the same area. However, this change in land-use could alter the provided ES in a way that the quality of the water and the people living downstream, also called beneficiaries/receivers would have to carry the costs for these changes and water quality alteration. If, on the other hand, the beneficiaries would reward the upstream ecosystem manager e.g. through financial payments for the continuous provision of the natural ES, forest conservation might turn into a profitable alternative (Engels et al., 2008). The theoretical logic behind this example is displayed in figure 4. Besides ES beneficiaries and ES providers also intermediaries such as local and international NGOs, government agencies as well as specialised agents, e.g. Trust Funds are sometimes involved in facilitating transactions for PES schemes (Landell-Mills & Porras, 2002). In Swallow et al. (2009, p.6) intermediaries are defined as "*entities that directly or indirectly shape interactions among ecosystem stewards, environmental service beneficiaries, and the ecosystem itself*". The tasks and actions carried out by intermediaries are manifold and include the provisioning of information for the design, monitoring process as well as assisting during and after negotiation process and contract. Furthermore, intermediaries can play a key role in "e*nforcing the terms of regulations and contracts, and offsetting the transaction costs of establishing and maintaining a working mechanism"* (Swallow et al. 2009, p.6).

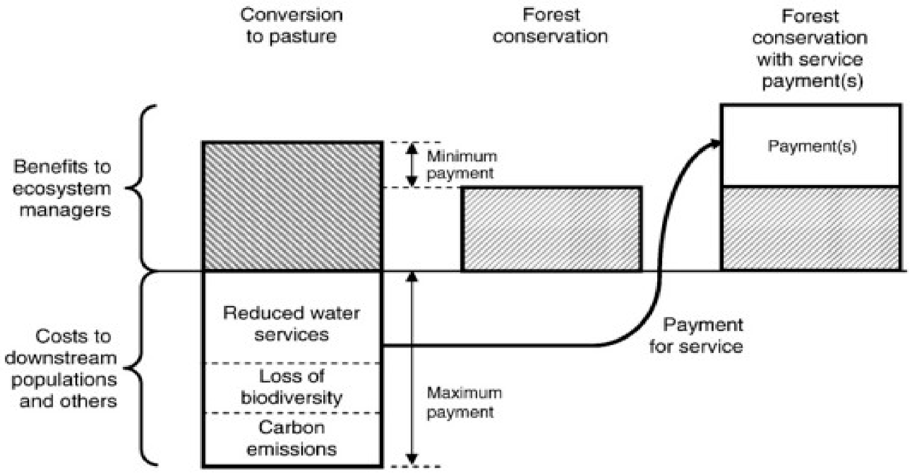

Figure 4: Theoretical concept of payments for ecosystem services

Source: Engels, Pagiola, Wunder, 2008. Designing payments for environmental services in theory and practice: An overview of the issues.

The above mentioned compensation mechanisms establish a market for a given ES. Hence, PES schemes are supposed to be more effective and cost-efficient than common command and control mechanisms due to their setup and potential of integrating externalities (Jack et al., 2007; Mayrand & Paquin, 2004). The most popular PES schemes are based on creating positive externalities from the services of watersheds, biodiversity, carbon sequestration, and landscape beauty. Even bundles of ES can be subject to PES programs. In these cases, multiple ES are combined and provided to beneficiaries (Wunder, 2005).

3.1 PES and ES as Commodities

It is rather obvious that PES cannot be used in every situation as a natural resource management tool to solve environmental problems. In order to elaborate whether PES will be successful it is first of all important to investigate if an appropriate market for the specific service or good exists, which is an essential precondition.

ES markets characteristics
Goods and services can be classified into four different categories as shown in table 1: (1) private good, (2) public good, (3) club/toll good and (4) common pool resource (CPR). The classification is carried out by their relation to the two determining factors: (1) excludability and (2) rivalry/substractability. In order to comply with the conditions of a free market, a good or service has to be exclusive (no one else is able to consume the same good who does not pay for it) and rivalry in consumption(the consumption of the good reduces the overall availability for others).

Table 1: *Excludability and rivalry of goods and services*

	Rivalry	
Excludability	*Low*	*High*
Low	Public Goods	Common Pool Resources (CPR)
High	Club/Toll Goods	Private Goods

Source: Landell-Mills and Porras 2002; p.9

In line with the above explained classifications of goods and services Reis and Synnevåg Sydness (2007) distinguish between three types of ES[3].

1. Environmental services, which share the characteristics of non-rivalry and difficult excludability of others from the benefits – fall under the term public good e.g. public beaches (until a certain degree they are non-rival in consumption), flood control, and protection from storm surges and tsunamis. Hence, the conditions to be effective on the market of such services are not met and the intervention and support of the state in terms of giving incentives for the ES steward to provide the service are essential in order to secure the preconditions for a successful PES program (Reis and Synnevåg Sydness, 2007).
2. In the second category are services which can actually be characterised as public goods but by making them more exclusive they are converted into club goods e.g. a beach front or a Marine Protected Area with an entrance fee. However, similar to the above explained public goods, in the case of club goods the responsible authorities should also be involved by limiting the access to the service and regulating the payments to the ecosystem manager(s) e.g. by price setting.
3. Ecosystem services with the characteristics of common pool resources – not exclusive but rivalry in consumption - make up the third category. Examples of such ecosystem services are related to water quantity and supply on a terrestrial scale. Regarding marine and coastal ES, an example are fisheries. If not regulated everyone has a free access to fisheries. However, if the consumption is high the fish populations become less dense and finally overfished. Due to their characteristics, however, they can be subject to trade between private actors, if and only if the ES beneficiaries are able to develop a way to manage the access to the ES. Marine and coastal ES examples for that are individual transferable quotas in which the participants need to acquire a right to fish a certain share. This quota share, however, is transferable on a market.
   In order to safeguard the successful implementation of a PES scheme with this kind of ecosystem services and to avoid the 'free-rider' problem the prerequisites worked out by Ostrom (1990) in '*Governing the commons - The evolution of institutions for collective action*' are helpful. Ostrom completed a comprehensive and much cited work about how common pool resources can be managed by communities in a proper way and avoiding the outcomes as predicted by Garrett

3 Derived from watershed ecosystem services in Reis and Synnevåg Sydness (2007)

Hardin (1968) known as the '*tragedy of the commons*'. Three prerequisites are of major importance as well as in relation to the success conditions of PES schemes: (1) Boundaries have to be clearly defined of who is allowed access to the ES as well as the boundaries of the geographical dimension of the CPR. (2) Appropriate and effective monitoring tools and (3) graduate sanctions for eventual non-compliance of ES stewards and ES beneficiaries (Reis and Synnevåg Sydness, 2007). More on these matters will follow in chapter 5.

With regards to the preconditions for the implementation of a successful PES scheme the above described categorisation of the ES are important in order to develop fitting institutional regulations, adequate management mechanisms and to choose the appropriate involvement of governmental agencies (Reis and Synnevåg Sydness, 2007; Smith et al., 2006).

3.2 Types of PES

Even though PES are incentive-based mechanisms and usually can be distingueshed from other conservation tools, there are different types or categories of PES schemes. In order to achieve conservation of natural resources or ecosystems PES schemes can either be (1) '*area-based*' or (2) '*product-based*' (Wunder, 2005). According to Wunder (2005) the area-based schemes are the most prominent ones and refer to agreements on pre-established caps on land- and/or resource-use e.g. protected areas or conservation grants. Product-based schemes on the other hand, refer to certifications for e.g. environmentally friendly production which is manifested in a surcharge on the existing market price in order to compensate for the adopted and ideally more sustainable management practices. The transaction in such schemes occurs between private actors while the buyers are consumers who have a preference for such products. However, intermediaries e.g. a certification agency or traders of certified products take an essential role in these PES schemes (Smith et al., 2006). Examples for such schemes are organic farming, certified timber, fish or ecotourism (Wunder, 2005).

Wunder et al. (2008) distinguish between two types of PES schemes with regards to their funding, (1) '*user-financed*' and (2) '*government-financed*'. Greiber (2009) describes these two different PES types as private and public PES respectively. Within a user-financed or private PES program the beneficiaries are paying directly for the provisioning of the service themselves and the conditions of the PES scheme are subject to foregone negotiations between the two or more actors (often through intermediaries) (Asquith & Wunder, 2008; Greiber, 2009; Wunder et al., 2008).

On the other hand, as the name already suggests, the government-financed or public PES schemes involve governmental agencies which act on behalf of service users. The participation however is only voluntary from the side of the service provider but not from the side of the service-user since the payment is based on fiscal mechanisms like subsidies or taxes (Greiber, 2009; Wunder et al., 2008). However, most of the time the ES providers neither have a say in the development and design of the scheme nor can they influence the payment rates they receive. Hence, government-financed programs mostly do not comply completely with the *five theoretical PES principles* and therefore can be referred to as '*PES-like*' initiatives (Wunder 2005; p.21). Nevertheless, such projects often have several objectives and are larger in scale. They therefore have other advantages like the possibility of profiting from economies of scale (Asquith & Wunder, 2008). The public PES schemes are usually implemented country-wide as for example the PES program in Costa Rica[4] (Pagiola, 2008)

However, as pointed out in Wunder et al. (2008), it is not always possible to distinguish between government- and user-financed PES programs since many of them are hybrids. Concerning the user-financed PES programs, Wunder and colleagues (2008) found, despite their small sample of case studies, that such programs came closest to the *five theoretical PES principles*. They were better adjusted to the specific conditions and local circumstances, had more effective monitoring practices, and less scattered objectives in comparison to the government-financed PES programs. Nevertheless, it is not always possible for private users to raise the needed finical funds in order to set up a PES program themselves (Wunder et al., 2008). It is here where the involvement of the government plays an important role since it is the last instance that would be able to establish and operate a PES program that "*offers an important tool to improve the supply of conservation*" (Wunder et al., 2008; p.851).

Another distinction can be made between (1) '*use-restricting*' and (2) '*asset-building*' PES schemes. The first is based on the idea that ecosystem managers are rewarded or compensated for the emerging opportunity costs from conservation measures or decreased natural resource extraction and land development as well as for the active protection against external threats (Wunder, 2005). The latter, asset-building PES schemes are intended to reward the restoration of ES within a given area. Wunder and Boerner (2010) refer to this type of PES as '*use-modification*' and include Clean Development Mechanisms (CDM) to this category. However, both articles (Wunder, 2005; Wunder & Boerner, 2010) point out that these use-modifications can have negative effects on the livelihood and rural employment of the local population.

The differentiation between these various types of PES is important for the application of the later developed framework in chapter 5 for MPES interventions. The different types require different prerequisites and other determining key factors, which will be elaborated on later in this research project. Additionally, the above described types of PES will be used for a typology of MPES schemes in chapter 7 after applying the preliminary assessment framework on the different MPES case studies. The type of PES is also a determining factor for the applied payment or compensation mechanism, described in chapter 3.3.

4 Pago por Servicios Ambientales (PSA)

3.3 Types of Payments and Compensation

As already mentioned the 'P' in PES stands for payments. However, in practice and also in theory the payments do not necessarily have to occur in monetary terms. Nevertheless, as already shown in figure 4 the payments or compensation received by the ecosystem manager must exceed his/her opportunity costs (or the benefits received from alternative land-use practices). At the same time the costs should not be higher than the perceived ES benefits otherwise it is very unlikely that a PES will be adopted by the involved stakeholders and participants (Engel et al., 2008).

In general the payments can have the form of monetary/cash payments, compensations or rewards (Noorwijk & Leimona, 2010). Swallow et al. (2009) differentiate between "Compensation for Environmental Services" (CES) and "Rewards for Environmental Services" (RES). CES refer to the mechanism where either the ES beneficiary is getting compensated for the loss of ES or for a decrease in its provisioning. RES on the other hand, are inducements for the ES provider in order to maintain or increase the provision of ES.

Noorwijk and Leimona (2010, p.12) distinguish between three different paradigms for '*compensation and rewards (including payments) for environmental services*': (1) Commoditized Environmental Service (CES), (2) Compensating for Opportunities Skipped (COS) *or paying land users for accepting restrictions (either voluntary or mandatory) on their use of land* and (3) Co-Investment in Stewardship (CIS).

Asquith and colleagues (2008) differentiate between (1) cash/monetary and (2) 'in-kind' payments or rewards as means of compensation for the provisioning of ES. The next chapter will deal with marine payments for ecosystem services and will give an overview of the various ES and possible payments schemes.

The above summarised theory, characteristics, types and prerequisites of terrestrial PES interventions are the foundation for MPES.

4 Theory on Marine Payments for Ecosystem Services

Marine Payments for Ecosystem Services are very similar to the terrestrial PES version. One of the differences is the different kind of resources and ES at hand and their properties. Especially with regards to property rights of some of the ES like fish, nutrients, reefs, mangroves and seagrass it is often unclear, they depend and vary from country to country and even overlap national, regional or local as well as traditional or informal ownership (Ingram & Wilkie, 2009). Additionally, the fluid nature of most marine and coastal resources make it difficult to trade or identify "sellers" since marine ES can easily travel across administrative and governance systems with literally no transaction costs. Hence, Ingram & Wilkie (2009, p.3) state that

> "*for effective PES markets to develop in the marine context, it will be critical to have clear, legally recognized and functional governance regimes over resources that are somehow constrained to a given area (ie. restricted range or sedentary), so, that the conservation of a service can be directly linked to specific natural resource practices undertaken by someone/entity who is being compensated for their stewardship*".

It is important to point out that the valuation and marketing of marine and coastal ecosystem services faces challenges, which extend the ones from terrestrial PES e.g. clear property and tenure rights, as well as the fact that the benefits, which are generated in one place are often felt and used elsewhere (Forest Trends, 2010). As Agardy (2010, p.5) points out "*the issue of rights is an important one. Clarification of rights can greatly improve the ability to reverse degradation and improve ocean health*". Nevertheless, among conservationists and businesses the interests and awareness of creating potential markets for ES from marine and coastal environment are growing (Pagiola, 2008b; Forest Trends, 2010). Apart from markets dealing with the trade of carbon emissions other incentive driven mechanisms are also likely to develop, such as marine biodiversity conservation, marine species banking, and habitat management. With regards to this development and the problems and lack of public of financial resources, Agardy (2010) point out the importance and potential of marine spatial planning and ocean zoning as a new marine policy mechanism. Ocean zoning is the counterpart to the terrestrial version which is based on a special management approach. Here, natural resource managers are able to assign and point out "*ecologically important areas as well as ecosystem vulnerabilities and sensitivities*" (Agardy 2010, p.5). Additionally, ocean zoning could deliver and tackle the issues of missing or unclear use and property right, which is one of the fundamentals of PES in general and hence, is the basis for true stewardship. Examples for the use of ocean zoning and marine spatial planning can be observed in the area of South-east Asia. According to Agardy (2010), especially in combination with innovative financing mechanisms such as markets for ES and PES mechanisms, ocean zoning holds great potential due to two factors: (1) the creation of use and/or property rights as well as clear responsibilities that attract private investors, and (2) coastal zoning that give the opportunity to include and establish ‘trading zones’ which allow and foster transactions for PES schemes. For example: “*in ‘trading zones’, developers could potentially buy credits for wetlands protection from environmental groups or private land owners, or the insurance industry could invest in barrier beach protection in order to minimize their own risks”* (Agardy, 2010, p.5).

Several ES can be identified as potential assets for MPES schemes

- beach maintenance and production
- shoreline stabilization
- marine and coastal carbon storage and sequestration
- fish nursery habitats
- marine species, habitat, and biodiversity conservation
- marine species bioprospecting
- coastal water quality and pollution filtration (Forest Trends, 2010)

The above listed ES and their role within a potential MPES scheme will be described more detailed within this chapter. However, several of these ES will be described and analysed more precisely in order to establish the theoretical background for the upcoming framework and the analytical case studies.

4.1 Types of MPES

In line with the different types of terrestrial PES, MPES can also be categorized into different types. Forest Trends and The Katoomba Group (2010) distinguish between the following types of markets for MPES.

1. **Compensation of private resource and coastal landowners/managers by a public entity:**

   Within these country-specific types of MPES schemes a public institution or government agency pays resource owners, rights holders, and/or managers directly for their efforts to maintain or enhance marine and coastal ES. Such deals can emerge on a compliance market for carbon offsets in which the government as a public entity is involved and compensated a coastal land-owner or resource manager for protecting mangrove forests as blue carbon sinks.

2. **Establishment of a formal market, which enables open trade between beneficiaries and suppliers.**

   These types of MPES can be divided into two different approaches, a (a) regulatory cap or floor mechanism, or (b) based on a voluntary agreement:

   a) The concept of regulatory Ecosystem Service markets is that the demand is regulated through legislative means with the introduction of a 'cap' on the specific ecosystem service. Hence, the consumers or at least the ones who are causing the degradation of the ES have to respond to the changing market situation by either complying with the regulations or by trading with other actors who are able to stay below the enforced cap. Usually buyers are private-sector companies or other institutions which are defined by legislation in beforehand (Forest Trends, 2010). An example of such a regulatory ecosystem service market comes from New Zealand, which implemented an Individual Transferable Quota (ITQ) system. The Total Allowable Catch (TAC) is allocated via catch quotas to all fishers. The quotas resemble a fixed percentage of the TAC and have the characteristics of secure property rights since it is embedded in the New Zealand legislation. This example will be explained more detailed in chapter 6 and is part of the case studies and will be used for the analysis of the developed framework.

   b) Voluntary markets for MPES are defined by voluntary agreements with usually a private actor who is interested in reducing its carbon footprint through emission trading. This kind of PES can also be defined as a self-organized private deal.

3. **Self-organized private deals:**
   Here providers get contacted directly by individual ES beneficiaries in order to establish a payment scheme. Such schemes usually emerge if there is no proper market for the ES and happen with little or no involvement of a government agency. The ES beneficiaries can either be private companies or also conservationists e.g. an NGO who pays the ES provider to maintain or enhance the quality and/or quantity of the provided ES by adopting or changing management practices. An example for self-organized private deals are carbon offsets through the protection and conservation of mangrove forests as blue carbon sinks. Self-organized private deals for carbon offsets could emerge on a voluntary market between private actors and coastal area managers/land-owners.

4. **Marine Protected Areas (MPA):**
   As already described above MPAs regulate the use and access to an area as well as to the natural resources from marine and coastal ecosystems within. Hence, MPAs contribute to the conservation of biodiversity and to the overall prosperity of the marine and coastal environment which then can attract ecotourism, education and research. In this way MPAs can become financially self-sustaining. MPAs can be established by national or local authorities (including

communities), by multinational institutions (e.g. Regional Seas Agreements), or by private landowners. The primary ES delivered from MPAs are biodiversity, fisheries, recreational and aesthetic values. Several MPAs are already established, which are (more or less) financially independent due to the introduction of tourist fees (Alban et al., 2008; Forest Trends, 2010). Additionally, MPAs are one of the most famous marine and coastal protection measures. Hence, this MPES type will be described more in detail in chapter 6 and part of the application of the preliminary framework.

4.2 Types of Payments and Compensation

Similar to terrestrial PES, for MPES schemes different payment and compensation options are also available. Usually the type of payment/compensation depends on the ES at hand and the actual type of MPES scheme used. It can be differentiated between (1) direct financial payments, (2) financial support for community goals, (3) in-kind payments, and (4) recognition of rights.

1. Direct payments are primarily based on forgone opportunity costs or sometimes even the compensation of lost livelihood due to the protection of an ecosystem and/or the produced ES. Examples are the establishments of non-take areas or the conversion of a publicly accessible mangrove forest into a protected one (Forest Trends and The Katoomba Group, 2010).
2. Financial support can be given for specific community goals e.g. education and health, building a school or a clinic, or more sustainable but expensive fishing gear for fishers (Forest Trends and The Katoomba Group, 2010).
3. In-kind payments are the compensation of conservation and protection efforts with other services and goods such as knowledge, capacity building, and other tangible goods (Forest Trends and The Katoomba Group, 2010).
4. The recognition of rights is based on the concept of providing participants or ES stewards with rights to *participate in decision-making processes,* harvesting and management of ES. It is especially linked to the introduction of the individual transferable quotas (ITQ) in New Zealand (Forest Trends and The Katoomba Group, 2010).

5 Success Conditions and Analytical Framework

When is a PES scheme successful? This generally depends on the specific objectives agreed on by (ideally) all participants of the particular PES scheme. In theory however, PES are supposed to generate at least as much benefits for the land manager for his/her land-use changes in order to provide a certain environmental service as he or she would have made without the changes (Engel *et al*., 2008). Hence, the primary objective is to generate benefits for all participants. Ecosystem conservation and an eventually more efficient use of environmental resources and Ecosystem Services are desired side effects (Bracer et al., 2008). Still, whether the changes in land-use and management can actually be considered sustainable is another question and as Engel *et al*. (2008, p.665) already pointed out "*PES is not intended as a silver bullet that can address any environmental problem*". On the other hand, it is important to evaluate the financial aspects like funding of a PES program and its cost-efficiency. Social implications and poverty reduction through PES can therefore not be considered primary objectives of the same (Pagiola, 2007). Though, if the potential benefits of PES are reduced or its proper implementation is hindered through social constrains, these aspects are at least passively influencing the success conditions for PES. No PES scheme, however, is identical to the next one and it is therefore difficult to develop a blueprint which fits all (Porras et al. 2008). Nevertheless, it is possible to collect and evaluate certain indicators of various PES schemes which contributed to its successful implementation. These conditions apply on the one hand to the design but also to the implementation and actual performance of the PES scheme.

This chapter will first focus on design principles of both PES and MPES as preconditions for the successful implementation before summarizing the various success conditions in themselves. Reis and Synnevåg Sydness (2007) worked out four different main categories for the manifold conditions which influence and determine the success of payments for watershed Ecosystem Services:

1. institutional conditions,
2. biophysical conditions,
3. social conditions, and
4. economic conditions.

Even though their study focuses on watershed related ecosystem services and their success conditions as well as limitations, some of the factors are relevant for other terrestrial and marine PES as well.

5.1 Institutional Conditions

Institutions are defined as formal as well as informal rules and conventions within a society in order to regulate and coordinate peoples' behaviour in given situations (Corbera et al. 2009; Muñoz & Holländer 2009). In general, institutions for environmental and resource management can be designed and implemented at various social levels, from the local community level to international regimes. Additionally, theses institutions usually interact with other formal (laws, property rights) and/or informal (traditions and habits) already existing institutions and can be affected by them (Corbera & Brown, 2008). Hence, the institutional conditions also include political and legal institutions.

Since PES schemes regulate and coordinate the compensation for the provisioning of a ES and therefore, the behaviour of the participants can be interpreted as new institutions (Corbera et al. 2009; Muñoz & Holländer 2009). In line with this argumentation is the statement of Corbera & Brown (2008, p.1957) which says that

> "*markets for Ecosystem Services are evolving institutions which attempt to enhance or change natural resource managers' behaviour in relation to ecosystem management through the provision of economic incentives. In theory, at least, these incentives should be generated by a self-sustained market in which consumers of Ecosystem Services channel financial resources to ecosystem managers*".

Depending on the type of Ecosystem Service e.g. public good, club good or CPR, as well as on the involved ES beneficiaries the potentially planned PES scheme requires certain institutional settings and regulations to guarantee its success. For this reason the institutional outline has to be set up in a way that it meets the demand of the service at hand with regards to the market-based mechanisms. Corbera and colleagues (2009) used a comprehensive multi-dimensional framework in order to shed light on the development and effectiveness of PES schemes by investigating the importance of institutional design, performance, and interplay. Parts of this framework will be explained more detailed further down. With regards to the effectiveness and fairness of the implementation of PES, the study additionally reveals the importance of capacity and scale issues. Capacity is important in order to "*design consistent schemes and projects and to generate the required trust among all stakeholders*" (Corbera et al., 2009, p.758). Geographical as well as political scale is also important since it determines the amount of stakeholders as well as the interference and interplay with other institutions. The study focused on Mexico's Programme of Payments for Carbon, Biodiversity and Agroforestry Services but concluded with several success factors which hold true for other PES schemes as well. In order to ensure the long-term effectiveness of new institutions for PES interventions it is important to evaluate the progress and drawbacks of existing PES schemes by ensuring the flexibility in procedural design, ongoing learning and continuous institutional adaptation.

Concerning the design of PES the study showed that it is essential to define the kind and nature of the ES for which the providers are rewarded as well as the establishment of appropriate and standardised evaluation of the services at stake (Corbera et al., 2009). Corbera and colleagues (2009, p.758) argue that "*the framework advanced in* [...]" their *"paper is useful to organise future PES research, regardless of PES schemes' geographical and governance scale*". In order to set the terms right for a successful PES the following institutional conditions have to be considered: ES beneficiaries as well as the ecosystem manager(s) have to be clearly defined and, moreover, have to agree to their participation[5] (Reis & Synnevåg Sydness, 2007). An effective intermediary organisation e.g. a NGO and/or community organisation which assures and secures the connection and communication between the service providers and the ES beneficiaries is important. It could assist in assuring the free flow of information between all participants at all time. However, intermediaries have to be accountable and actors' rights and responsibilities have to be clearly defined within contracts while assuring equal and even power relations (Corbera et al., 2009; Reis & Synnevåg Sydness, 2007).

In compliance with Ostrom's (1990) work, effective mechanisms that guarantee the compliance of all participants, through monitoring, conflict resolution and eventual sanctioning are needed. At the same time the accountability of the same executive mechanisms must be guaranteed. Since PES are subject to complexity and uncertainty the institutional regulations have to be flexible enough to adopt to possible changes.

Another important aspect especially in the developing countries is the proper definition of property and tenure rights (Reis & Synnevåg Sydness, 2007). One can distinguish between private and government ownership. Within a country various "*institutional structures of ownership exist*" and often "*complicate matters and must be fully considered when assessing local implementation*" (Murray et al. 2011, p.20 Box 1) In addition to that also different types of formal and informal uses come into play as for example land-use rights, resource use right or community rights, which are often linked to customary use rights. Hence, property are a necessary precondition for any PES intervention (Swallow & Meinzen-Dick, 2009) in order to ensure accountability of the ES providers.

Clements and colleagues (2010) showed that a higher institutional diversity leads to more sustainable outcomes of PES projects, however, also bear higher costs and reduce the proportion of payments received by the ES providers or local community. Nevertheless, in Clements et al. (2010) it is argued that PES programs can have positive influences on institutional settings as well as on natural resource management in general on a local basis. They state that

> "*PES programs can address two critical constraints, firstly by providing an incentive to reform institutional arrangements (for example clarification of property rights), and secondly by increasing the financial returns from collective management through provision of additional payments under conditions where sustainable extraction alone would not be profitable. At the village-level, the combination of a stronger institutional framework and payments leads to a greater local incentive for collective action, i.e. the village moves closer towards fulfilling the design principles articulated by Ostrom and others*" (Clements et al., 2010, p.1290).

5 see *five theoretical PES principles*

Corbera and colleagues (2009) applied an institutional analysis in order to determine the success and performance of several case studies. They claim that their research and results can be used for further research in this field and hence, will be adopted in part for deriving a theoretical framework which focuses on the institutional factors while including social and economic aspects as well. Using this information and knowledge for the analysis of Marine Payments for Ecosystems Services will shed light on their potential.

Institutional design: First of all it includes the reasons for the choice of PES as a policy tool. Secondly it is important to investigate who is shaping the rules for the PES scheme. The rules have to be effective and guiding in order to achieve the primarily set goals of the PES scheme. Pricing, additionality, conditionality and transaction costs are further aspects which have to be taken into account at this level.

Institutional interplay: This domain looks at how various institutions affect each other and their outcomes. Corbera et al. (2009) distinguish between two different forms of institutional interplay: symmetrical versus unidirectional and vertical versus horizontal. One speaks of symmetrical interactions if two institutions affect each other in a similar way, unidirectional interaction, on the other hand, refers to the event if one institution is affected more by the other one. Vertical interaction refers to the interplay of institutions from two different and distinct levels of social organization, whereas horizontal interaction takes place on the same level of social organization. Hence, at this stage potential synergies or conflicts between the PES scheme and other institutions can be identified, e.g. the existence of property rights.

The three domains outlined above are additionally influenced by two other analytical domains: organizational ***capacity*** and the ***scale*** of PES design and implementation. Capacity is defined as "*the availability of social, institutional and material capital to design and implement PES programmes so as to achieve their stated objectives*" (Corbera et al. 2009, p.747). Therefore, not only the effectiveness of the implemented PES arrangements are of interest but also the capacity and effectiveness of the institution which implements them.

Corbera and colleagues (2009, p.747) used the definition of scale developed in Gibson et al. (2000) which defines scale as "*the spatial, temporal, quantitative, or analytical dimensions used by scientists to measure and study objects and processes*". Therefore, from an institutional perspective it is of interest to look at similar institutional arrangements and their possibly related or comparable behaviour within different levels of social organizations. This is also true for PES and PES-like schemes, which are a combination of various interacting institutional arrangements while created and managed from different scales of socio-political organization (Corbera et al. 2009). It is therefore essential to understand how the scale of PES is affecting the other above mentioned aspects of institutional design and interplay.

According to Corbera and colleagues (2009, p.745) as well as in Clements et al. (2010) the above briefly mentioned and generally applicable design principles by Ostrom and others are meant to characterize successful and enduring institutions for natural resource management and were identified and further developed by Agrawal, 2002; Baland & Platteau, 1996; Ostrom, 1990. The above summarized literature for PES design principles can be combined with the various design principles in order to make them applicable to general PES interventions:

1. the PES intervention is devised and agreed upon by all participants;
2. clearly defined property or tenure rights
3. ES provisioning is easy to monitor;
4. non-compliance with the contract are enforceable;
5. sanctions are graduated;
6. adjudication is available at low cost;
7. monitors and intermediaries are accountable, legitimate and transparent to users;
8. institutions are devised at multiple levels; and
9. procedures exist for revising and further develop the PES scheme (Based on Corbera et al., 2009, p.745).

5.2 Biophysical Conditions

As the most important biophysical condition contributing to the overall PES success Reis and Synnevåg Sydness (2007) mention the unmistakable linkage between the provided ecosystem service and the associated activity, which provides this service. Furthermore, the provided services must be defined properly, made measurable, and must be subject to reliable scientific knowledge in order to create and ensure trust and credibility between the involved actors (Corbera et al. 2009, Reis & Synnevåg Sydness 2007). However, as Landell-Mills and Porras (2002) point out, this highly specific information is especially rare in the developing countries. Properly and clear defined boundaries help to specify the area, relevant ecosystem services, and involved stakeholders (Ostrom 1990; Reis and Synnevåg Sydness 2007). This clarifies who is participating, which ES are to be provided, and which management practices are linked to the provisioning of the specific ES. Measurability of the provided ES is another important aspect and has to be clearly linked to the changes in management practices. The ecological affects of the specific ES and related management practices have to be studied closely while performing a baseline study which indicates the current state of the ecosystem and the provided services. A PES intervention holds more potential for success if the ES and natural resources at hand are becoming less available and even scarce (Bracer, 2008)

Also the geographical dimension of a PES scheme plays an important role. The larger the scale and the further the distance between the beneficiaries and the ES providers, the higher the uncertainty of the gathered scientific data might be (Reis and Synnevåg Sydness, 2007). This factor is especially relevant for MPES since the spatial dimension is even greater and changes in management practices might be visible or accounted for in a different location often far away from their origin.

5.3 Social Conditions

As already pointed out above, the theoretical objectives of PES schemes do not include poverty reduction *per se* (Pagiola, 2007). However, if certain conditions are met PES can also contribute to reducing poverty. Most of these conditions, at least to a certain extent, are also important for the success of the whole PES scheme. Power structures cannot be neglected since they might be a reason for unequal negotiations between the landowners and the ES beneficiaries. Hence, equal power relations are essential in order to avoid unequal negotiations and socially unfair outcomes of the PES scheme. This aspect is also related to the inclusion of every stakeholder and their active participation. A certain level of social capacity and the ability of stakeholder to participate in the design of the PES intervention are of high importance. After the establishment of the PES scheme participants need to be involved in planning and training workshops as well as in future decision-making processes. Needs and priorities of local communities have to be respected and included within the establishment of the PES scheme in order to achieve more sustainable and long-lasting outcomes. PES interventions tend to be more successful if the affected communities are organized and have the ability to influence the outcome. It is therefore that a feeling of meaningful participation, legitimacy and stewardship is created (Morrison & Wendelin, 2010). The effective flow of information between all participants is important

1. in advance of the implementation of the PES scheme in order to achieve a broad participation, discussion, capacity building and decision making, and
2. after setting up the PES program to safeguard the ongoing acceptance and adequate functioning of the scheme through transparency (FAO, 2004).

Poor landowners or ecosystem managers must have the opportunity to participate in the PES scheme. At the same time it must be secured that the established PES scheme does not affect employment situation for landless people in a negative way. Especially with regards to water the access of poor people to the resource/service must be secured at all times (Reis & Synnevåg Sydness, 2007). But also other uses of the available ES have to be considered. Communities often depend on the local availability and presence of ecosystems and the natural resources they produce as their livelihood. Hence, the establishment of a PES intervention should consider and be complementary to the community’s livelihood and safeguards the communities’ access to it (Morrison & Wendelin, 2010). Cultural resistance is another criterion which might affect the outcome or even the development of a PES scheme.

5.4 Economic Conditions

Especially with regards to ecosystem services and their different market characteristics it is important to pay attention to the possible creation of a market and whether it will take off and can be sustained on a long-term (Landell-Mills & Porras, 2002).

> "*The development of an appropriate financing platform is key in the establishment of a successful PES system. The objective is to generate a continuous flow of financial resources into the system to fund payments over the long term*" (Mayrand & Paquin, 2004 p.26).

Here it is essential to make sure that the transaction costs of the intended PES scheme do not exceed the expected benefits. Also the opportunity costs for the service providers should not be higher or at least should not perceived to be higher than the actual benefits. Hence, the willingness to pay on the side of the ES beneficiaries has to be sufficient to cover the opportunity costs. Additionally, the buyers must perceive that their financial contributions are well spent and it is essential that the funds are used for the problems at stake e.g. ecosystem conservation (Reis & Synnevåg Sydness, 2007). With regards to the long-time perspective of a PES project a sustainable way of financing has to be found, so that the PES project can be operated and maintained with local funds and is not dependent on external financial aids (Corbera et al., 2009).

Therefore, it is important to establish a sustainable financing framework where ES providers are compensated by ES beneficiaries on a long-term base, which includes and supports sustainable resource management (Corbera et al., 2009). Such a sustainable financing framework has to cover several costs associated with the establishment of a PES scheme:

1. The establishment of the system itself including baseline study and other scientific research, improvement or creation of ineffective or missing institutions, stakeholder involvement, meetings and training.
2. Continuous compensation or payment of the land owner/ manager.
3. Other continuous management costs such as monitoring, rule enforcement and transaction costs.

PES schemes tend to work best if the established contracts and payments between ES beneficiaries and providers are flexible in order to adjust to changing conditions, while being ongoing and without an expire date (Mayrand & Paquin, 2004).

5.5 Synthesis of Success Conditions and Framework

The above described success conditions for PES will be the foundation and part of the development of the preliminary assessment framework in order to elaborate on the potential of and preconditions for marine payments for ecosystem services. Even though, the conditions and characteristics of ES within marine and coastal environment differ from the terrestrial counterpart, some of the success conditions and lessons learned may be used for the implementation and creation for MPES.

From the literature review and from the analysis of various case studies several generally valid success conditions could be derived. As already mentioned above the success conditions were categorised into institutional, biophysical, social, and economic, conditions, whereas the focus lies on the institutional preconditions. However, this does not imply that the institutional conditions are the most important ones. It is important to point out that especially with regards to sustainable management of CPRs proper institutional settings are essential in order to avoid resource degradation as a result of resource mismanagement as described in Hardin's “The Tragedy of the Commons” (1968). Hence, and in accordance with Ostrom's (1990) work several institutional principles are essential preconditions for successful CPR management. These design principles (see chapter 5.1) are additional preconditions with regards to sustainable management of CPRs and are important for the institutional design of PES interventions (Corbera et al., 2009).

The literature-derived and above mentioned independent variables indicating the success conditions of PES schemes built the preliminary assessment framework. This framework contains the objective to outline and summarize essential preconditions and design principles which have to be considered before and during the planning and creation of potential MPES schemes in order to set favourable conditions for the successful implementation.

Even though PES schemes are always site specific several success conditions can be summarized. One of the most fundamental preconditions is secure land tenure and/or user rights. Secondly, suppliers and beneficiaries must be clearly defined and need to have an incentive in participating in the payment scheme. Furthermore, the provided ES hast to be defined and made measurable. Additionally, the activities used for the active supply of the ES at stake have to be clearly linked to the provided ES. Hence, the ES supplier must be able to deliver while the ES receiver must be able to pay for the delivered service. The payment must exceed the foregone opportunity costs of the ES suppliers. Co-operational institutions e.g. associations, councils or farmer groups on both the supplier and beneficiary side are of great advantage if not essential for the reduction of transaction costs. A supportive government which also respects the participant’s decisions as well as a supportive legal and regulatory framework is important. Generally the institutional rules must be flexible enough in order to adapt to changing conditions. Capacity building, as well as frequent and meaningful participation can generate and secure legitimacy and trust among all stakeholders. Prevailing power relations have to be taken into account and the inclusion and participation of poor stakeholders must be secured.

5.6 Preliminary Assessment Framework

With regards to the objective of this study which focuses on the contribution of MPES to marine and coastal conservation and protection, it is essential to collect and determine several assessment criteria in order to create a comprehensive and comparable preliminary assessment framework. It is called preliminary assessment framework since it is based mainly on data and knowledge from terrestrial PES schemes and has its roots in the literature and several case study analyses. The purpose of the preliminary assessment framework is to summarize and point out under which conditions and design strategies the implementation of an MPES will most likely be successful and hence contribute towards marine and coastal conservation and protection. The applicability of this framework will be tested on the different case studies introduced in chapter 6.

To set the stage for the framework for the potential success of MPES it is important to keep Sommerville and colleagues' (2009) conceptual framework and revised definition of PES in mind, because it set the essential criteria of conditionality and positive incentives. These two criteria build the essence of a PES intervention and hence are the preconditions of any PES or MPES scheme. The other two essential considerations listed are (1) additionality and (2) the institutional context. The latter is a complex evaluation with regards to the framework and hence is a key factor for success in its own. Additionality is often used as the measure and degree of effectiveness (Wunder, 2007; Sommerville et al., 2009). However, since additionality is based on complex and often uncertain scientific data concerning the baseline study, service provisioning and eventual leakages, it is very difficult and time intensive to estimate. Due to its complexity and other reasons, using additionality as a determining factor is infeasible and the degree of additionality is not necessarily linked to the success of an intervention. Nevertheless, additionality has an essential role and should be used as a guiding consideration in the valuation of PES schemes. Hence, a complete analysis of additionality will not be part of this preliminary framework but is still a prerequisite of every PES or MPES scheme.

Since the marine version of payment schemes for Ecosystem Services has its roots in terrestrial PES in which several practical experiences have been collected already and since PES come additionally with an extensive literature body, success conditions for PES will be one part of the new framework. On these data the preliminary framework will be based. The various criteria from the framework will then be used on the preconditions and as to when, where and under which conditions the implementation of a specific type of MPES might work. Last but not least, the type of MPES scheme is essential and highly depends on the preconditions and type of ES at hand. The following list is a summary[6] of all independent variables affecting and influencing the effectiveness and success of PES in general. Therefore, they can be applied to MPES interventions:

6 The previous chapters already discussed each cause more detailed

- **Institutional context**
    - Supporting government, legal and policy environment.
    - PES project harmonizes with other regional and/or local institutions, natural management programmes (institutional interplay).
    - Presence, role and accountability of intermediary/local institutions.
    - Clear property, tenure, and/or user-rights.
    - Effective facilitating structures (contracting, administration, buying and selling services, enforcement & monitoring).
- **Biophysical context**
    - Clearly defined system boundaries.
    - Clear linkage between provided ecosystem services and the associated activity.
    - Measurability of the provided ecosystem services based on reliable scientific knowledge.
- **Social context**
    - Active participation of stakeholders.
    - Organizational capacity of communities.
    - Equal power relations.
    - Trust and legitimacy among stakeholders.
    - PES intervention complementary to the community's livelihood.
- **Economic context**
    - Sufficient and sustainable funding.
    - Transaction costs do not exceed the expected benefits

6 Case Studies and Potential MPES Schemes

This chapter of the research project will look at two different MPES case studies (Individual Transferable Quotas and Marine Protected Areas) and one potential MPES scheme (Mangrove forests as potential blue carbon sinks). Each subchapter will first explain the theory and specific concept of the MPES intervention before giving a short introduction on the specific case. After the introduction of the case study follows the linkage between the case and the MPES theory before applying the above developed preliminary assessment framework. Each subchapter will conclude with the found results.

6.1 Individual Transferable Quotas

Fishery management and the global problem of overfishing are directly linked to common pool resources management and property rights. Most fisheries can be defined as a common pool resource (characteristics of low excludability but high rivalry of consumption) and hence lack clear property rights. Already 50 years ago the economists Gordon and Scott identified and linked overfishing to the 'common pool' problem and furthermore, predicted "*that open access would lead to excess fishing effort, dissipation of rents, and inefficient depletion of fish populations*" (Newell et al., 2002). In common pool fisheries all fishers compete for their share of the total harvest. This leads to a race for the fish that drives fishers to invest in expensive fishing gear, faster vessels or other technology (Grafton, 1996). Nevertheless, this overcapitalisation does not affect or increase the total return from the fisheries but rather changes the individual share of each fisher from the fishery. This leads to a continuous increase in fishing and harvesting costs. In order to avoid or at least decrease these economic inefficiencies, the race for fish and the overcapitalisation it is essential to control individual outputs of fishers and not only the total harvest (Grafton, 1996). However, during the last decades several counteractions have been developed and some are already being applied. They deal with the elimination of the common property rights problem; the so called property rights based approach (Arnason, 2002, p.1f). Rather recent policy measures or, to be more specific, '*market-based approaches*' for the improvement of fisheries management and challenge of overcoming the above mentioned common pool problems, aim at combining and improving economic measures with conservation objectives.

Approaches to overcome the lack of property rights are territorial use rights, Individual Catch Quotas and community fishing rights. Additionally and with regards to MPES these approaches of fishery management fall under the term 'catch shares'. Examples of catch shares are individual transferable quotas, territorial user rights, co-operatives and community quotas. According to Forest Trends and Katoomba Groupe (2010) individual transferable quotas (ITQs) can be regarded as MPES schemes (Melnychuk et al., 2011).

During the last few years ITQs received more and more attention and have been widely implemented with the aim to overcome common property problems within fisheries (Arnason, 2009).

The basic idea behind ITQs is to limit fishing efforts by setting a Total Allowable Catch (TAC) which is then divided and allocated to the various participants e.g. fishing operations. These quota shares can then usually be sold and/or leased on the market (Newell et al., 2002; Branch, 2009). With regards to the common property rights problem it is important to state that the "*[i]ndividual catch quotas attempt to solve the common property problem not by defining property in the fish stocks themselves but by allocating individual harvesting rights from these stocks*" (Arnason, 2002, p.1f). Ideally the allocation of share and a guaranteed share of the TAC should set more incentives for sustainable fishing practices and reduce the danger of overfishing fish stocks as well as solving the problem of a race to fish[7] by securing exclusive and long-term access to fishing opportunities (Melnychuk et al., 2011 ; Newell et al. 2002; Arnason 2005). Participants and share holders have therefore got a direct interest in the preservation and sustainability of the fishery since their value increases with health and size. Especially when the ITQs are permanent they fulfil the characteristics of complete property rights (Arnason, 2002). Therefore, the common and standard economic theory should apply and will ideally steer fishery towards a more efficient use. Furthermore, due to the possibility to trade shares on the market, less efficient fishing operations will sell their shares to more efficient operations due to the economical incentives (Batstone & Sharp, 1999) and thereby additionally improve *'input controls' (regulating fleet capacity, days at sea, etc.)* (Newell et al. 2002,; Branch, 2009). Hence, "*this should both reduce excess capacity and increase the efficiency of vessels operating in the fishery*" (Newell, 2003).

The theoretical approach of ITQs has been supported by empirical work from world wide examples, which show that carefully and appropriately designed ITQs lead to an increase in economic rents from a previously common property fishery (Arnason, 2009). It is, however, important to note that the theoretical potential of ITQs to provide a solution to the common pool problem is highly dependent on the existing and working competitive market for the trade of the fishing quotas which must convey appropriate price signals. The price signals of the quota not only resemble important information for the participants and for the expected profitability of fishery and hence for future decisions, but also for decision- and policy-makers with regards to the biological and economic health of a fishery at hand (Batstone and Sharp, 1999, Newell et al., 2002).

6.1.1 Case of ITQs in New Zealand

Through the establishment of exclusive economic zones (EEZ) during the end of the 1970s parts of the ocean and coastal commons became property of the bordering countries. Around the same time, in 1986 to be precise, New Zealand (NZ) and Iceland were the pioneers in establishing ITQ based management systems within their EEZ. Inspired by Iceland and New Zealand, 15 countries followed them and by now ITQs are

7 In situations where fishing is allowed until a fishery-wide TAC is reached.

used to manage more than 80 different species worldwide (Newell, 2004, p.437f). Nevertheless, no other country is using ITQs as advanced and extensively as New Zealand is doing right now. In New Zealand, ITQs are used for the management of all significant commercial species (Lock & Leslie, 2007).

Hence, the ITQ example from New Zealand provides both space- as well as time wise the longest and most extensive experience with ITQ based systems and with it delivers the most comprehensive set of data in this field. This fact is also apparent within the literature which is an important criterion for the selection of New Zealand as a case study for ITQs as an MPES scheme (Newell et al., 2002).

New Zealand with a population of less than 4 million people and the status as a developed country highly depends on its primary production (e.g. farming, forestry, eco/nature-based tourism and fishing) (Hughey, 2000). New Zealand's EEZ is one of the worlds largest and encompasses a total area of 1.2 million square nautical miles. Even though New Zealand has a relatively small population and a comparatively huge EEZ, most of NZ fisheries suffered from intensive resource extraction. This fact, however, can also be related to the relative unproductiveness of the fishing area. As many nations with a high degree of fishing activity, also in NZ typical symptoms, created through weak fishery management mechanisms, like resource depletion and degradation and over-capitalisation in the fishing industry emerged (Hughey, 2000). As a consequence[8] NZ introduced and adopted a new form of fisheries management and established their QMS. Since then, within the waters of NZ commercial fishing is unsubsidised and based on a sustainable approach and objectives (Batstone & Sharp, 1999). The primary reason of the establishment of the QMS was to treat and handle fisheries as an essential economic resource that should be managed in a sustainable manner.

With the implementation of the ITQ system after primarily involving the fishing industry in the decision making and development process, New Zealand set the environmental objectives in a way that they would reach a state of sustainable fishery in the long run. The target is to reach a so called Maximum Sustainable Yield (MSY) which defines and regulates the Total Allowable Catch each year (Newell et al., 2002). The established ITQ system is one part of an overall regulating and encompassing Quota Management System (QMS), which also regulates and sets standards for fishing areas and techniques as well as for fishing seasons.

Looking at NZ QMS establishment it is important to distinguish between inshore and deep-sea fishery. As early as 1983 New Zealand introduced a so called Deep-Water Trawl Policy which was basically a property rights-based fishery management tool. The Deep Water Trawl Policy was based on the ITQ approach and allocated deep-sea fishing quotas to operating fishing companies. This approach quickly turned into a success. Nevertheless, NZ still had to deal with fish stock degradation and over-capitalisation and hence established an all-encompassing ITQ system that included all major fisheries (Hatcher & Pascoe, 2002, p.47f).

Quotas were first based on fixed tonnage and the holder was allowed to fish in a specific region and from a specific stock. The idea behind this regulation was that the

8 And beneath other reasons i.e. decentralisation

government had the option to buy back a certain amount of non-harvested tonnage of a species from a quota holder for conservation purposes. The intensive financial means necessary forced the government to adjust the ITQ system and convert the amount of tons to a percentage based approach. Today, each quota holder has a basic property right for a permanent and annual share in the TAC of a specific species.

The created market for the tradeable quotas was established and secured by the permanent quota shares and is furthermore perfectly divisible and transferable. The only prerequisite is that the owner has a New Zealand nationality or the owning company is at least to 75% in New Zealand ownership. Additionally, the government set an aggregation limit that reduces the total quota owned by a single instance to 35% of a specific fish stock and a total 20% within an inshore quota.

Since 1840 traditional Fishing interests of Maori have been guaranteed by the New Zealand government within the Treaty of Waitangi. Furthermore these rights have been acknowledged in the Fisheries legislation since 1877. With the introduction of the QMS the Maori feared their exclusion from fishing practices and contested their traditional rights. Several years of negotiations had to follow before the government realised that the inclusion of Maori customary and commercial fishing is in fact an integral part of fisheries. Without the recognition of Maori fishing rights the implementation of the QMS would not have been successful. The final settlement of the conflict settled all commercial Fishing claims and was written down as the Treaty of Waitangi (Fishery Claims) Settlement Act 1992. Today the Maori as collective own 23% of quota (Batstone & Sharp 1999).

In 2001 New Zealand introduced another regulation, called Annual Catch Entitlement (ACE) to their QMS. ACE is a freely tradeable fishing right which is temporarily restricted to a "fishing year'. Hence, the owner of an ACE is permitted to fish the equivalent of a specific stock within this time-frame without actually holding the quota. Each quota owner receives an equivalent level of ACE to their owned quota share. The ACE can then be traded on a market (online) with other entities. It is also possible to acquire the matching ACE after landing more fish then actually entitled to according to the owned share of quotas. In order to comply with the objective of sustainable fisheries management the total amount of ACE is equal to the TAC (Stewart & Callagher 2011).

6.1.2 ITQ System as an MPES Scheme

This section will first elaborate on the theory of the New Zealand ITQ system and its characteristics as an MPES scheme. Secondly, according to the developed framework the institutional, political and legal, biophysical, social, and economic context of New Zealand QMS will be analysed.

Forest Trends and The Katoomba Group (2010) listed ITQs as a type of MPES intervention which is based on the creation of a formal market that enables open trade between buyers and sellers. The market is based on a regulatory cap with regards to the provision of the specific ES. Hence, ITQs resemble a real market for ecosystem services and in fact are an example of a true MPES market.

With regards to the definition of ITQ systems as an MPES scheme the criteria of **conditionality** is fulfilled with the purchase of a fishing quota and the specific share of the TAC. The individual quota indicates how much fish of a specific species the permit holder is allowed to catch. These shares can also be sold from one holder to another, in which case the buyer pays for the fishing quota and the initial holder gives up his right/reduces his or her total amount to fish while receiving the financial compensation. Therefore, the ES is only allowed to be harvested/fished with the acquired fishing quotas. It is therefore that the involved buyers and sellers are clearly defined. With this conditionality the ecosystem service – tonnage of fish or percentage of TAC – is clearly defined and its price is established on the market. Hence, an externality is internalized. This externality is equal to the market price of the quota is internalized since extra quotas have to be purchased on the market or the decision not to sell the allocated share incorporates the same market price which they resigned. The conditionality of this MPES scheme is directly linked to measures of the ES itself but not to other environmental indicators.

Positive incentive: Off-shore fisheries can either be characterised as a public good with a regulated use or can be state-owned and therefore hold the property rights to the fish grounds. By acquiring fishing quotas the fisherman or the fishing company legally holds a defensible right to land a specific amount of fish in a specific time-frame. Through sustainable management of the fishery the quota holder contributes to the health and productivity of the resources and hence secures future income. This incentive also holds with the increase in economical value of the quota share with the increasing health and productivity of the ecosystem. Two positive incentives can be linked to ITQ managed fisheries:

1. financial compensation for leasing or selling quota rights and
2. higher economic revenue while securing a more sustainable management of the fisheries and hence affect the motivation of the stakeholders to engage in the scheme.

Additionality: Without setting and limiting the TAC to a more sustainable level and distributing secure fishing rights, fisheries would have been managed less sustainable and with a lower economic revenue as seen in other more traditional fishery management approaches (Grafton, 1996; Kerr et al., 2003). From an empirical point of view, several studies already confirmed the economic additionality of ITQ managed fisheries, whereas ecological additionality is still subject to some uncertainty and highly depends on the overall performance and effectiveness of the programme implementation. Additionally, the regulating body has less expenditures of public funds, apart from some administrative costs, while achieving the same goals as other top-down regulating measures e.g. buying out excessive fishing capacity or paying fishing companies for reducing their fishing practices (Spergel and Moye, 2004).It is therefore safe to say that even if ITQs do not resemble a first best solution to common property fishery they can still be regarded as a desirable fishery management approach as long as the results are better than the ones from other alternatives and more traditional management tools (Grafton, 1996).

6.1.3 Institutional Context

Since the ITQ system was planned, introduced and managed by the New Zealand government and is backed up by the legal system of New Zealand it is save to say that a very supportive government is in place and that the further development of the programme and adaptation to new insights will be carried out by the government.

Under the Fisheries Act of 1996 the Ministry of Fisheries is responsible for the sustainable management of all fishery resources, whereas the protection and conservation of marine mammals and reptiles, seabirds, and other protected species (e.g. corals and sharks) is the responsibility of the Minister of Conservation (Department of Conservation, New Zealand). The introduction and recognition of the EEZ system on an international scale, based on international law, enabled the path for the implementation of an ITQ system since countries had the exclusive jurisdiction over their 200-mile zone. Since the implementation of the broader Quota Management System (QMS) which is the overarching institution and incorporates the ITQ system, in 1986 it has been subject to several changes and improvements. Apart from setting the legal and administrative foundation for the ITQ system the QMS also regulates, plans and controls fishing areas, seasons and techniques (Lock & Leslie, 2007). The QMS and with it all quota rights which are actual property rights due to their legal status and hence protected by New Zealand law have full legal support and the policy environment has been adjusted to the new QMS. It is complemented by the Ministry of conservation and a conservation services levy fee as described above. However, the presence of an EEZ which is unencumbered by other or foreign fishing industries was a precondition for the trust, legitimacy, and legal framework and hence contributed to the success of the scheme.

As stated above the Ministry of conservation has the responsibility to conserve and protect certain marine animals (as defined in the Wildlife Act 1953 and in the Marine Mammals Protection Act 1978). In order to reduce and manage the impacts and effects of commercial fishing practices on protected marine species New Zealand's Government implemented a fisheries conservation services levy in 1995. The purpose of the collection of the levy is used for financial support for fisheries research, by-catch control and compliance as well as administration of New Zealand's QMS. The fee is being set, adjusted and collected by the Ministry of Fisheries and then transferred and used by the Ministry of Conservation. However, the conservation services levy is not being paid by all fishers but rather by the part of the fishing industry which creates adverse effects of commercial fishing on protected marine species. An annual adjustment of the levy incentivises the responsible fishing sector to adjust their practices in order to reduce their impact on the protected species and hence reduce or negate the need for conservation services levies. This kind of institutional interplay takes place on a symmetrical and horizontal level, since each institution was implemented and administered by governmental agencies while affecting each other. The two schemes have a clear synergy and support each other while setting incentives, usually economic and financial ones for marine protection and conservation.

After initialisation of the QMS the government can be regarded as an intermediary. It is no longer actively involved in the trading of quotas but rather for establishing the policy environment, safeguarding of legal rights, monitoring, and enforcement.

Within the QMS each and every quota holder is entitled to a fixed percentage of the TAC within a fixed area equivalent to the owned quota. These quotas enjoy a full legal status and resemble a fixed, long-term property rights over the specific fisheries. Furthermore, these property rights are freely transferable to other entities.

Even though the government initially created a centralized trading platform it was quickly replaced by informal bilateral trading and by brokers. Trade, monitoring, compliance and rules are all subject to federal law and facilitated by the government. There is a legislative restriction on how much percentage (not more than 20 %) a single quota-holder is allowed to own of a given species within an inshore area. For deep sea species the limit is set to a maximum of 35 percent per quota holder (Spergel and Moye, 2004). Also, the aggregation of quotas by foreigners is limited. The trade of quotas is no longer regulated by the Ministry; however, newly traded quotas have to be reported to the Ministry before they can be used. Landed fish can only be sold to licensed fish receivers and has to be in line with the acquired quotas. New Zealand's legislation requires that all harvested fish has to be landed, reported and accounted for, regardless of their economic value or management status (included in the ITQ system or not) (Bremner et al., 2009). In order to ensure the compliance with the ITQ system a comprehensive reporting system is in place. It tracks the landed fish from vessels to licensed fish receivers in the form of export records. In addition to the documented landings, the compliance is enforced by an at-sea surveillance program which consists of governmental on-board observers on selected vessels and military aircrafts. Non-compliance and misreporting is regarded as a criminal offence and will be legally prosecuted (Kerr et al., 2003). It is also important to state that only after the consultation of the fishing industry the NZ government implemented the QMS. On of the most important steps is to set the TAC each year[9]. This is done by the Minister of Fisheries. The process includes various stakeholders like scientists, the Ministry, industry, and environmental groups and has the overall objective to reach and secure the maximum sustainable yield.

6.1.4 Biophysical Context

The physical boundaries of the MPES scheme are provided by the establishment of the EEZ and are the foundation for the spatial divertification and implementation of the ITQ system. Additionally, the quotas specify the type of ES (fish species) and area in which the holder is eligible to harvest.

New Zealand faced severe problems of overfishing, resource degradation and over-capitalisation before the implementation of the ITQ system. Among other reasons this was one of the main factors influencing the adaptation of the QMS. The government's and fishing industry's realisation of the bad condition of fish stocks eased the

9 For more on TAC and MSY see Lock and Leslie, 2007

introduction of the ITQ programme. In general a baseline study with regards to fisheries involves a high degree of uncertainty. Nevertheless, it can be stated that the implementation of the ITQ system has more advantages and better outcomes then other, traditional fisheries management mechanisms. If, however, the current application of the ITQ programme actually lead to a more sustainable fishery is still subject to debate, even though several articles indicate a stabilisation of many fisheries (Batstone and Sharp, 1999; Arnason, 2002; Branch, 2009; Stewart and Callagher, 2011). Nevertheless, by selling owned quotas the initial holder gives up his or her right to the equivalent percentage of the TAC. This ensures a clear linkage between the provided ES (fish to be harvested) and the abandonment of the right to harvest.

Nevertheless, several external factors influencing the development of the ITQ system have to be considered as well. An example here might be fishing practices outside New Zealands EEZ. Additionally and due to the migratory characteristics of fish and the parallel establishments of other environmental management schemes e.g. MPAs, it is difficult to isolate the success conditions completely.

6.1.5 Social Context

A general problem of ITQ based interventions is the primary allocation of quota and fishing rights. Usually the initial allocation of quotas is gratis and is based on historical data and participation of fishers and fishing companies (Grafton, 1996). In the case of New Zealand this was the same, which however added a petition mechanism for fishers in order to adjust the initial allocation (Newell et al., 2003). Even though the initial allocation excluded several small fishers, over the past years the average number of participants or, to be more precise, quota holders, has been about 1,500. Kerr and colleagues (2003, p.6) state that "*a healthy number* [of small players] *remained in every important market*". Additionally, New Zealand's government set a limit of aggregation in order to avoid large accumulations of quotas by few big fishing companies. New Zealand's fishing industry consists of two different sectors – (1) a deep water sector and (2) an inshore fishing industry. The first is characterised by few but large fishing companies which are additionally in close cooperation with processing companies. The latter of the two industries, the inshore sector, is composed of smaller fishing companies and small scale fishers (Yandle, 2003). According to Hughey and colleagues (2000), the allocation of quotas led to a large number of fishers that were geographically dispersed and often used low fishing technology.

However, with the implementation of the ITQ system, its design and the incentives created from rights-based fisheries management encouraged and finally led to several stakeholder cooperation initiatives (Batstone and Sharp, 1999; Hughey et al., 2000). By 1997 twenty-one different organisations had been established with different objectives and structures aiming at cooperative quota management, research on fish stocks and TAC negotiations. This new development of stakeholder cooperation and active participation adds to the overall legitimacy, economical and biological effectiveness as well as stability of the ITQ programme.

With the introduction of the ACE regime a reduction in concentration in ACE for important inshore species could be observed. Hence, the new ACE regime increased the participation of small-scale fishers which had been one of the downsides after the initialisation of the QMS (Stewart & Callagher, 2011). Still it seems that big fishing companies hold greater power, especially through financial assets, to accumulate quotas (even though the aggregation is legally limited) and to influence the future development of the QMS.

The positive response from the industry, overall compliance with the ITQ system and the high degree of trade indicate a certain degree of trust among the different stakeholders. However, monitoring and enforcement are still necessary and essential in order to avoid problems of free-riding. The legal framework in which the QMS is embedded contributes to the overall legitimacy and acceptance of the system.

The inclusion of Maori and their fixed rights to quotas indicate an attempt for a fair involvement of different stakeholders (even though only after some dispute settlement), additionally the legal aggregation limit secures a more diverse group of stakeholders, furthermore, due to the introduction of ACE a higher percentage of small-scale fishers could be achieved. These characteristics are essential preconditions for the persistence and success of such a scheme.

6.1.6 Economic Context

As pointed out by almost all scholars that are dealing with New Zealand's QMS the economical benefits created through the introduction of the ITQ system are substantial. After the initialisation of the programme the government needed only few financial resources to sustain the QMS. Additionally, the government stopped subsidising the fishing sector and thereby reducing its expenditure as well. The initially created market has been taken over by bilateral trading and brokers.

Especially after the introduction of the ACT regime market activity increased and suggested that transaction costs were relatively low. The increased profitability of fishers made the whole programme economically attractive and hence gained support from the industry.

6.1.7 Results of NZ ITQ System as an MPES Scheme

The implementation of New Zealand’s QMS can be regarded as successful in comparison to the initial state of the fisheries and the fishing industry. This has been stated several times by different scholars within a broad array of literature (Batstone and Sharp, 1999; Arnason, 2002; Branch, 2009; Stewart and Callagher, 2011). Especially from an economic point of view the ITQ and ACT regimes had a positive contribution. Fishing efforts and costs have been reduced while the profitability increased after the implementation. Low trading transaction costs enabled the free trade of quotas which favour more efficient fishing practices and technology.

From an ecological point of view the implementation of the ITQ system had positive effects on targeted species. The effects on non-target species, however, have not been investigated in studies so far. Through the creation of secure property rights and the incentive to increase quota values due to healthy fish stocks, stewardship with regards to the fisheries among fishers was encouraged and created positive ecological benefits (Branch, 2009). The link between ecological health and financial returns for the quota holders encourages sustainable fishing but also compliance, enforcement and reduction of illegal fishing. However, an essential and determining precondition is the correct adjustment of the TAC by the government and is based on reliable scientific data. In addition, monitoring and the enforcement of rules are still important aspects. The stable political conditions and the relatively small size of the fishing sector make these aspects more achievable and effective.

In the case of the ITQ system in New Zealand all the listed success conditions were met, even though some were met in more ways than others and some limitations still exist. Additionally, several key findings can be isolated, which in particular influenced the overall successful implementation of New Zealands ITQ system:

1. A clear definition of the service (fixed percentage of a fish species of the TAC) and the applicable boundaries (fisheries management areas).
2. Property rights are clear, secure and embedded within the national law.
3. A stable and legitimate political system which provides the necessary institutional context.
4. Institutional flexibility and adaptations in order to react to short comings (e.g. the introduction of ACE regime).
5. Ongoing and secure economical benefits for all or most of the participants.
6. Effort to increase the amount of participants by giving a fixed share of quotas to the Maori and introducing the ACE regime.

6.2 Blue Carbon Sinks

Together with the rising understanding of the impact climate change has on the environment and human well-being comes the increasing awareness and interest of how anthropogenic emissions of greenhouse gases are caused and affected through deforestation, management of natural carbon sinks, and related policy measures. Additionally, it is of interest how the above mentioned factors influence the attribution of an economical value on carbon and the possibilities how this can be the foundation for the establishment of economic incentives to reduce anthropogenic emissions (Laffoley & Grimsditch, 2009).

The emissions of small particles and volatile organic compounds (VOC) through the process of burning wood, biofuels but mainly fossil fuels, also called "*black carbon*", are one of the mayor contributors to anthropogenic induced global warming besides the CO_2 emissions - called "*brown carbon*" - from conventional industry and energy use. Hence, most policies, with regards to reducing greenhouse gases focus on the reduction of CO_2 and black carbon. All carbon that is removed from the atmosphere and then stored in plants and soils through the natural process of photosynthesis is called "*green carbon*" (Nellemann et al., 2009; Yee, 2010). Even though the reduction of emissions from burning fossil fuels is essential in order to counteract climate change and through protecting and restoring natural ecosystems the mitigation of global warming can be preceded.

A forest is an example of a natural carbon sink that captures and stores atmospheric carbon very effectively over long periods of time and binds it as biomass or in sediments and soils. Nellemann and colleagues (2009) defined the term carbon sink as

> "*any process, activity or mechanism that removes a greenhouse gas, an aerosol or a precursor of a greenhouse gas or aerosol from the atmosphere. Natural sinks for CO_2 are for example forests, soils and oceans*" (Nellemann et al., 2009, p. 16).

However, through land use changes as for example deforestation the stored and fixed carbon e.g. as biomass or within soil layers is being released as CO_2 back into the atmosphere. Additionally, the future potential of the forest to act as a carbon sink, that is to capture and store CO_2 is reduced tremendously. Hence, and in order to reduce the release of additional carbon and to guarantee the future potential of natural carbon sinks efforts usually focus on decreasing deforestation on a terrestrial scale e.g. temperate and tropical forests (Murray et al., 2011). Several policies and conservation methods are used in this field but are beyond the focus of this research paper. However, as also pointed out in previous chapters, PES schemes are one possible and rather new approach to combine economic incentives with environmental protection. The terrestrial version of PES e.g. tries to reach more conservation and protection of forests by the establishment and introduction of payment schemes for carbon storage or biodiversity conservation.

It is important to point out at this stage that the current Emission Trading System of the European Union (EU-ETS) does not include green carbon. Even though the Kyoto Protocol's Clean Development Mechanism (CDM) does include credits for forests as carbon sinks, practice proved that - besides other reasons - mainly due to low prices and demand, CDM turned out to be in fact a black carbon mechanism (Nellemann et al. 2009). Only recently the importance of green carbon as a mitigating agent for climate change has been recognized as it was in 2009 in Copenhagen during the United Nations Framework Convention on Climate Change Conference of the Parties (COP). Several mechanisms that take green carbon into account like REDD+, Forest Carbon for Mitigation and others exist.

Nevertheless, not only terrestrial ecosystems are important with regards to carbon sequestration and storage as recent studies indicated but also coastal and marine ecosystems. "*The ocean is the largest carbon sink on Earth* […]" but until recently got little attention concerning its role for mitigating global climate change (Laffoley & Grimsditch, 2009, p.1). It is estimated that about 55% of all carbon is being bound within the living organisms of the world's oceans. At most importance with regards to carbon sequestration and storage are mangroves, marshes, coral reefs and seagrasses – the captured carbon within coastal and marine ecosystems it is called "*blue carbon*".

The oceans are responsible for storing and cycling more than 90% of the Earth's CO_2 (40Tt). Less than 0.5% of the oceans sea bed is covered by vegetated habitats like mangroves, salt marshes and seagrasses. In comparison to the biomass on land this adds up to only 0.05%. Nevertheless the annually captured and stored carbon by coastal and marine ecosystems is equivalent to the amount stored by land vegetations. Hence, ocean's vegetated habitats are among the most productive and effective carbon sinks on Earth. (IUCN, UNEP, 2009,). Unlike rainforests, which are able to store carbon for decades and even centuries, carbon stored in the oceans will remain there for millennia.

However, the rate of degradation and destruction of these coastal and marine ecosystems is about 5-10 times faster in comparison to rainforests (Nellemann et al. 2009). Due to habitat conversion like aquaculture, agriculture, deforestation, as well as industrial and urban development, it has been estimated that the total loss of the three key coastal and marine ecosystems, namely seagrasses, salt marshes, and mangroves is about 29%, 35% and up to 67%, respectively[10] (Murray et al., 2011). The proceeding degradation of coastal and marine ecosystems is severe from an ecological point of view but at the same time the exploitation of these ecosystems is of high economical importance (Murray et al., 2010). This effect is enhanced through failures of the market which lack to integrate and assign economic value to ES and especially those produced by marine and coastal ecosystems. It is therefore that the responsible managers of an ecosystem at hand often decide about the clearance of the ecosystem in order to sell the ES on the marketplace. This leads to a high degree of ecosystem degradation and destruction. Besides a high variety of essential goods and services as well as being a fundamental part of many livelihoods, the key coastal and marine ecosystems and habitats also play an important role in the global carbon cycle and store substantial amounts of carbon (Laffoley & Grimsditch, 2009; Murray et al., 2010). Through the clearance and disturbance of these ecosystems the captured and stored carbon is being released into the atmosphere as additional greenhouse gases (GHG). Apart from the potential to act as carbon sinks marine and coastal ecosystems have further important advantages and values:

- Filtering capacity of terrestrial nutrients and pollution, which allow the growth and existence of valuable coral reefs.
- Coastal ecosystems, especially mangroves and coral reefs, offer coastal protection against storm floods, waves, and even tsunamis and hence protect coastal communities.
- Intact coastal ecosystems like mangroves and salt marshes reduce coastal erosion.
- Additionally these ecosystems are the bases for coastal communities for the production of food, offer nurseries for certain fish stocks, and are the source of

10 Loss of habitat in comparison to their historical area, which original extent is still subject to uncertainty.

economical revenue through commercial fishery (Laffoley & Grimsditch, 2009)

The carbon removed from the atmosphere through photosynthesis by coastal ecosystems is either stored as biomass or as soil organic carbon[11]. Figure 5 and table 2 show the global average carbon pools and the average sequestration rates of the focal coastal habitats, respectively. As shown in figure 5 the key ecosystems store most of their carbon within the soil and just a little percentage within their actual living biomass (Murray et al., 2010).

> "*The carbon sequestration rate quantifies how much carbon is added to the biomass and soil carbon pools annually. Because these intact ecosystems typically have mature vegetation that maintains a steady stock, virtually all of the sequestration ends up buried in the soil carbon stock*" (Murray et al., 2010: p.5).

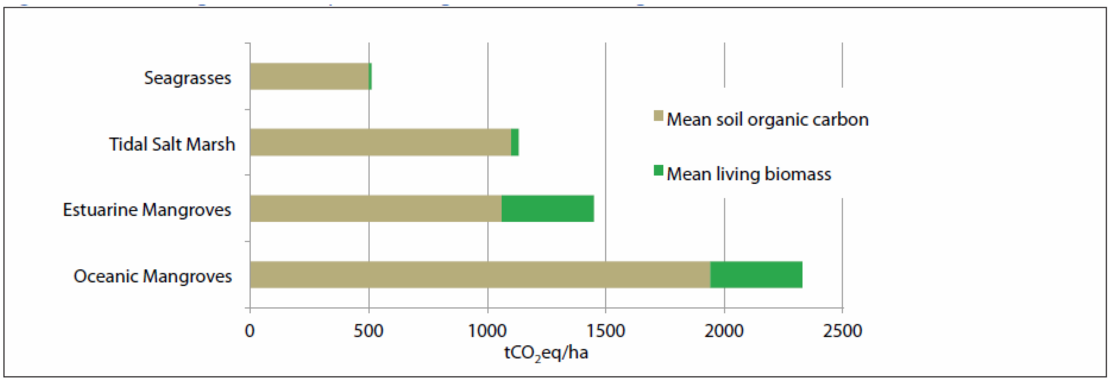

Figure 5: *Global average for carbon pools of focal coastal habitats*

Source: Murray et al., 2010, p.3

Table 2: *Global averages and standard deviations of the carbon sequestration rates and global ranges for the carbon pools, by habitat*

Habitat Type	Annual Carbon Sequestration Rate ($tCO_2eq/ha/yr$)	Living biomass (tCO_2eq/ha)	Soil organic carbon (tCO_2eq/ha)
Seagrass	4.4 ± 0.95[a]	0.4–18.3[b]	66–1,467[c]
Tidal Marsh	7.97 ± 8.52[d]	12–60[e]	330–4,436[f]
Estuarine Mangroves	6.32 ± 4.8[g]	237–563[h]	1,060[h]
Oceanic Mangroves	6.32 ± 4.8[g]	237–563[h]	1,690–2,020[h]

Sources: (a) Duarte et al., in press; (b) Duarte and Chiscano 1999; N. Marba and J.W. Fourqurean, pers. comm.; (c) Duarte and Chiscano 1999; N. Marba and J.W. Fourqurean, pers. comm.; (d) Morgan and Short 2002; PWA and SAIC 2009; Yu and Chmura 2009; Brevik and Homburg 2004; Bridgham et al. 2006; Chmura et al. 2003; Choi and Wang 2001; Choi and Wang 2004; Connor et al. 2001; Craft and Richardson 1998; Duarte et al. 2005; Giani et al. 1996; Hussein et al. 2004; Johnson et al. 2007; Mudd et al. 2009; Nellemann et al. 2009; PWA and SAIC 2009; (e) Morgan and Short 2002; Bridgham et al. 2006; Yu and Chmura 2009; (f) Bridgham et al. 2006; PWA and SAIC 2009; Chmura et al. 2003; (g) Bouillion et al. 2009; Bridgham et al. 2006; Chmura et al. 2003; Duarte et al. 2005; Fujimoto et al. 1999; Jennerjahn and Ittekkot 2002; Nellemann et al. 2009; PWA and SAIC 2009; Twilley et al. 1992; (h) D.C. Donato and J.B. Kauffman, pers. comm. See Appendix for complete references.

Source: Murray et al., 2010, p.3

As already briefly mentioned above the world wide degradation and conversion of coastal and marine habitats can be related to market failures[12]. Through certain market forces landowners and managers are driven to explore and convert their ES and habitats

11 A little percentage of the captured carbon is returned into the atmosphere via respiration and oxidation.

12 Beneath other reasons

into or for marketable goods and services. Other causes for the unsustainable management and ongoing conversion of precious natural ecosystems are linked to the failure or unwillingness of governments to enforce environmental protection, and other regulations. The lack of incentives, as well as the absence of working and enforceable mechanisms to compensate landowners, managers, or governments for their opportunity costs[13], in case they choose to protect their habitats and with it the stored carbon, compromises the incentives to protect natural and functioning ecosystems. Murray and colleagues (2011) point out that the current development of increasing habitat destruction is likely to continue due to a growing population and the actual per capita consumption, if the introduction of payment schemes for the compensation of landowners and managers to protect and conserve natural carbon sinks is being delayed or fails all together.

Economic mechanisms, such as emission trading and with it the establishment of 'carbon' markets, which aim at the reduction of GHG emissions, could mitigate the issue stated above. However, until today the success of these mechanisms with regards to habitat conservation and terrestrial carbon reductions is rather limited and with regards to blue carbon even non-existent (Murray et al., 2011). Additionally, carbon emitted or stored through management practices within coastal and marine ecosystems are not included in any international climate change mechanisms e.g. UNFCCC, Kyoto, CDM, etc. This leads to a constant under estimation of the world wide annual anthropogenic carbon emissions. At the same time, captured and stored carbon through sustainable habitat management is economically not rewarded and furthermore will not *count towards meeting national and international climate change commitments* (Laffoley & Grimsditch, 2009)

6.2.1 REDD+, PES and Mangroves

A rather new mechanism, which aims at the reduction of carbon emissions from deforestation through means of economical incentives, is called Reducing Emissions from Deforestation and Forest Degradation (REDD). The introduction of REDD is based on the importance that forests have for climate change mitigation. However, the protection of forests and their potential to contribute in a sustainable way towards carbon storage are closely related to its economical value in order to compete with other (less sustainable or even unsustainable) development strategies.

Its development is still in progress and hence, not yet standardized and still lacks a clear and precise definition (Phelps et al., 2010). The basic idea of a REDD mechanism is very similar to PES interventions and was initially discussed and promoted at the "*Conference of the Parties (COP) of the UN Framework Convention on Climate Change (UNFCCC) in Montreal in 2005 by the Coalition for Rainforest Nations*" (Ghazoul et al., 2010, p.396). The mechanism is based on the idea that the industrialised counties as well as donors, corporations, NGOs, and/or individuals compensate the developing countries for lost income opportunities which would occur from

13 Opportunity costs of alternative uses e.g. aquaculture.

deforestation and forest degradation in a “business as usual” scenario. Hence, the reduced or saved carbon emissions in the developing countries can then be used or purchased by industrialised countries in order to meet their set targets in emission reductions (Ghazoul et al., 2010). REDD+ is an addition to the normal REDD which aims at increased carbon sequestration and storage through additional activities such as forest conservation, forest restoration and their sustainable management (Kanowski et al., 2011). In other words, the industrialised countries pay for forest conservation and protection in the developing countries. The aspect of conditionality is an essential factor and as Phelps and colleagues (2010: p.312) state: “*Payments will require demonstrated emissions reductions through improved forest protection, sustainable forest management, and/or enhancement of carbon stocks*”.

In December 2009 REDD+ was a key point at the Copenhagen Climate Change Conference. The discussions and resulted in a ‘Copenhagen Accord’ which is supposed to be one of the main pillars for future climate change mitigation (Corbera & Schroeder, 2011; Ghazoul et al., 2010). “*The accord did not, however, specify binding emission reduction targets, and details on implementation and governance of a REDD mechanism remain scant*” (Ghazoul et al., 2010, p.396f). Furthermore, various factors and implementation aspects for example whether the provided incentives are publicly funded through multilateral or bilateral agreements, the involvement of the private sector, and a potential link of REDD credits to other carbon markets are still subject to debate and leave the programme with a high degree of uncertainty (Corbera & Schroeder, 2011). Compared to other environmental protection approaches, REDD+ have some advantages that can be associated with the linkages of economic incentives, land-use and environmental conservation. However, several limitations and aspects of concern have to be mentioned and taken into consideration as well. Issues are ethical dilemma, additionality, system leakages, permanence, national sovereignty and native land rights (customary use-rights), equity, and crashing carbon market (Ghazoul et al., 2010). Most of these issues are similar if not identical to the common constraint and limitations of terrestrial PES. If, however, these constraints are taken into account, a proper institutional context and other success conditions are available and in place, REDD+ should hold the same potential as PES schemes do. (Costenbader, 2009; Ghazoul et al., 2010).

A compliance market and voluntary carbon markets can be distinguished[14]. “*Compliance markets are created and regulated by mandatory national, regional or international carbon reduction regimes*” (Kollmuss et al., 2008: p.4). Voluntary markets, on the other hand, are based on voluntary agreements between individuals/private actors or governments and the project hosts. Despite the numerous cases of voluntary carbon markets no generalized standards or definitions have been adapted so far (Kollmuss et al., 2008). Several institutions help with the development, implementation and financing of REDD+ programmes. Each of these institutions receives funding from industrialised countries which will be invested into the research, capacity and readiness of the host countries as well as for the payment schemes themselves. The Forest Carbon Partnership Facility (FCPF), Forest Investment Program (FIP) and the United Nations Collaborative Programme on Reduced Emissions from

14 Over the last few years more then a dozen different voluntary carbon markets emerged.

Deforestation and Forest Degradation (UN-REDD) are the most present and famous institutions.

As already mentioned above, several marine and coastal ecosystems have an immense potential for carbon sequestration and storage. Mangroves are one type of these ecosystems and their protection could conceivably be included in the establishment of REDD+ mechanisms or other protocols. Nevertheless, REDD+ does not include carbon in other coastal or marine ecosystems and neither is the potential of carbon storage within soils included. As stated in Murray et al. (2011, p.ES-3): "*an omission that could result in failure to protect some of the largest and most vulnerable carbon stocks on Earth*".

6.2.2 Blue Carbon Credits as an MPES for Mangrove Protection

Carbon sequestration and storage has already been used in combination with economical incentives and within PES interventions on a terrestrial level. Mangroves are a coastal ecosystem and their importance for forestry value, fisheries value, storm protection, water quality or pollutant uptake, and sediment retention is considerably high and has been described above. However, the unsustainable use and degradation are continuous and until today, apart from some pilot projects, there are no payment schemes in place, which value the various essential environmental goods and services they generate (Gordon et al., 2011; Murray et al., 2011). Additionally, in relation to blue carbon no mechanisms or pilot projects have been established in the voluntary or compliance market (Gordon et al., 2011). The question if blue carbon will be subject to a compliance market depends on the outcomes of the current REDD+ development.

The above developed and described preliminary assessment framework has been used to analyse already existing MPES examples and showed their characteristics, advantages, limitations and indicated several important key factors. This section deals with the missed potential of mangroves as blue carbon sinks and other important ES. The framework will be used to identify the presence or absence of essential key factors and preconditions of such MPES schemes. Here the purpose is to determine and investigate the limiting factors and bottlenecks, which are hindering the implementation of MPES interventions by applying the developed preliminary assessment framework. With an increased understanding of the missing features and key factors recommendations can be made for a successful development and implementation of MPES, if feasible.

Concerning the general definition of PES/MPES and with regards to Sommerville and colleagues (2009) conceptual framework the **conditionality** of the MPES intervention is essential. With regards to mangroves and carbon storage it means that if and only if mangroves are protected from otherwise destructive land-uses a (financial) compensation will take place. Not only the protection and conservation of already existing mangroves could be considered but also the re-afforestation/renaturalisation of mangroves in coastal areas. Though, the REDD+ mechanism explicitly deals with the reduction of deforestation and hence, re- and afforestation will not be considered in this analysis.

The primary services which will be subject to the MPES scheme is the amount of carbon which is captured and stored within the biomass[15], or to be more precise tons of CO_2. However, as already pointed out mangroves provide several other benefits and essential services which can be coupled or included in an MPES intervention. First of all, the focus will be on carbon capture and storage since global markets for carbon credits[16] already exist and are the substance of a REDD+ mechanism.

> "*Carbon payments monetize reductions in the greenhouse gas emissions and provide compensation for a unit of carbon, where one carbon credit equals one metric ton of carbon dioxide equivalent (mt CO_2e). The price per ton of carbon represents the price investors were willing to pay in order to sequester and store one ton of carbon. Several carbon markets exist worldwide*" (Yee, 2010, p.14).

Permanence or also called temporal leakage is another important factor which has to be taken into account when dealing with carbon storage as part of the conditionality. Carbon credits are relative to the reductions in deforestation, payments are delivered for the saved and stored carbon. Hence, the saved carbon within undisturbed mangroves has to be secured as long as the duration of the compensation. In case the project comes to an end and the land-use/conservation changes the accumulated credits and the contribution to climate change mitigation are lost (Yee, 2010).

Monitoring as well as enforcement of compliance (of the ES provider as well as of the beneficiary) is another essential criterion, which has to be clarified within the conditionality of an MPES intervention. However, monitoring and enforcement will be part of the institutional context under the criteria of facilitating structures.

The **Positive incentives** are straightforward in this MPES scheme and come in form of compensation, usually monetary payments that are based on the opportunity costs of alternative land use. Nevertheless, the scale at which such positive incentives are used is important. Hence, it has to be differentiated between the compensation of individuals or governmental agencies, which would be responsible for the change in land-use. Especially with regards to emission credits and UN-REDD which take place under the United Nations Framework Convention on Climate Change the management takes place on a national level, depending on the setup individuals might not feel the positive incentives directly and the criteria of voluntary participation as a prerequisite of PES stated by Wunder (2005) might be blurred.

Additionality is not only a consideration for MPES schemes but also a prerequisite for any REDD+ mechanism. Any project aiming at receiving compensation for carbon offsets has to provide enough evidence about the additional carbon savings in comparison to a business as usual scenario or with regards to a historical baseline analysis. "*Historical land use or projected land use with the highest opportunity cost are the most logical baselines*" (Yee, 2010, p.17). The additional carbon stored in such a

15 The stored carbon within soils and sediments is not yet recognized within international protocols even though it bears a high potential.

16 Carbon credits and carbon offsets are synonymous terms and are used interchangeably.

project is the difference between the baseline scenario and the carbon stored in a protected and sound mangrove ecosystem.

Eventual spatial leakage affects the overall additionality and positive outcomes of a project. If land-use restrictions lead to a migration of destructive mangrove management to another location, one speaks of spatial leakage. The scale of the project is of importance. If implemented on a national level, the boundaries of the project are equal to the ones of the country. Hence, spatial leakage is less likely to occur, however, transaction, monitoring and enforcement costs are increasing tremendously at the same time (Yee, 2010; Murray et al., 2011). Regardless of the size, scope and location of the MPES intervention, effective monitoring and enforcement are essential, which brings us to the next key factor – the institutional context.

6.2.3 Institutional Context

The governmental support of the implementation of a carbon offset programme is potentially secured due to the international scale of the project and the involvement of the government in the initialisation of the programme. This is especially true for MPES projects on international scale based on a compliance market where the development of agreements takes place on an international base. If, however, blue carbon sinks are based on private deals and on a voluntary market the government might not be included in the development of the programme which might lead to other unforeseen complications with other governmental regulations and objectives.

For the purpose of government-financed projects it is plausible that the government is interested in establishing a supporting legal and policy environment for the programme. The interest and willingness to participate in international PES schemes shows the support of the local governments in such schemes.

Any international REDD+ intervention can be described as a multiple-level PES scheme. Such a multiple-level PES incorporates many different and diverse stakeholders with different agendas and even spread around the globe. A complete analysis of these schemes would go beyond the scope of this research project (Costenbader, 2009). Nevertheless, several key conditions and obstacles, which affect the success of an MPES scheme, can be worked out. At this point, however, it is already save to say that any PES scheme at such a scale involves high transaction costs due to the complexity and variety of stakeholders and institutional interplay.

Several informal as well as formal institutions have to be considered while some are even essential for the success of an MPES intervention. The interplay of a blue carbon related MPES intervention with other natural resource management programmes can not be analysed due to missing examples and information. However, it is important to keep in mind that depending on the scale and location of the MPES scheme other environmental management tools have to be considered and taken into account. In some cases the government might even be able to reduce the costs of top-down implemented environmental protection through the introduction of sound MPES schemes.

With regards to UN-REDD programmes it is important to start off with the already existing intermediary, the United Nations Framework Convention on Climate Change, which started the development and several test projects[17]. It acts as an independent body which assesses the potential and capacity of each project (Gordon et al., 2011). Currently mechanisms for preparation and readiness are being carried out by the UNFCCC to acquire data and knowledge about the state and capacity of potential REDD+ host countries. This institutional body is designed to reduce the risk of the actual implementation of the project.

If mangroves as blue carbon sinks were included in the UN-REDD programme, the necessary institutions and interplay would be under the roof of the UNFCCC and the interplay and success would depends on the actual implementation and performance of the REDD+ framework (Gordon et al., 2011).

However, with regards to coastal ecosystems various property rights and ownership structures exist and can easily could transform into obstacles for the implementation of blue carbon MPES interventions. Especially in the developing countries several other entities apart from the land owner often have use rights such as managing or harvesting rights over a mangrove forest. In addition, customary rights often prevail and are not always considered. Hence, it is often unclear who has the actual power to influence and steer management practices in order to secure eventual carbon storage (Costenbader, 2009). Carbon offset programmes and REDD+ projects come with considerable investment risks which, under unsecured and unclear tenure, property, or land-use rights face high uncertainty for the success and accountability of the entire project. It is therefore, that a potential project host country is obliged to address land tenure issues as well as guaranteeing the active and long-term participation of local communities in the development of their national action plans. These preconditions are stated within 'The Advance Negotiating Text'[18] which additionally includes a framework that focuses on the existence and protection of these rights in potential REDD+ countries (Lyster, 2011). Even though, the stated implementation requirements for tenure rights and local community participation within '*The Advanced Negotiating Text*' are innovative, the actual "*implementation of these rights requires clarity with respect to the type of tenure which grants property rights in forest carbon*" (Lyster, 2011: p.126).

Carbon offsets for a compliance market are based on international regimes and trade agreements. For voluntary carbon offsets case specific agreement and regulation are established. In both cases the involved institutions range from international to local scale. Their accountability, legitimacy, performance and interplay are important to secure the success of the programme. For this reason the institutional outline has to be set up in a way that it meets the demand of the service at hand with regards to the market-based mechanisms. Under the roof of a UN-REDD intervention the facilitating structures such as contracting, administration, monitoring and enforcement would be collectively carried out by the project's host government, the responsible and administrative body within the UN-REDD. The activities of the UN-REDD are supported by three other United Nations organizations: Food and Agriculture

17 Test projects do not include blue carbon programmes.

18 Available at http://unfccc.int/resource/docs/2010/awglca10/eng/06.pdf

Organization (FAO), the United Nations Development Programme (UNDP) and the United Nations Environment Programme (UNEP) (UN-REDD, 2010). Especially with regards to blue carbon sink and mangrove forest the involvement of all stakeholders in the development of any MPES or REDD+ intervention is essential for its success. Since blue carbon sinks are not subject to any REDD+ scheme it is important to include local communities and all stakeholders in the development of a potential blue carbon payment scheme.

6.2.4 Biophysical Context

Clear defined system boundaries are difficult to establish for mangrove ecosystems as blue carbon sinks, especially because mangroves form the "*basis for complex and extensive ecosystems at the interface of terrestrial, freshwater and marine ecosystems*" (Gilbert & Janssen, 1998, p.323f).

The ecosystem services on hand seem to be pretty straightforward – tons of carbon stored in mangroves, which without the compensation scheme would have been transformed into other land-uses and hence, emitted into the atmosphere. Also the pressing issue of global warming and the emission reduction goals of the industrialised countries seem to work as incentives enough to set up feasible payment schemes for blue carbon.

However, several biophysical issues are related to blue carbon sequestration and payment schemes. In general the calculation of saved carbon is still subject to complexity and some uncertainty (Kollmuss et al., 2008). Especially mangroves, which save most of their carbon within their soil are subject to these issues. The problem is the lack of clear methodologies, which measure the stored carbon within soils (Gordon et al., 2011). Hence, future REDD+ schemes need to include soil carbon in their mechanisms in order to provide the very essence of blue carbon MPES schemes.

6.2.5 Social Context

Under the UN-REDD stakeholder involvement and continuous participation is planned to be guaranteed through the principles of transparency, accountability, representation, access to information, and participation and inclusion (UN-REDD, 2010). Additionally, "*This has so far included several workshops on governance issues related to REDD+, outreach consultations and guidance and requirements on consultative processes in the development of national strategies. The UN-REDD Programme aims to facilitate the development of guidelines for seeking Free, Prior and Informed Consent from indigenous peoples*" (Westholm, 2010, p.7). Furthermore, the establishment of REDD+ projects should generate multiple benefits reaching from ecological to social benefits.

At this stage of the research no clear statement can be made about the organisational capacity of communities as well as on the issue of trust and legitimacy among stakeholders due to the lack of information and non-existence of actual case studies which could provide background knowledge on this matter.

It is very likely that unequal power relations emerge from the establishment of blue carbon sinks. Multiple stakeholders are involved. Some of them might be multinational co-operations or governmental agencies with great political and social influence in comparison to small stakeholders or resource manager on a community level. If these unequal power relations affect the negotiations and development of a blue carbon scheme the outcome is likely to be unfair and unsustainable on a long-term.

Mangroves are the source of many ES and often are part of the livelihood of local communities. An increase of payments for the protection of mangrove forests and an eventual restriction of use could influence traditional users and other customary practices within these mangrove forests. The capacity of local communities to participate and profit from the implementation of a MPES scheme depends on the specific region and project characteristics but have to be taken into account. Additionally, the benefits and received compensation derived from the protection of such areas could lead to conflicts within communities. The current assessment of preparation and readiness of project host countries is one institution that is designed to take these factors into consideration. However, as long as social implications for local communities and indigenous people are not entirely included and accounted for resistance and ineffective implementation of blue carbon payment schemes are likely to occur (Costenbader, 2009; Larson, 2011).

6.2.6 Economic Context

The specific carbon market[19] regulates the price per ton of saved CO_2 and therefore determines the total compensation of a project's achievement. With regards to a UN-REDD project the different UN organizations act as intermediaries and facilitate financial matters and transactions between the industrialised countries and the project host country. Continuous payments from the industrialised countries are essential in order to sustain the financing mechanism. As for all PES and MPES schemes payments or compensations have to exceed the forgone opportunity costs in order to secure the persistence of the project.

However, due to a high degree of complexity and the multiple-level characteristics any REDD+ project entails high transaction costs, which have to be internalised in order to guarantee the longevity of such an MPES scheme.

6.2.7 Results of the Assessment of Mangrove Related MPES

The current development of the REDD+ scheme does not include blue carbon and hence, no REDD+ demonstration projects are available or planned. It is therefore, that potential carbon projects miss out on benefits from data collection and experiences from

19 Either a voluntary carbon market or the compliance market.

REDD+ testing opportunities (Gordon et al., 2011). As long as the storage of carbon within the soil is not included within official REDD+ protocols the potential of integrating blue carbon offsets is close to zero. Hence, the establishment of related MPES schemes depends on the accounting for soil carbon and practical methods to measure the saved amount of CO_2. The institutional context of potential blue carbon MPES schemes is complex especially due to the multi-layer PES outline. A complete institutional analysis has not been carried out and would go beyond the scope of this research project. However, several key factors within the institutional context were worked out by applying the developed framework. One of the major determining conditions for the success is related to the handling of property and tenure rights as well as the role and integration of local communities and indigenous people. The groundwork has been laid with the adaptation of 'The Advance Negotiating Text'. Nevertheless, certain aspects of tenure and property rights are still subject to uncertainty and hold a great potential of risk. In this sense, REDD+ projects could either benefit communities due to the establishment of tenure rights and encouraging communities' integrity and their potential to manage mangrove forests. On the other hand, REDD+ projects could undermine customary and tenure rights and exclude local communities from the active participation of their local ecosystems and hence limit local livelihoods. These social and institutional implications can backfire and affect the implementation and success of the payment scheme.

Since blue carbon sinks do not exist yet no real factors contributing to its success could be determined. Hence, the following factors could be determined as the main shortcomings and bottlenecks which have to be resolved and overcome in order to increase the likelihood for success of the implementation of blue carbon sinks as MPES schemes:

1. Carbon stored within the soils of mangroves is not included in any REDD+ protocol and hence lacks clear measurability.
2. On an international scale complex institutional interactions prevail and multiple-level PES schemes are subject to high transaction costs.
3. Still unclear conditions for tenure and property rights issues.
4. Protection of mangroves could interfere with local communities livelihoods.
5. Problems of unequal power relations between countries, co-operations, and local communities.

6.3 Marine Protected Areas and Tourism

Chapter 2.2 focused on a general approach for marine and coastal protection and conservation, while pointing out the importance MPAs have with regards to that matter. One type of MPES is the combination of marine protected areas (MPAs), which focus on the conservation of marine species, habitat and biodiversity, and tourism. Here tourists pay through user and entry fees for the conservation of biodiversity, recreational

value and aesthetics (Forest Trends and The Katoomba Group, 2010).
Hence, it is important to see and investigate the potential of MPAs for marine and coastal conservation in a wider economic and social context. Especially MPAs combined with tourism are an interesting concept. Stewart (1993) argues in favour of marine conservation regimes in order to control and manage sustainable tourism. Since tourism has negative as well as positive effects, proper marine conservation measures including MPAs can help to find a balance between conservation and development (Stewart, 1993). Also Reid-Grant & Bhat (2009) showed in a study that with the introduction of user fees and taxes it is possible to cover the management costs of a MPA which are usually prone to a lack of financial resources.

Throughout the literature and advocates of MPAs several positive effects are linked to the establishment of such protected areas, "spillover" is one of the effects often mentioned. It is assumed that MPAs have a positive effect on fish density and size and the movement of species across reserve boundaries could increase local catches. However, as explained in Rowley (1992) clear scientific data and proof is still needed to confirm this hypothesis. Nevertheless, spillover effects are very likely to happen based on the current knowledge on fish behaviour and movement (Roberts & Polunin, 1991; Rowley, 1992). "Larval export" is another potential benefit of an MPA, which suggests that the regional fishery benefits from the export of larval from reserves. Even though the export of larvae from protected areas to regional surroundings has a great potential it also still lacks scientific proof. Additionally, MPAs may contribute to the preservation and protection of fishery stocks genetic resources. Location and size of marine reserves depends on their primary objectives (Roberts & Polunin, 1991).

MPAs are an essential tool with regards to marine and coastal protection and conservation since they have the potential to protect critical areas, intensely fished and exploited species, and furthermore, MPAs can be used as buffers to eventual fishery mismanagement, and other unforeseen or unusual events (Gary et al., 1998). However, Gary and colleagues (1998) point out, that marine reserves alone are insufficient for sustainable marine and coastal protection due to the nature of marine environments such as migratory fishery, continuous flow of water masses, population replenishment and other risks e.g. chemical pollution. Hence, they opt for additional adequate protection of ecosystems and species outside of MPAs.

Even though certain types of MPAs can be regarded as MPES schemes several characteristics of MPAs have to be considered first before applying the preliminary framework. The primary objective of MPAs is usually biodiversity conservation, however, also other objectives have been on the agenda of MPAs such as the sustained provision of ecosystem services, cultural and spiritual values, and providing opportunities and space for research and education (National Research Council, 2001; Leslie, 2005). Alban and colleagues (2008) distinguishe between three different types of objectives of MPAs:

1. protection of biodiversity,
2. sustainable fisheries management, and
3. development of non-extractive uses of marine and coastal ecosystems in

terms of recreational uses e.g. ecotourism.

However, often a combination of the three objectives within an MPA is the cases, nevertheless, these variations of purposes do not always occur in harmony and in some cases it leads to conflicting interests of different stakeholders. However, Carter (2002) states that some scholars define MPAs as tools for integrated coastal and marine management and hence should manage the various uses of and interests in marine and coastal ecosystems and the provided ecosystem services. The third possible purpose of MPAs, the development of non-extractive uses of marine and coastal ecosystems for e.g. ecotourism is based on the observation that the conservation and protection of specific components of the ecosystem e.g. corals, seagrass, marine mammals increases the willingness to pay of visitors. Hence, the conservation of biodiversity and the protection of ecosystems result in increased financial revenue from an MPA. The increased income can then be used to cover (or partly cover) the management costs of the MPA. Consequently, if the quality of the environment is secured ecotourism can be regarded as an economic incentive to conserve and protect biodiversity and ecosystems and therefore can be a possibility to advance and promote sustainable use of ecosystem services and natural resources (Alban et al. 2008). Another option for the use of the increased consumers' surplus is the compensation for the restriction of fishing and other extractive activities. If local communities benefit from the creation of additional livelihoods and income from the establishment of ecotourism it can lead to a higher social acceptance of the MPA.

However, tourism can also have negative effects on ecosystems and the MPA. Unrestricted and uncontrolled tourism can have negative effect on the environment and can lead to overcrowding of the protected area. Even some of the tourist's activities like SCUBA diving and marine mammal watching can affect the environment in a negative way and hence, reduce the attractiveness of the MPA all together. In Alban et al. (2008) it is suggested that these negative factors can be limited and avoided by

1. adjusting the visitor numbers to the carrying capacity of the ecosystem,
2. control the numbers of visitors, and
3. control the mobility of the tourists within the MPA.

The introduction of user fees can be regarded as one management tool to regulate the access and number of visitors. Additionally it is one way to capture the consumers' (visitors') surplus. Hence, make the MPA self-sustaining by covering management and protection costs (Alban et al., 2008).

The benefits and the global amount of welfare provided to society created by an MPA is subject to varying distribution, especially with regards to potential changes in use-rights of involved stakeholders. Even if an MPA has a net positive effect on the global welfare its distribution might not always favour all participants and some stakeholders might even experience negative impacts from the creation of an MPA. Especially in developing countries with a scarcity of job opportunities, the opportunity costs of ecosystem conservation is an important issue (Carter, 2002; Alban et al., 2008). If local communities are not involved in the decision making and the creation of the MPA adverse effects and frustration can be observed which influence the effectiveness and even success of the project. Generally the distributional impacts are related to location and size of the MPA compared to the fishery, state and level of development of the country, as well as to the condition and availability of job opportunities. The perception of the MPA by the affected stakeholders is of high importance and determines the acceptability of the project. Especially fishermen are often opposed to the establishment of MPAs since they are usually affected directly and often regard them as the end of their fishing activity (Beaumont, 1997; Brown et al., 2001; Alban et al. 2008). However, some cases sowed that fishermen can represent an important stakeholder group which support an MPA. Alban and colleagues (2008) point out that the acceptability additionally depends on the knowledge and awareness of the actual state of ecosystem degradation.

In any case it is essential to establish and use intact institutions in order to deal with and set up compensatory measures (Alban et al., 2008). This could "*help to increase the social acceptability of the project and additionally help to preserve the belief in the fairness of institutions, which may improve the likelihood of compliance with the rules, and limit the transaction costs related to the change*" (Alban et al., 2008, p.16f). With regards to MPAs several compensatory measures exist:

1. money transfers,
2. building of harbour facilities,
3. assistance with the development of alternative fishing activities,
4. assistance with converting to or diversifying into tourism-related activities, buy-back programs,
5. allocation of exclusive use-rights, such as catch quotas, and
6. territorial use rights of fishing (Alban et al., 2008).

In Gary et al. (1998) two variables were identified to determine the effectiveness of a properly designed and managed MPA: First of all it is related to the apparent threat, its intensity or spreading characteristics and how controllable it is. Secondly, the spatial scale of the threat is important. Or in other words, an MPA is potentially effective in protecting species if the spatial dimension of the apparent threat is comparably small and its intensity manageable within the scale of the marine protected area. As Gary and colleagues (1998, p.85) put it: "the only successful control is where the scale of management is as large as the scale of the threat".

Nevertheless, not only the biological and oceanographic factors are important for the success and effectiveness of MPAs but also social, cultural, economic, and institutional aspects have to be considered carefully (Charles & Wilson, 2009; Lundquist & Granek, 2005; Stewart & Possingham, 2005). Furthermore, it is pointed out that the institutional design and interplay are key/factors for the success and effectiveness of MPAs. Additionally, it is essential that all stakeholders are included and have the feeling of meaningful participation in the decision-making process, during the creation as well as in the management of the MPA (Stewart, 1993; Beaumont, 1997; Brown et al., 2001; Alban et al. 2008). In order to assess these social, cultural, economic, and institutional factors of MPAs and to make a statement about the success of MPAs the developed preliminary assessment framework will be applied to it.

6.3.1 The Case of Bonaire National Marine Park

One of the case studies, which will be used throughout this study, is the Bonaire National Marine Park (BNMP). The island of Bonaire is located 100km north of Venezuela in the southern Caribbean with a population of 15,000. Bonaire was part of the Netherlands Antilles until October 2010. Now together with St. Eustatius and Saba, Bonaire (called 'the BES Islands') became a special municipality of the Netherlands. The governing council holds the executive power over the BES Islands but nevertheless, they are still subject to Dutch law. 84 percent of Bonaire's Gross Domestic Product (GDP) comes from tourism and is the primary economic income (Cooper, 2011).

The BNMP was established in 1979 with help of the Dutch Government and the World Wildlife Fund Netherlands and includes the island of Bonaire, the water surrounding Bonaire, including the satellite island, as well as the waters around 'Klein Bonaire', from the high-tide mark to 60 meters of depth and encompasses a total area of 27 km^2. The entire marine park is within the territorial as well as the jurisdiction of the Island of Bonaire. Several ecosystems can be found within the BNMP such as ringing reefs, seagrass beds, beach areas, mangroves, lagoon areas, and karstic systems while facing various main challenges like overfishing, nutrient enrichment, development/conversion of land use, poaching, heavy recreational use (snorkelling/diving), sedimentation, terrestrial run off, and illegal sand mining (STINAPA, 2012). Apart from natural disturbances (hurricane events and coral bleaching) and tourism threats within this area are pollution, overfishing, and coastal development.

In the 1980s the Bonaire Island Government ceded the right to manage the MPA due to a lack of capacity and financial means and entrusted STINAPA (Stichting Nationale Pareken) Bonaire with the management of the BNMP. A public consultation in the early 1990 with a focus on the direction of Bonaire's future directions and development resulted in a clear statement by the residents for a sustainable and environmentally conscious development. This broad coincide shaped and steered most development policies ever since (Cooper, 2011).

The marine park started off with serious financial issues and hence was lacking staff and enforcement of rules, which lead to the situation in which the BNMP was only a 'paper-park' (Dixon et al. 2000; Cooper, 2011). These developments and the concern about the lack of formal and adequate management of the MPA on the one hand and an increasing number of visitors, especially divers, lead to an evaluation of the prevailing situation in 1990. Consequently a user and dive fee was introduced. It was high enough to cover for most of the management expenses including salaries, operating costs and capital depreciation (Dixon et al. 2000). Today the MPA is able to cover about 80% of their expenses with the user fees[20] (de Meyer & Simal, 2004; Cooper, 2011). STINAPA is

20 According to Cooper (2011, p.17) "*The user fee (or the "Nature Fee" as it is known) costs USD 25 for scuba divers for a year's pass, or USD 10 for a day pass for any activity other than scuba diving (e.g. swimming, snorkelling, windsurfing, boating, kayaking), which is also valid for a year*"

entitled to keep the total amount of user fees but is obliged to use and invest it in and for the management of the MPA. In 2005 an additional trust fund was established to ensure the sustainable funding of the programme. This trust fund also includes other MPAs apart from the BNMP.

Additionally, the BNMP is protected by the Marine Environment Ordinance (A.B. 1991 Nr. 8) and by the end of 1999 the Central Government of the Netherlands Antilles declared it officially a National Park[21]. The BNMPs mission is to: "*protect and manage the island's natural, cultural and historical resources, while allowing ecologically sustainable use, for the benefit of future generations*" (STINAPA, 2012). As already mentioned above and as one of the main criteria for the selection of the case studies, the BNMP is a multi-use marine protected area and the following main stakeholders are involved in the planning and management of the marine park:

- Government
- Tour operators
- Accommodations
- Schools
- Building and zoning department
- Environment and Nature Management department
- Legal department
- Harbour office
- Agricultural department
- Dive operators and other water sport activity providers
- Other NGOs
- Volunteer groups

Another criterion for the selection of the BNMP as a case study is its openness for tourism, which accounts to approximately 38,000 visitors per year. Furthermore, the Bonaire national Marine Park declares the following goals within their management objectives:

- *"Maintain and restore the ecosystems, biological diversity, and ecological processes*
- *Protect and restore the cultural and historical resources of significance*
- *Manage the Marine Park as a regionally and globally significant example of a successful multi-use marine protected area through: education and outreach, research, monitoring, law enforcement, maintenance, and administration*
- *Allow use of the Marine Park by promoting non-destructive activities and working with stakeholders to establish guidelines and regulations to minimize impact on the environment"* (STINAPA, 2012).

21 Dudley (2008) defines National Parks as "[…] *large natural or near natural areas set aside to protect large-scale ecological processes, along with the complement of species and ecosystems characteristic of the area, which also provide a foundation for environmentally and culturally compatible spiritual, scientific, educational, recreational and visitor opportunities*" (Dudley, 2008; p.16).

The management of the marine park is administered by a non-governmental organization called STINAPA. Geoghegan (2001) describes the level of management as moderate to high. Additionally, it is stated that the adjacent communities have a high dependency on the MPA and the attracted tourism due to its small size, low availability and presence of other resources, a dry climate, an diversified economy and the relatively remote location (Dixon et al. 2000; Geoghegan, 2001).

Responsible for the management of the Bonaire National Marine Park is the BNMP Management Committee, which is created from representatives from the Government, the Council of Underwater Resort Operators, and the Bonaire Hotel Tourism Association, and STINAPA, which has a management agreement with the Island Government. According to Dixon and his colleagues (1993) this management Committee is working and functioning well.

The successful management and conservation of the marine ecosystem around Bonaire are due to the establishment and proper management of the BNMP. However, also the support and engagement of the local dive operators are an essential element of the success (Dixon et al., 2000; Cooper, 2011). Since dive operators depend on the health of coral reefs, biodiversity and an abundance of different species they have a high incentive in monitoring and enforcements of protection regulations. Nevertheless, the combination of various uses of the MPA also brought some trade-offs. Diver use of the protected area caused an increased stress for some marine ecosystems if exceeding a certain threshold but on the other hand is the source for a substantial increase in revenue and consumer surplus. An economical and ecological combined study by Dixon and colleagues (2000) showed that with a proper management and diving education it is possible to have a mutual enhancement of ecosystem conservation and dive tourism which can even be higher than the current level of 200,000 divers per year.

6.3.2 BMNP as an MPES Scheme

According to Forest Trends and The Katoomba Group (2010), MPAs are one specific type of market for MPES interventions. Multi-use MPAs, which allow tourism, education and research contribute to marine and coastal biodiversity and resources protection. With the charge of visitors or user fees the provided services are part of a payment scheme in which beneficiaries compensate the supplier for the (enhanced) provisioning of the service. In several publications and articles the Bonaire National Marine Park is described as an example for success in combining marine conservation, socio-economic development, and tourism, while achieving sustainable and self-sufficient financing of the programme (Dixon et al., 2000, deMeyer & Simal, 2004; Spergel, 2005; Cooper, 2011).

These payments and delivery of the services are the **conditionality** of the MPA payment scheme. However, the conditionality only will hold true if and only if a stop in payments results in the ending of the supply of the services. If an MPA is self-sufficient and depends on entry and user-rights fees this dependence holds true. This is also the case for the BNMP, which became financially independent over the years. The provided services are, however, less clearly defined than in other MPES schemes. Rather bundled services are provided by the BMNP including biodiversity, fish nurseries, recreational and aesthetic aspects and other ecosystem services. Some of the provided services can not be made exclusive for the beneficiaries only. Examples are migratory fish, spillover or positive effect from biodiversity conservation which are felt outside of the MPA.

The **positive incentives** for the ES providers, the BNMP as an entity, are financial flows from visitors and other users. These monetary payments are used for the future protection of the MPA and to sustain the provisioning of the services.

Measuring the **additionality** after the implementation of the BNMP is rather difficult and has several dimensions. Without payments and hence without the MPA in place the ecological state of the provided services would differ from the actual state. To which degree is nevertheless difficult to assess. In addition, the creation of the BNMP also affected the social and economic structures of the local inhabitants positively. From a sustainable point of view this is definitely an essential contribution of the MPA. Identifying the total extent and dimension of additionality through the establishment of the BNMP would go beyond of the scope of this research project. However, the following sections deal with key factors as well as institutional, political and legal, biophysical, social, and economic conditions, which enable the success of the establishment of the BNMP. Controversial as it sounds, tourism (now partly used for the protection of the MPA) was one of the main threats to the region. With the establishment of the BNMP also regulations and legal measures were introduced.

6.3.3 Institutional Context

The establishment of the MPA on Bonaire was induced by the local government. After several years the government left the management of the MPA to STINAPA which still has the full support and legitimacy of the government.

The BNMP received the National Park Status in 1999, which was a milestone for adapting legislative and regulatory measures with regards to environmental issues. Additionally the legislation is adapted to the needs of the BNMP and *vice versa*. An Example is the change in the procedure and guidelines for coastal constructions.

The most important institutions on the national level of Bonaire with regards to the MPA are the Governing Council with the executive power; the Bonaire Island Council (*Elected body responsible for nature policy, legislation conservation and preservation on Bonaire and Transcribes Nature Conservation Framework Law of the Caribbean Netherlands into Bonaire legislation*); Department of Physical Planning, Environmental Resources and Infrastructure which ensures the effective management of natural resources and carries out land-use planning; STINAPA is responsible for the management of the BNMP, environmental awareness rising and education, and is also included in decision-making processes for nature-related issues outside the MPA; and several other civil society organizations and resource users (fishers, dive operators) which help with (Participate in consultations on matters relating to conservation and resource use) and environmental education (Cooper, 2011).

STINAPA as the institution responsible for the management of the MPA can be regarded as an intermediary. Management, resource use and other practices by local communities, fishers and dive operators can be related to the ecological health, biodiversity and state of the BNMP – provision of ecosystem services. Since tourism is the greatest income on the island the local people depend on a sustainable provision of such ES. The intermediary (STINAPA) regulates and steers the connection of the service providers and the beneficiaries – the tourists. Since the BNMP is almost self sufficient this MPES scheme can be regarded as a success. The dependence of the local population on the provided service creates incentives for self-monitoring and enforcement of the established regulations.

The MPA is relatively small and the involved stakeholders are easy to determine (rather homogeneous). Use-rights are clearly defined and regulated although the benefits are not created in one specific area but results from an interdisciplinary and sustainable management of various resources. Local people involved in the tourism sector profit more or less directly from the payment scheme. Others, like fishers profit indirectly through healthy fisheries due to spillover from the MPA.

STINAPA is responsible for and provides the facilitating structures such as administration of the park, contracting (e.g. of dive operators), entrance fee but also for monitoring and enforcement of rules. However, also other entities such as fishers, dive operators, researchers and other stakeholders are involved in processes of monitoring and even enforcement/reporting of non-compliance of rules (STINAPA, 2012).

6.3.4 Biophysical Context

The boundaries of the BNMP are clearly defined and the total area is relatively small in size. Hence, they have favourable conditions for the affective management of the area.

The importance of the available ecosystems might not have been crucial or close to disappearance, however, local people are highly dependent on the MPA and the related income from tourism due to little other available resources. The implemented regulations and enforcement of rules are linked to the overall health of the various ecosystems and increased biodiversity. For the local communities it became apparent how protection and conservation are linked to improved environmental conditions and the clear link to increased eco-tourism.

However, it is not possible to clearly measure or quantify the provided services. Especially scenic beauty and biodiversity are difficult to measure. Therefore, the overall health of the marine ecosystems is an important estimate about the success of the MPA. Studies have been carried out to estimate the carrying capacity of the MPA and a limitation of visitors and user of BNMP (Dixon et al. 2000; Geoghegan, 2001)

6.3.5 Social Context

Throughout the establishment and further development of the BNMP different stakeholders and institutions were involved and created. Today the establishment of the MPA is supported by various institutions including civil society and enjoys full legitimacy and support. The continuous involvement of the different parties into capacity building, policy making processes, and other decision-making processes feeds into this trust of stakeholders and increases the effectiveness of the MPA.

Ongoing integration, rising awareness, education and capacity building ensures a sustainable approach towards the integration of all stakeholders and especially the general public. The importance of communication between the STINAPA and the general public has been recognised and is actively encouraged. Hence, a "*culture of dialogue has been institutionalised and a collaborative working relationship has developed*" (Cooper, 2011; p.26) which resulted from the approach of dividing power between stakeholders, government and the intermediary.

Furthermore, acceptance of the MPA within the local communities is essential. Since the Island of Bonaire has little other resources available and the tourism is the biggest economy, the MPA is widely accepted. The marine park has legal and social legitimacy and the management committee encourages active and meaningful participation of the main stakeholders and hence allow for effective governance.

It is also important to state that with the establishment of the MPA the local and communities' livelihood has not been compromised. In contrary, as tourism holds the greatest potential for income on a broad scale the establishment of the MPA received great support from the civil society. Also local fishers support the MPA and report positive outcomes with regards to the health of fish stocks and the marine ecosystem as a whole (Dixon et al. 2000; Geoghegan, 2001; STINAPA, 2012).

6.3.6 Economic Context

The availability of financial resources directly affects in many cases the level of management of an MPA (Geoghegan, 2001). These financial resources usually come from Government allocations (in this case The Netherlands), donor assistance (in the first phase from the World Wildlife Fund, Netherlands), and from visitor and user fees. The BNMP managed to cover 80% of their costs by user fees. Together with the trust fund this indicates a sustainable way of financing the MPA. The local people have little alternatives to tourisms due to a lack of natural resources and agricultural production. Most of the local population are not receiving direct payments from the MPES scheme but profit from the sustainable approach towards tourism and natural resource management.

Management costs of MPAs are generally high and even though the BNMP is almost able to cover their expenses no data could be found on the actual transaction costs. Hence, no statement can be made if and how the transaction costs relate to the achieved benefits.

6.3.7 Results of the Assessment of BNMP

In the literature the case of the Bonaire National Marine Park was described as a success story. The application of the preliminary assessment framework shows that most of the success conditions are fulfilled. However, the characteristics of the BNMP and the relation between all stakeholders, including the government, intermediary and the general public have been favourable from the beginning on. The creation of the MPA on the island of Bonaire clearly harmonizes with other legal, local and regional, and governmental institutions as described above. STINAPA, responsible for the management of the MPA, can be seen as an intermediary and successfully implemented user fees, which helped the BNMP to become almost financially independent. The civil society and general public supported the establishment and development of the MPA. An important prerequisite was the environmental awareness, relatively small number of stakeholders, capacity, and relatively homogeneous groups. After the implementation of the BNMP the general public and the civil society have been involved in various policy- and decision-making processes. The acceptance of the MPA by the public is essential especially with regards to user- and tenure rights. Unequal power relations are not an issue and constant projects for education and capacity building contribute to a mutual understanding of the importance of the MPA for ecological, economical and social reasons.

To sum up the overall success of the BNMP can be linked to the following bullet points:

1. A supportive government which initiated and funded the establishment of the MPA.
2. A supportive legal and policy environment which harmonized with the MPA and other institutions.
3. An accountable and legitimate intermediary which manages the MPA.
4. Acceptance by and active participation of a capable civil society and broad public which is included in decision-making and monitoring processes.
5. Clear defined system boundaries
6. Secure economical benefits and a high dependency on the MPA by local communities which favoured a behaviour of stewardship among them.

7 Discussion

The case studies used had different characteristics in terms of location, size, time of establishment, type of payment scheme, involved stakeholders, and used ES. However, the different case studies shed light on the developed preliminary assessment framework and its applicability.

The cases of ITQs in New Zealand and of the BNMP are two examples of the successful implementation and adaptation of two different MPES schemes. In both cases most or even all elements of the developed framework have been present, even though to a varying degree. With regards to the case study of blue carbon sinks as an MPES schemes, the framework has been applied to a case which has not been implemented yet regardless of its high potential for climate change mitigation. In this case the framework helped to identify several bottlenecks, shortcomings and obstacles.

The foundation of the developed framework was the theory on terrestrial PES and was based on case studies, design principles, preconditions and their derived success conditions. Keeping in mind that the selected case studies do not represent the full scale of possible MPES and that the analysis was more in breadth then in depth several conclusions from the application can be made. The application of the framework showed, even though to a limited extend that the general statements over success or non-success from the literature for the individual cases are coherent with the forgoing predictions based on the presence of the different success conditions in the framework. Hence, the framework could be used for further applications in order to make a first statement, to assess or predict the potential success or to indicate shortcomings and bottlenecks of an MPES scheme.

Looking at the two cases of success, the ITQ and the MPA example, it becomes apparent that institutional factors in general play a major role for the implementation and final success of any MPES intervention. In both cases a supporting institutional environment has been in place from the beginning on and different stakeholders have been engaged in the design as well as in the further development of the MPES scheme. Institutional flexibility and adaptation to shortcomings have been shown in both cases in order to deal with trade-offs or other negative outcomes.

Another related important criterion was the development of some sort of stewardship behaviour of the ES providing participants, which encourages sustainable resource management and behaviour while giving incentives for active participation in the scheme. The case of the BNMP has a special position with regards to the development of stewardship behaviour. Since tourism is the biggest available local economy a high percentage is dependent on it. A degradation in environmental standards, biodiversity and/or beauty scenery could easily lead to a decrease in tourism. Hence, a sustainable management of the local terrestrial and especially marine ecosystem services can be directly linked to financial incomes and potential development through tourism. This high dependency is linked with the communities' capacity, education and environmental awareness and contributed positively to the successful implementation of the MPA.

Looking at the shortcomings and bottlenecks from the blue carbon example, which were mainly based on the complexity of institutional design and interplay, and the lack and unsecured conditions of tenure and user-rights, it becomes more apparent that the institutional context of any PES or MPES scheme is of high importance and should be one of the leading considerations before implementing or even considering such schemes.

The favourable institutional conditions in the ITQ and MPA case were complemented through a supporting government as well as legal and policy environment. However, it has to be stated at this point that both, the NZ ITQ example and the BNMP have been designed, introduced and initially financed by the government. Even though today each of the two MPES interventions has been developed further and certain changes have been applied, initially both started as a government-financed MPES schemes. Hence, in these cases the proactive and supporting behaviour of the governments can be explained. At the same time the implementation of the government-financed MPES schemes and the participation might not have been voluntary in all cases. Nevertheless, both cases have shown that they managed to involve most if not all important stakeholders and achieved their continuous participation. From the start on stakeholders have been included in the design of the QMS as for example the fishing industry in the development of the ITQ system and hence, gained increased trust and legitimacy.

On a broader scale the developed and applied framework also indicates that the terrestrial version of PES gives important examples and delivers valid information about success conditions, which can and should be used when dealing with Marine Payments for Ecosystem Services. Nevertheless, it is important to mention that due to the application of the framework, which was done on a broad scale; several other preconditions, success conditions or limitations might not have been identified and not included in the development of the framework. This lack of accuracy is due to the complexity and uncertainty of the topic itself and its interdisciplinary nature (including institutional, biophysical, social and economic conditions) but also based on the used approach. In addition, the framework is solely based on literature and no empirical research has been carried out to extend the focus of it. With regards to the importance of institutional settings for successful implementation of MPES and PES schemes, the framework could also be further complemented by Ostrom's (1990) institutional design principles for effective CPR management.

Considering the broad scale and limiting testing opportunities of the framework further testing (including other MPES types but also more examples of the same types) and refinements should be carried out. For now the framework does not specialise on the marine environment and is as valid for the marine version of PES as it is for the terrestrial one. Hence, a more narrow and detailed framework dealing with design and success conditions for each MPES type should be considered for more precise implementation guidelines and predictions.

From the elaboration of the different types of terrestrial PES interventions and through the analysis of the three MPES case studies it is possible to establish and differentiate between the three MPES types due to several characteristics. It is important to mention that the below listed MPES schemes do not reflect the full array of MPES types and only focuses on the typology of ITQs, MPAs, and blue carbon sinks, which have been subject to the analysis of this paper:

Individual Transferable Quotas:

ITQs are based on fisheries, which is a '*common pool recourse*', and its management. Using the classification of terrestrial PES schemes ITQs can be classified as an '*area-based*' MPES scheme which is based on agreements on pre-established caps on resource-use (Wunder, 2005). Hence, the resource-use is regulated through legislative means by the pre-established cap. On the created market the quotas can be traded freely under the established cap. If the ES are traded on the market a potential beneficiary pays a quota holder for the acquisition of a higher percentage of the TAC or in other words to be eligible to catch more fish.

After the initial funding and establishment of an ITQ by the government the MPES scheme can be characterised as a '*user-financed*' scheme. Direct trade of quotas are handled between users on a proper market. Even though the system works on a country wide level the actual transactions are taking place between users: one user gives up his/her right to fish to another entity.

The payment/compensation scheme in place exit out of two mechanisms. First the payment can be categorised as the '*recognition of rights*' and is based on the concept of providing participants or the ES stewards with the rights to participate in potential decision-making processes and to manage and harvest the fisheries. The second payment mechanism is the open trade of quotas on the market between different parties and can be classified as '*self-organized private deals*'.

Blue Carbon Sinks:

The provided services by blue carbon sinks are the long-term storage of carbon and the additional sequestration of CO_2. These provided ecosystem services can be characterised as '*public goods*' since it is impossible to exclude someone from consuming them and they are (at least to a certain extend) non-rival in consumption. Even though the primary ES provided by blue carbon sinks are the captured and stored carbon also the conservation of biodiversity are often provided simultaneously. The benefits of these services are distributed on a global scale.

Regarding the type of MPES blue carbon sinks can be divided into two groups. The type of MPES can either be '*use-restricting*' in the case of the protection of an already existing blue carbon sink or '*asset-building*'/'*use-modification*' if new carbon sinks are created in terms of reforestation of mangrove forests. In the case of '*use-restricting*' MPES schemes the ES provider will be compensated or rewarded for the emerging opportunity costs due to changes in land-use and management practices as well as for the protection of the area against external threats (Wunder, 2005). If an area will be converted into a blue carbon sink e.g. through the plantation and reforestation of mangroves one speaks of a '*use-modification*' type of MPES scheme. Here the ES providers or managers will be rewarded for the restoration of an area. 'U*se-*

modification' payment schemes are also subject to CDM are hence are carried out on a global level.

Also with regards to the funding of blue carbon sinks a differentiation can be made. Depending on the scale and whether it is a voluntary agreement or subject to compliance market the MPES can either be '*user-financed*' or '*government-financed*' respectively.

Marine Protected Areas:

The provided ES through the establishment of MPAs can be described as '*club goods*'. In general the provided ecosystem services are '*public goods*' such as biodiversity, carbon sequestration , environmental integrity, as well as beautiful scenery e.g. public beaches, coral reefs etc.. However, by making public goods more exclusive or in other words by excluding people from enjoying the provided services for example by limiting the number of allowed visitors and through the introduction of entrance-fees some of the pubic goods are turned into 'club goods'. ES such as biodiversity and carbon sequestration which are also often linked to MPAs have a public good characteristic.

MPAs can be classified as a '*area-based*' MPES type since they provide several ES created within the protected area. However, if ecotourism is one of the provided services it can also be described as a '*product-based*' MPES scheme in which visitors and tourists pay for the provided services of landscape beauty directly. Here the tourists as private actors pay (often) an intermediary for the provisioning of the services (Wunder, 2005).

With regards to the funding of MPAs a further differentiation can be made. Usually MPAs are '*government-financed*' or at least induced by the government. If, however, an MPA becomes financially independent due to user-fees they become '*user-financed*' MPES schemes. This is especially true for ecotourism as the main income and compensation for adopted conservation and protection measures.

The typology above does not state or indicate any details about the potential or likelihood of success of MPES schemes but is an important classification of characteristics of the different types. However, the potential success of each of the above mentioned cases can be enhanced by using and applying the developed preliminary assessment framework. Hence, the framework and the typology act as a guidance for design principles for MPES interventions in general and a MPES case specific classification and characterisation respectively.

8 Conclusions

The current trends of marine and coastal ecosystem degradation, overfishing as well as the loss of habitats, biodiversity, and community livelihoods indicate the lack of effective marine and coastal protection and conservation measures. Marine Payments for Ecosystem Services can be regarded as a new form of marine and coastal conservation combined with economic incentives. Instead of traditional top-down and command and control mechanisms, MPES interventions aim at the provision of positive environmental externalities through the compensation for changes in land-use practices. Hence, more equitable and sustainable outcomes can be achieved. MPES schemes can contribute towards marine and coastal conservation in different ways and situations. The various types of MPES are

1. compensation of private resource and coastal landowners by a public entity e.g. protection of mangrove forests within a carbon payment scheme,
2. establishment of a formal market, which enables open trading between beneficiaries and suppliers e.g. through the establishment of a ITQ system and a market for the free trade of quotas.
3. self-organized private deals e.g. blue carbon offsets based on a voluntary agreement between private parties, and
4. MPAs combined with user-fees.

The terrestrial version of PES has been around for a longer period of time and several experiences, conclusions, and lessons-learnt have been derived from their implementation. The developed framework in this research project was based on identified key factors and success conditions derived from a desk research on PES schemes.

Testing the preliminary assessment framework on three different MPES case studies showed the applicability and feasibility of the developed framework. As described in the discussion the broad design of the framework allows the analysis of success conditions of different types of MPES schemes. The success of two different case studies, the ITQ case in NZ and the MPA on Bonaire, and the non-success/bottlenecks (until now) of a third case study could be explained by applying the preliminary assessment framework. Keeping in mind the broad scale and other limitations of the developed framework it was useful in supporting the hypothesis that success conditions of PES can be used while holding valid and important information for the implementation of MPES. Based on success conditions of terrestrial PES the framework showed that certain aspects of institutional, biophysical, social, and economic conditions are transferable form PES to MPES schemes. Each category had to be considered carefully but with the application of the framework it was possible to allocate certain weight to some of the success conditions.

Apart from ensuring financial sustainability of the project the economic incentives and compensation for the ES providers have to be assured. The ES providers have to be compensated for their actions and changes in land-use practices in a way that their opportunity costs are covered and that their economic income is secured on the long run.

However, special attention has to be paid to the institutional context of a payment scheme. It has been shown that a supporting government as well as a supportive legal and policy environment are important preconditions for the implementation of MPES. Also the involvement of all stakeholders from the beginning on and means of encouraging their active participation in decision-making processes are essential. Among the most important features of any PES or MPES scheme are secured long-term tenure, property or use-rights. They can be regarded as essential to any PES scheme. With regards to MPES, however, clear property rights are even more problematic than for the terrestrial version due to the characteristics of many ES that can be regarded as common pool resources or public goods. Additionally, some of the ES like fish are mobile and the benefits might be felt in a different location. User- or property rights are essential to encourage the behaviour of stewardship over a resource or ecosystem.

The motivation behind stewardship is a clear link between ES provisioning and the received compensation from the ES beneficiary. Increased stewardship could lead to more sustainable outcomes. In the case of the ITQ system in New Zealand the property rights over fisheries have been artificially created and lead to an increase behaviour in stewardship which contributed to the successful implementation of the MPES scheme. A similar increase in stewardship behaviour was also observed in the case of the BNMP. The actions of the stakeholder could be linked to the provisioning of the ES which is the essence of the marine park and therefore essential for the local economy. The case of blue carbon sinks as MPES emphasized that the importance of tenure rights and the inclusion of local communities not only for the success of the payment scheme but also as a precondition for the implementation.

The experience gained from various PES cases is valid and should be considered carefully before and during the implementation of MPES. The success of MPES highly depends on preconditions but as shown in the case of New Zealand and Bonaire could contribute towards sustainable coastal and marine conservation. Nevertheless, MPES do not work in any given situation and are not a panacea for coastal and marine conservation. Especially with regards to the livelihoods of local communities wrongly established MPES schemes could result in controversial effects and could generate negative effects for local communities and indigenous people.

In order to make sound and reliable predictions about the affects and effectiveness of MPES more attention and research is necessary. Any MPES scheme should contribute towards good governance and increase local capabilities for natural resource management practices. Attention should be paid to the institutional conditions and their interplay. Good and effective local institutions are essential for achieving sustainable management practices for natural ecosystems. For this reason, especially tenure, property and use-rights have to be considered and further investigated in order to reach equitable and sustainable solutions which include local communities, their social implications, economic development and marine and coastal conservation.

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