IVM Institute for Environmental Studies

The total economic value of nature on Bonaire

Exploring the future with an ecological-economic simulation model

Jorge Amrit Cado van der Lely

Pieter van Beukering

Lidia Muresan

Dario Zambrano Cortes

Esther Wolfs

Stijn Schep

Report R12-XX
02 January 2013

This report is released by: Pieter van Beukering

Associate Professor, Environmental Economics

This report was commissioned by: Ministry for Economic Affairs, Agriculture and Innovation
It was internally reviewed by: Luke Brander

IVM
Institute for Environmental Studies
VU University Amsterdam
De Boelelaan 1087
1081 HV AMSTERDAM
The Netherlands
T +31-20-598 9555
F +31-20-598 9553
E info@ivm.vu.nl

Commissioned by:
Mr. Hayo Haanstra
Ministry of Economic Affairs
P.O. Box 20401
2500 EK The Hague
The Netherlands
T
F
E

IVM Institute for
Environmental Studies

VU UNIVERSITY AMSTERDAM

Contents

Summary

Healthy ecosystems such as coral reefs and mangroves are critical to Bonairean society. In the last decades, various local and global developments have resulted in serious threats to these fragile ecosystems of Bonaire, thereby jeopardizing the foundations of the island's economy. Therefore, it is crucial to understand how nature contributes to Bonaire's economy and its wellbeing in order to make well-founded decisions when managing the economy and nature of this beautiful tropical island. This research aims to determine the economic value of the main ecosystem services that are provided by the natural resources of Bonaire and their overall importance to society. The challenge of this project is to deliver sound scientific insights that will guide decision-making regarding the protection of Bonaire's ecosystems and the management of the island's economy.

By assigning economic values to the main ecosystem services of Bonaire, this research draws attention to the economic benefits of biodiversity and highlights the growing costs of biodiversity loss and ecosystem degradation. The study addresses the most relevant ecosystems and ecosystem services for Bonaire and applies a range of economic valuation and evaluation tools. By surveying over 1,500 persons, including tourists, fishermen, local residents, and citizens of the Netherlands, this study estimated the willingness of individuals to pay for the protection of Bonairean nature, as well as mechanisms (e.g. user fees) through which such payments would be transferred. Furthermore, a scenario analysis is conducted to inform decision makers about the most effective strategies to protect the ecosystems of Bonaire. This study intensively involved stakeholders from the start to finish, which facilitated data collection while simultaneously building capacity in applying the concept of ecosystem services among the target audience.

In total, more than 10 different ecosystem services have been valued in monetary terms. The total economic value (TEV) of the ecosystem services provided by the marine and terrestrial ecosystems of Bonaire is $105 million per year. This TEV and its underlying components can be used to build a strategy for effective conservation measures on Bonaire. After extensively analyzing different scenarios for future ecosystem services values one result becomes very clear: an ounce of prevention is worth a pound of cure. In other words, it is more efficient to prevent extensive environmental damage than trying to revitalize the environment while there are still threats at hand. With the current threats unmanaged, the TEV of Bonairean nature will decrease from $105 million today to around $60 million in ten years time and to less than $40 million in 30 years. The project is well documented and provides several extensive online reports, five easily accessible policy briefs and a beautiful film documentary that translates the scientific results into real life situations on Bonaire.

List of abbreviations

BNMP – Bonaire National Marine Park

CBA – Cost Benefit Analysis

CPV – Coastal Protection Value

CT – Cruise Tourists

FPA – Fisheries Protected Areas

GDP – Gross Domestic Product

N – Nitrogen

Nb - number

NPV – Net Present Value

P – Phosphorus

SOR – State of the Reef

SOT – Stay-over Tourists

TEV – Total Economic Value

TIN – Total Inorganic Nitrogen

USVI – United States Virgin Islands

WTP – Willingness to Pay

Acknowledgements

This study would not have been possible without the support of numerous people and organisations on Bonaire. First of all, we want to thank ministry of Economic Affairs, Agriculture and Innovation Caribbean Netherlands, especially Hayo Haanstra, Astrid Hilgers, Ruth Schipper-Tops and Pieter van Baren for making this research possible in the first place and special thanks to Paul Hoetjes, for facilitating the study and for helping us overcome hurdles that we encountered during the course of the study. Additionally, we would like to thank the Directorate of Spatial Planning and Development, Unit Nature and Environment, especially Frank van Slobbe for his valuable input, and special thanks to Boudewijn 'Bous' Scholts Carbo TTC Inc for validating our report. Other people that helped us are Kris Kats, Jozef van Brussel and Jan Jaap van Almenkerk from ministry of Infrastructure & Environment the Netherlands. A special thanks to Danilo Christiaan, Irida, Sue, Stephanie, Carine, Alan and Mikey from Mangazina di Rei. A thanks to our artist who draw the beautiful pictures for the Choice experiments, Mechtild Thode and a thanks for the support of her husband Glenn Thode when he was the island governor of Bonaire. For the information from the tourism industry a special thanks to all businesses that have participated in the business survey and a special thanks to Joanny Trinidad and Marjolijn van Kooten TCB and Lara Chirino, Irene Dingjan BONHATA, Martien van der Valk Bonaire Hospitality Group and Bonaire Chamber of Commerce, to Tante Vita and Papi Cecilia, Bas Noij Bonaire Explorer Association, Bart Snelders CURO. And a very special thanks to Corine Gerharts and Jan Baten Bonaire Tours & Vacations and their team for their incredible support together with Yvonne and Chris Schultheiss Bonaire Destination Services and their team for helping out to collect many cruise passengers data. Also a special thanks to all the guides of Sue Felix Archie tours Bonaire. And a thank you to Augusto Montbrun and Paul Coolen Buddy Dive resort to show us the coral nursery.

A thank you to all the people who have supported in collecting data, especially Jan Kloos and Rob Sint Jago Island Territory harbour and Hans Voerman Outdoor Bonaire, and Elly Albers Mangrove Center. And a thank you to all the fishermen who took the time to talk to us, especially Silvio Domacasse, Jason Muller and Pancho Cicilia

Furthermore special thanks for the support by Erik Meester, Dolfi Debrot and Diana Slijkerman IMARES for answering all kinds of ecological questions during our research and delivering us data. And thanks for accommodating our students and supporting our meetings by Rita Peachey at CIEE Research Station.

We would also like to express our gratitude towards all the interviewers who made this study possible. Thanks to Anna Maira Vaseur, Harrold Vasseur, Timothy Vaseur, Jarelys Cecilia, Janin Martes, Vernon Kromhout.

1 Introduction

In the current era of financial insecurity and environmental degradation it becomes clear that conventional investments in the economy do not always contribute to a healthy environment. In response, a growing community is working to show that the economy and environment are strongly interlinked, and that in fact a healthy environment is critical to financial as well as human wellbeing. This is especially the case for the island of Bonaire, for which the main sectors in the economy are strongly nature-dependent. Bonaire's unique nature is very diverse. The coastal waters contain coral reefs, mangroves and sea grass systems, and on land the island is characterized by dry forest and farmland (Kunukus). Historically, Bonaire's inhabitants lived in balance with this natural environment. However, many pressures including the fast economic development of the island have led to environmental degradation and a loss of the ecosystem services of which the people of Bonaire take benefit. Therefore, it is crucial to understand how nature contributes to Bonaire's human wellbeing.

The importance of nature to the economy of Bonaire has become an even more crucial issue due to the recent change in constitutional status of the island. Since 10 October 2010 Bonaire, Saba and St Eustatius (Statia) are part of the Netherlands. These three islands are referred to as the Caribbean Netherlands. The islands in the Caribbean Netherlands now have the constitutional status of special Dutch municipality. For both the Netherlands and the Caribbean Netherlands the new constitutional arrangement has major and policy[1] and nature-related implications. A unique and significant area of high value nature and stock of biodiversity is added to the Netherland's Kingdom. As shown in Table 1.1, the Caribbean Netherlands measures more than 2,800 km^2 of marine reserves. For the Netherlands this is implies a substantial expansion of nature. Politicians and policy makers commit Dutch governmental budget to important policy issues, of which a limited share is earmarked for conservation and preservation of the unique and endemic nature on Bonaire.

There is general concern about the future of Bonaire. The local government is working hard to preserve the natural and cultural heritage of the island. At the same time the local government plans to increase tourism, build casinos, restaurants and piers and other recreational facilities. In light of these plans and the current environmental state of Bonaire, it is paramount that during the development of the economy, a sustainable approach is implemented. This is evident in the fact that the economical benefits acquired from the goods and services an ecosystem provides are depended on the qualitative state of the ecosystem in question. If one system is affected so is the other.

To determine a sustainable approach and ensure economic development of the island of Bonaire, this study conducts a socio-economic evaluation of Bonaire's. The use of a dynamic simulation model to analyse ecological and economical processes will provide insight into how these two systems influence one another. The use of this model also allows the evaluation of different interventions, i.e. interventions that are aimed at improving the environment or at improving the local economy. Each intervention has costs and benefits, both ecologically and economically. Using different valuations techniques (which are explained later on the report) the monetary value of each ecosystem service is calculated. The main goal of this report is to determine the economic costs and benefits of different intervention with the aim of ensuring a sustainable economic development for future generations.

Table 1.1 Characteristics of nature in the Netherlands' Mainland and the Caribbean Netherlands

1 The new legal status of the islands in the Caribbean Netherlands affects local environmental legislation, policies and regulations. Local residents start paying tax to the Netherland's treasury but are also more entitled to claim government service and support.

Nature indicator	Netherlands Mainland	Caribbean Netherlands
Area of terrestrial nature parks	12,685 km^2 (is 30% of total area)	49.4 km^2 (15.7 % of total area)
Area of marine nature parks	2,330 km^2 (is 4% of total area)***	75 km^2 (0.3% of total area) With Sababank = 2,754 km^2 (11% of total area)
Number of animal species*	27,000	2,831****
Number of endemic animal species	14**	85**** of which 25 in Caribbean Netherlands
Number of plant species*	3,900	1,259****
Number of endemic plant species	0	7****

Sources: Dutch Caribbean Nature Alliance, 2012; Staatsbosbeheer, 2012; WUR, 2012.
* Note however not all species are known and new species are still being discovered.
** www.natuurinformatie.nl names 2 species of sponges and 10 ciliary worms and one mouse subspecie and a butterfly.
*** 3 protected areas in the North Sea are in the Exclusive Economic Zone; Vlakte van Raan (17,521 ha), Voordelta (92,367 ha) and North Sea Coastal Zone (123,134 ha). Total area Dutch North Sea is 57,000 km^2.
**** Number of species in Dutch Caribbean (including Aruba, Curacao and St Maarten).

This report is structured as follows.[2] Chapter 2 provides important background information about the island of Bonaire, including the current ecosystem services found there, a more detailed explanation of the current threats on coral reefs as well as the boundaries and limitations of this study. Chapter 3 describe the overall context and approach for building the dynamic simulation model of coral reefs ecosystem along with their ecological and economic benefits. Chapter 4 presents a detailed construction of the model and each sub-module. The results obtained for the baseline scenario and the three other scenarios are presented in Chapter 5. The conclusions and the recommendations are drawn in Chapter 6.

2 Please note that some of the estimated effects may deviate from the values reported in the individual research reports of this overall study on the value of nature of Bonaire. These differences are explained by the technical limitations of the model structure as well as the difficulty in coordinating the completion of the individual studies and the modelling activities which took place at the same time. Overall, these differences do not fundamentally affect the outcome of the model simulations.

2 Background

2.1 Economy and demography

The Caribbean Archipelago includes the Netherlands Antilles (800 km^2), which are divided in two groups of islands: the leeward group which incorporates the islands of Aruba, Bonaire and Curaçao, also known as the ABC islands, and the windward group which consist of the islands of Saba, St Eustatius and St Maarten (see Figure 2.1). Bonaire is located 46 km east from Curaçao, 80 km north from Venezuela and 129 km east from Aruba. The surface of Bonaire is 288 km^2 plus another 6 km^2 for the adjacent island of Klein Bonaire. It measures 38 km from North to South and a maximum of 11 km wide from East to West (Wolfs, 2011; CBS, 2005). The capital is Kralendijk, the biggest city on the island. Population count varies from source to source but according to *the Centraal Bureau voor de Statistiek* (CBS, 2012), after the census from 2010 there were 15,666 people living on the island.

Figure 2.1 – Location of Bonaire in the Caribbean Sea (source: CBS, 2010)

The number of households in Bonaire is 5,336 according to CBS (2010), but this number is expected to increase in the coming years. In 2009, approximately 296 building permits were issued (CBS, 2010).This number varies due to immigration and emigration. The total number of immigrants registered in 2010 was 1,200 while the number of emigrants was approximately 1,028. The principal country of origin for both immigrants and emigrants is the Netherlands (CBS, 2011). About 86% of Bonaire residents are Dutch, while the remaining 14% have different nationalities such as Dominicans, Venezuelans, Colombians and Peruvians (CBS, 2005). The official language is Dutch. The local language Papiamentu, includes elements of English, Spanish, Portuguese, African and Dutch. This language is common in Bonaire, as well as in Curaçao and Aruba (CBS, 2010).

The Bonaire's GDP in 2008 was 224 million USD (CBS, 2010). Bonaire's principal economic pillar is represented by the tourism sector which increases fast from a year to another (CBS, 2005). The sectors, which contributed the most to the island income, are presented in Figure 2.2 (CBS, 2005).

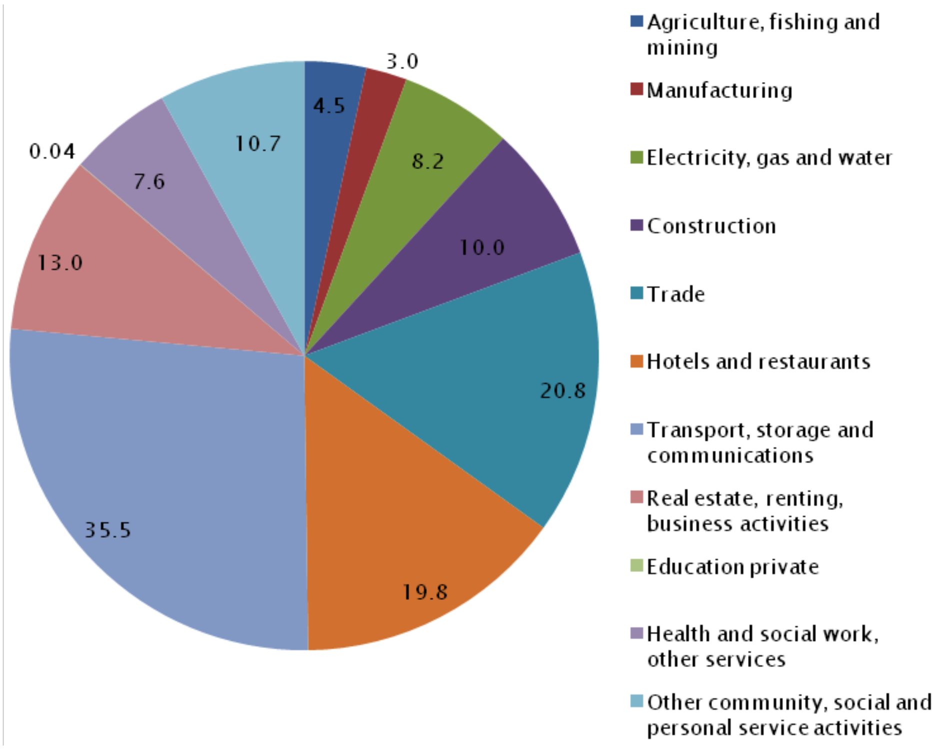

Figure 2.2 – Contribution of different sectors to Bonaire's GDP in 2008
Source: CBS, 2010

Visitors are attracted by the unique combination of terrestrial and marine ecosystems and the variety of activities they can enjoy on the island, such as diving, snorkelling, kayaking, windsurfing, sailing, bird watching, etc (Info Bonaire, 2012). As a result of the high number of visitors, the construction sector almost doubled and there are plans to further increase the number of houses and accommodation (Bonaire Department of Physical Planning, 2010).

2.2 Nature and ecosystems

The climate of Bonaire is arid tropical, fairly constant throughout the year with low rainfall (about 463.3 mm/year registered) and high temperatures during the year, varying between 26.6˚C and 28.4˚C (MSNA&A, 2008). This climate allows the existence of large and diverse ecosystems, both off and on land. On land Bonaire is characterized by dry forests and off land coral reef ecosystems predominate. Of the world's coral reefs, 8% are located in the Caribbean Sea and occupy a surface of 26,000 km^2. On Bonaire the total area covered by corals is approximately 27 km^2. Seventy different species of hard corals can be found in the Caribbean, 65 of which were identified in Bonaire (IUCN, 2011; Alevizon, 2009). Compared to the rest of the Caribbean the coral cover of Bonaire is relatively well preserved and represents one of the healthiest coral reefs in the Caribbean.

Bonaire also has a substantial terrestrial ecosystem, which mainly consists out of dry forest. Just like the coral reefs so have these dry forests experienced excessive stress. A long history of grazing, felling and clearance for cultivation have already destroyed approximately 66% of the dry forest in Latin America (Quesada et al 2009). Bonaire is no exception. Trees were felled (in particular *Haematoxylon brasiletto, Zanthoxylum flavum and Guaiacum officinale*) throughout Bonaire in the 17th century and large grazers such as goats, sheep, donkeys, cattle and horses were introduced and left to roam. Later in the 20th century, extensive deforestation for the cultivation of Aloe and the urban expansion for tourist facilities was completed (De Freitas et al 2005). Today, estimations are that around 30,000 goats are roaming free on the island, most of the traditional gardens, Kunukus, a traditional farming system are abandoned, agrarian industry is almost inconspicuous and original ecosystems occupy less than 30% of the island, which is in a degraded state.

The island of Bonaire has 5 Ramsar sites: Klein Bonaire, Pekelmeer, Salina Slagbaai, Gotomeer and Lac, and two National Parks, a terrestrial and a marine one. The terrestrial park, Washington Slagbaai National Park (WSNP), established in May 1969, has a surface of 5.6 km^2 and protects approximately 17% of the total land area of Bonaire. Different species of birds and reptiles, such as parrots, flamingos and iguanas can be found within the park (STINAPA-WSNP, 2012; Wolfs, 2011). The vegetation on Bonaire is drought resistant, and adapted to its climate. Most of the plants have thick leafs, water storage tissues or change their angle to avoid direct sunlight (STINAPA Bonaire, 2008).

The Bonaire National Marine Park (BNMP) was established in 1979 and is recognized by the International Coral Reef Initiative as "one of the best-managed marine parks in the world". It surrounds the island of Bonaire and Klein Bonaire up to 200 m from the coast and 60 m in depth. BNMP is managed by a local NGO, called STINAPA Bonaire, which provides education, monitoring, and research of Bonaire's biodiversity. To administer the park, an annual admission fee was established in 1992, for divers and snorkelers of $25 and $10 respectively (WRI.com; Thur, 2010). Bonaire's marine ecosystem is unique with regard to its species. The park consists of 2,700 ha of fringing coral reef, seagrass and mangrove ecosystem. Nevertheless, coral reefs present a fundamental structure for the majority of marine organisms. Of the 450 species of reef fish, the most common are Blue Tang, Bicolor Damsel, Stoplight Parrotfich, Brown Chromis and Bluehead Whasse. Different species of algae such as Sea Pearl and Mermaids Tea Cup can also be found in Bonaire's waters (IUCN, 2011). According to IUCN (2011) there are about 65 species of hard corals.

2.3 Threats and impacts

The last decades Bonaire's natural environment has experienced stress from both human activities and natural occurring events. The effects of which can be seen by a decline in coral cover throughout coastal waters and the lack of mature dry forests present on the island. Two hurricanes (Lenny in 1999 and Omar in 2008) caused a substantial amount of damage to the coral reefs. The continual expansion of humans on the island, the deforestation and the great numbers of free roaming live stock have put the terrestrial system under great stress. Coastal development and increased nutrient discharge also contributes to the further degradation of the marine environment.

The declining quality of the coral reefs on Bonaire follows a global trend. About 27% of the world's coral reefs in 2000 were in such a degraded state that recovery was highly unlikely. Expectations are that this number is going to increase even further (Parsons & Thur, 2007). A meta-analysis conducted by Gardner *et al.* (2003) revealed that from the 1970s until 2003, 263 study sites showed a high decrease of coral cover from ~50% to ~10%. In terms of biodiversity it is well known that islands have a natural vulnerability to extinctions which are accelerated mainly by habitat loss and invasive species (Blackburn et al 2004, Traveset and Richardson 2006). Modest transformation represents a threat on islands because scarce resources reach critical levels easily. The main driver behind the debasement of coral ecosystems are of anthropogenic origin. Unsustainable fisheries, pollution, sediment runoff, physical destruction and climate change all exacerbate the degrading state of corals. Notwithstanding the severity of such human induced stressors, natural events such as storms, hurricanes or coral diseases are also of great importance in the global decline of corals (NOAA, 2011). Below follows a detailed summary of the most important environmental threats on the islands.

Over fishing

Fishing techniques, like the use of explosives and overfishing of specific species, contribute to a decline of coral reef over the world (NOAA, 2008). One third of the reefs around Bonaire and Curacao are threatened by overfishing (WRI.com). Little information is available about what the effects of over fishing might have on species diversity. Nowadays, fishing practices focus on smaller predators such as groupers and herbivores like parrotfish (Burkepile & Hay, 2008). Due to the lack of herbivorous fish, algae have the possibility to grow unhindered hereby contributing even further to the stress corals experience. This makes corals even more vulnerable to diseases and death (ICRI; Debrot & Bugter, 2010). High levels of fishing can reduced genetic variation (due to specific species being overfished), alter ecological balance on the reef and change trophic interactions (McGinley & McClary, 2010; WRI.com). However, since 2004 two FPAs were established at the request of BNMP. These two areas represent 4 km of a no-take zone (The Nature Conservancy, 2012).

Physical destruction

The tourism sector represents a threat to the marine ecosystems of Bonaire. Due to the activities performed by tourists, such as diving and snorkeling, approximately 2.7% coral reefs are damaged every year (De Meyer, 1998). These activities have a direct impact on corals as a result of their direct contact or illegal anchoring (WRI.com). Since 1994 tourist's number increased from 57,000 to approximately 70,000 (TCB, 2010). However, the physical destruction of coral reef remained stable. This is a result of the educational and awareness programs on coral fragility (De Meyer, 1998).

Sedimentation

Sedimentation is mainly caused by the dredging associated with construction of different types of buildings and development of infrastructure. As an impact, the sediments released in water can affect the food web by killing the corals and other organisms essential for fish. Sediments also reduce the photosynthetic activity and light availability, and in high amounts they can even bury the reefs (Roger, 1990; Wieggers, 2011). The issue of sedimentation began with the expansion of tourists since 1994 (De Meyer, 1998; Harty, 2011). More tourists triggered a higher coastal and marine development which caused high levels of sediments being released into the water. Furthermore, deforestation and overall decline in terrestrial ecosystem has attenuated the capacity of the forest to prevent sediment run off.

Nutrients

Coral reef systems are characterized by oligotrophic conditions. It is in these conditions that corals have a competitive advantage. However, superfluous amounts of Nitrogen (N) and phosphorous (P) of anthropogenic origin, result in eutrophic coastal waters. Such conditions are favourable for algae and allow them to out-compete coral for space (Wieggers, 2011). The sewage water of Bonaire is collected in septic tanks and leaches into the sea through groundwater without being treated properly. In their paper, Kekem et al (2006) mentioned the main reason for coral reef decline to be the inflow of partly untreated surface and subsurface water. The faeces of free roaming live stock (e.g. goats and donkeys) also represent a source of nutrients (Kekem *et al*., 2006). Since Bonaire never had a sewage treatment plant it has become an important source of stress for corals. Especially now as the number of tourists increased and the concentration of nutrients entering coastal waters with it. A study done by Dailer *et al*. (2012) for Hawaii analyzed the effect of N and P on diverse species of algae, using different concentrations of nutrients. The outcome of this study revealed that the growth rate of algae increases with the percentage of wastewater affluent added.

Lionfish

Lionfish (*Pterois volitans* and *Pterois miles*) is a non-endemic fish species that has been one of the main sources causing a decline in fish populations. The first lionfish captured in Bonaire was in October 2009. Since then, their numbers have increased substantially. Known as predators, they disrupt the functioning of coral reef ecosystems resulting in a decrease of fish biodiversity. Many programmes have been developed to raise awareness and ask tourists to report their presence (Mumby et al, 2011). Lionfish represent a threat to reef fish, fish gut analyses has proven that their diet consist of juvenile fish. It seems that endemic fish species have not yet adapted to the presence of lionfish. In the Bahamas, lionfish reduced the number of coral reef fish by 80% (Vermeij, 2012). Lionfish are characterized by rapid expansion as a result of the limited number of natural predators they have.

Climate change

Sea level rise, increased water temperature, a higher frequency of hurricanes and an increase in ocean acidity are part of the IPCC scenarios[3] for climate which represent serious threats for coral reefs. Global warming can make coral reefs more vulnerable to diseases, affect their resilience capacity and can also kill the corals (Debrot & Bugter, 2010). An example is from 1982 and 1994 in Indonesia and the Pacific when almost half of the bleached corals died (Hoegh-Guldberg, 1999). During the 20th century, the average temperature of the world oceans increased by 0.74˚C (Hoegh-Guldberg et al, 2007). It is considered that coral reefs are already at their thermal limits and a further increase will lead to their bleaching, disease and mortality (Hoegh-Guldberg et al, 2007). In tropical environments, usually coral reefs experience water temperatures between 18˚ and 30˚C. Below and above this temperature, coral reefs are affected and are threatened by overgrown macroalgae (Hoegh-Guldberg, 1999).

[3] Scenarios proposed by IPCC include an increase in average air temperature with 1.1˚C to 6.4˚C; an increase in the level of precipitation in some parts, and decrease in others; sea level rise by 18-59cm and an increase in ocean acidity by 0.14-0.35pH.

3 General Approach and Methodology

3.1 Valuation of ecosystem services

Nature provides a wide range of benefits to people. For millennia, human beings have benefitted from some processes intrinsic to the functioning of ecosystems worldwide. These ecosystems generate a range of goods and services that support human well-being and generate economic benefits, collectively termed ecosystem services. The Millennium Ecosystem Assessment (MA 2005) introduced the concept of ecosystem services on the global agenda, recognizing four categories of services: supporting (e.g. nutrient cycling, soil formation and primary production), provisioning (e.g. food, freshwater, wood and fibre and fuel), regulating (e.g. climate regulation, flood and disease regulation and water purification), and cultural (aesthetic, spiritual, educational and recreational). Globally, about 30 million people depend entirely on coral reefs for their livelihood (NOAA, 2011).

Despite the fact that many people benefit from the ecosystem services, individuals or groups usually have insufficient incentives to maintain the natural capital, compromising ecosystems for continued provisioning of benefits (TEEB, 2010). Usually the flow of services and goods from nature to humans is undervalued by governments, businesses and the public and is only considered once they have been lost. Changes in ecosystems and the services they provide have impact on human welfare and wellbeing. These changes, be they intentional or accidental, affect provisioning, regulating, habitat and cultural services. By analysing how these changes affect the values of ecosystem services thus provide information on how to manage the environment. Furthermore it provides a means to communicate the value of ecosystem services in a comprehensive, objective and logical manner to all relevant stakeholders.

In order to help policy makers make comprehensive decisions concerning the management of ecosystem services, a series of steps must be taken.

1. The ecosystem services within the study area must be defined. What are economical and ecological benefits and goods delivered and provided by the ecosystem?
2. The relationship between the ecosystem services and the ecosystem must be defined. Which aspect of the ecosystem delivers the goods and services? This relationship provides insight into which ecological/biological parameters are of importance for the provision of the ecosystem services and goods. For example, in order for the fishers to prolong their practice they need to be able to catch fish. Thus the service would be defined as “fish” and the ecological parameter associated with this service would be “fish biomass” or “fish stock”.
3. Define the factors that influence the capacity of the ecosystem to provide the service or good. Returning to the previous example, “fish stock” is influenced by factors such as: fishing rate and other anthropogenic stressors (pollution, destruction of habitat etc), predation, availability of food, growth rate fish and reproduction rate fish.
4. Identify different interventions, each of which should elicit and highlight a different aspect, hereby juxtaposing the status quo, ecological orientated interventions and economical orientated interventions. Choosing a set of different interventions will provide insight in how they influence the ecosystem and thus, the ecosystem services and goods.

3.2 Simulation model

The functioning of ecosystems, its delivery of services and the final contribution to welfare is complex. To effectively evaluate the complex interface between ecological and economic processes, simulation modelling can play a useful role representing the main ecological functions and the interaction with the economic sectors. It is therefore important to critically asses what type of model and software to use.

In its broadest sense, a simulation is a tool to evaluate the performance of a system, existing or proposed, under different configurations of interest over a specified time frame. The model quantifies the changes of each intervention on ecosystem services provision and the output provides decision makers with information about costs, benefits, trade-offs, synergies and opportunities. Modelling is characterized by the practice of representing trends and physical processes in a rational and objective way (Systems Management College, 2001). It is important to realize that models are an oversimplified manifestation of reality.

There are several modelling techniques that allow a quantitative approach for the evaluation of the network of ecological and economic interactions. Models can be classified as deterministic when input and output variables are fixed values, stochastic when at least one of the input or output variables is probabilistic, static when time is not taken into account and dynamic when time variation is considered. A static model provides information about the system at a precise time, while a dynamic simulation model offers information over time and can show how a phenomenon will perform (Carson & John 2004, Systems Management College, 2001). Prevost et al, (2005), made a comparison between a static and dynamic simulation model to forecast the abundance of salmon in the River Bush (i.e. North Ireland). The results obtained from these two models revealed that dynamic models are a better choice for predicting ecological indicators because they are more flexible.

In economic sciences, modelling provides the opportunity to experiment logically, produce different scenarios, and evaluate the effect of different policy options. In the analysis of economical systems four types of models can be used: visual, mathematical, empirical and simulation models. Visual models are simplified graphs of a theoretical economy, while mathematical models are illustration of synchronized equations and diverse variables. Empirical models are mathematical models that use data gathered as variables, and simulation models represent mathematical equations in a transparent form to the user (Evans, 1997).

To reach the final purpose of this study a dynamic simulation model was built to present the relationship between ecology and economy. Empirical data was used to quantify the ecological and economical parameters within the model. The STELLA software[4] (Costanza & Voinov, 2001; Costanza & Gottlieb, 1998) was used to model the relation between ecology and economy on Bonaire. Using STELLA, the model analyzes the impacts of different interventions in a transparent and easy way, while as a dynamic model it offers the possibility to vary the parameters and simulate the changes of each intervention over time (Prevost et al, 2005).

3.3 Conceptual framework

Figure 3.1 represents the conceptual framework of the simulation developed for evaluation of management interventions in the ecological-economic domain of Bonaire. In Step 1, the qualitative state of both the terrestrial and marine ecosystem is defined. Specific ecological parameters of the terrestrial and marine ecosystem define their qualitative state. Using the different parameters, the ecosystem is simulated. The output of the simulation is the general state of both environments. Step 2 defines the economic benefits of the ecosystems services provided and sum them up to calculate the Total Economic Value (TEV) of nature of Bonaire. Step 3 and 4 identify the current threats and measure their effects on ecological indicators and on the economic value of corals over a time period of 30 years. Moreover, exogenous variables such as the global economy and an increase in Bonaire's population are also analyzed. Step 5 identifies

4 Stella Software is a simulation model that makes use of differential equations to make a dynamic model of economical and ecological processes. Furthermore it offers the chance of visualizing how these models work. (IseeSystems, 2012)

potential intervention measures and estimates their influence on ecological conditions and its subsequent impact on the economic value of nature. This value is compared to the TEV of the baseline scenario. Step 6 compares the costs and benefits of each management option and calculates the net present value (NPV) the management interventions.

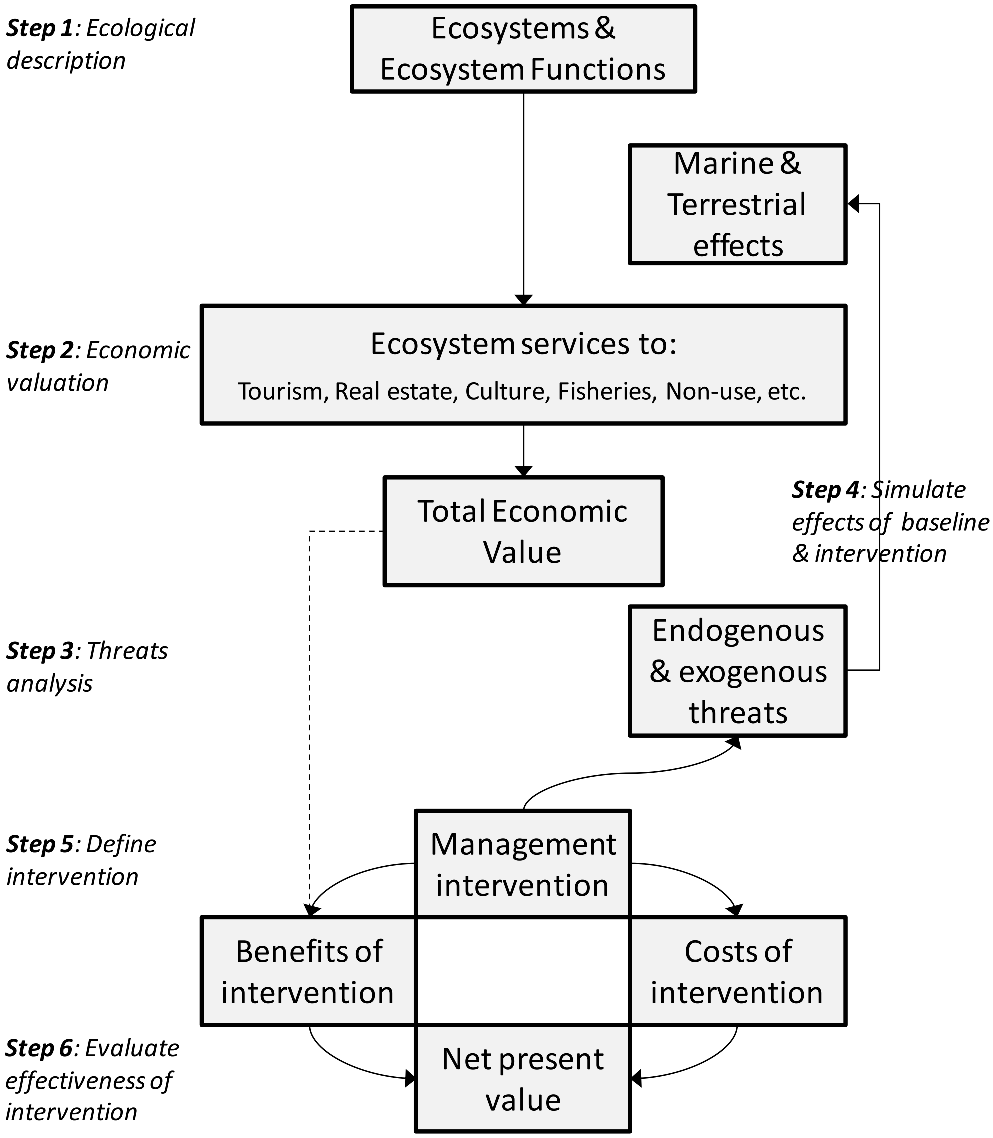

Figure 3.1 General Framework of the dynamic simulation model

3.4 Ecosystem services and economic benefits

Ecosystems generate a range of goods and services, known as benefits for Bonaire's society. As mentioned in Cesar et al (2002), goods provided by an ecosystem can be seen as renewable and non-renewable. Renewable goods can be lumber, fish or seaweed, and non-renewable goods are represented by sand and corals extracted and used as building materials. Coral reefs, for example, also provide a range of services such as: physical structure services for coastal protection (CP), biotic services within and between ecosystems for maintaining the habitat, bio-geo-chemical services for nitrogen fixation and CO_2 control, information services for climate and pollution control and social and cultural services for tourism, recreation and cultural values (Cesar et al, 2002). According to MEA (2005), the benefits of coral reefs for people are divided in different categories as represented in Table 3.1.

Table 3.1 Services provided by coral reefs

Services	Example
Food	Fish, seafood
Regulation and shoreline protection	Flood control, beach erosion
Cultural services	Spiritual, recreational and cultural benefits
Supporting services	Nutrient cycling that maintain condition for life on Earth

Source: adapted from Millennium Ecosystem Assessment (2005)

Every good or service provided by the ecosystems of Bonaire has an attached economic value. By summing up their values, the total economic value of coral reef ecosystems is obtained. The TEV represents the value of an ecosystem which brings benefits to people. To put an economic value on the goods and services provided by marine or terrestrial ecosystems it was necessary to conduct an extended study of the literature and projects done in past years to analyze Bonaire's benefits, along with a close communication with relevant stakeholders.

The value placed on an ecosystem service represents the level of preference of individuals on that good or service. The most common unit that expresses this value is "money" (van Beukering et al, 2007b). Even goods without a market price can be expressed in monetary terms by using monetary values such as willingness to pay (WTP), willingness to accept (WTA), market and non-market value, financial and economic value, costs and benefits, producer and consumer surplus, etc (van Beukering et al, 2007b)[5].

The TEV is calculated by summing the use and non-use value of coral reefs, which are defined by the type of their use (Figure 3.2). First, *direct use values* represent those goods and services that can be directly used by humans and have a market price. They can be consumptive (extractive) and non-consumptive (non-extractive). Extractive uses are represented by the goods which once consumed are not returned to the ecosystem, such as timber, fish for food and aquarium trade. Non-extractive uses are services provided by the ecosystem that are not extracted, for example recreation and education. Second, *indirect use values* are more difficult to value and are represented by diverse benefits provided by the ecosystem in an indirect way. Some examples are biological support to fisheries and turtles, physical protection of coast, carbon storage, etc. Third, *non-use values* illustrate the value place by people on different goods and services by taking into account any present or future use of them. Fourth, *bequest values* express the benefits that a good and service have for future generations, such as avoided damage due to climate change, while *existence values* represents the benefits of knowing that a good or service exists, for example, simply the existence of certain species gives happiness to some people. Fifth, a combination between use and non-use value result in a new sub-category, *the option value*. This value shows the significance a good or service have in the present for a potential future use. An example is the potential to derive a remedy for cancer from the substances found on reefs (van Beukering et al, 2007b).

[5] For a detailed and complete explanation for each of the different evaluation methods see the works of van Beukering *et al* (2007b)

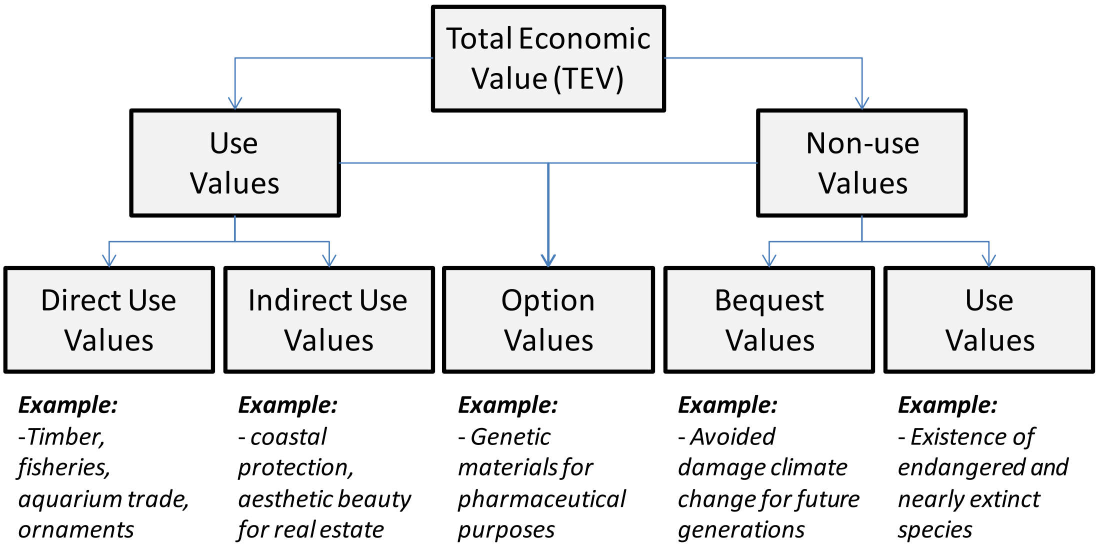

Figure 3.2 The TEV of an ecosystem

3.5 Economic valuation

To determine the TEV of nature both primary and secondary data are collected. As part of this study, these ecosystem services are measured through a range of valuation techniques (see Table 3.2). Market price represents the value at which a certain good or service is bought and sold in commercial markets (van Beukering et al, 2007b). Contingent valuation method estimate the value of an ecosystem service using surveys and asking people about their WTP for a specific service (van Beukering et al, 2007b). Hedonic pricing approximates the economic value of environmental services that affects market price, especially the housing price (van Beukering et al, 2007b). Houses at risk method estimates value of ecosystem services based on the costs of avoiding damages as a result of lost service (van Beukering et al, 2007b). The services analysed in this report are explained in detail in the next Chapter.

Table 3.2 Techniques used to valuate goods and services provided by coral reefs

Goods and Services	Valuation Technique
Tourism & recreation	Market price & Choice experiment
Non-use values	Contingent valuation method & Choice experiment
Fisheries	Market price & Choice experiment
Amenity	Hedonic pricing
Coastal protection	Avoided damage cost
Agriculture & livestock	Market pricing
Medicinal & pharmaceutical	Market pricing
Carbon sequestration	Market pricing
Research value	Net factor income approach
Art value	Net factor income approach

The TEV is calculated by summing up all the aforementioned values represented in Table 3.2. On a global scale the net benefits of coral reefs were determined to be around $30 billion per year (Cesar, Burke and Pet-Soede, 2003). The largest share of this is attributed to tourism and recreation with $10 billion, followed by coastal protection with $9 billion. For Guam, the TEV of coral reef was determined to be $127.3M per year, with 75% contributing the tourism sector (van Beukering et al, 2007a). The TEV of Hawaii coral reefs per year was determined as well by Cesar et al (2002). This was calculated to be $364M and the NPV equals $9.722M at 3%

discount rate for a time period of 50 years. Van Beukering et al (2011) calculated the TEV of USVI's coral reefs to be around $201M per year with 51% due to tourists.

3.6 Intervention methods

The impacts of anthropogenic threats can be avoided or mitigated through effective management intervention. In this study, several potential management options are analyzed and compared with a baseline scenario. An extended cost and benefits analysis (CBA) of the different scenarios and interventions provides policy makers with an 'objective' means of deciding which management options are best suited for the development of Bonaire. Below follows a series of different interventions that were incorporated in the scenario analyses:

1. *Construction of a sewage treatment plant* to reduce the amount of nutrients released into the coastal waters. Although this intervention requires large financial investments, a sewage treatment plant would contribute to an improved state of the marine environment by creating conditions more favourable for the proliferation of coral species. Note that the Bonairian government is currently constructing a sewage plant on the island (Kekem et al, 2006).
2. *Removal of free roaming livestock i.e. goats, sheep and donkeys*. This allows degraded forests on the island to recover to its original state of mature forests. Mature forests help decrease the total amount of sediment being washed into coastal waters. Sedimentation has negative impact on reef resilience.
3. *Active reforestation*: In addition to removing free roaming livestock, actively planting trees throughout the island would contribute to a faster regeneration of mature forests.
4. *Eradication of lionfish* which represent a direct threat to coral reef ecosystems. Lionfish are a non endemic fish species which have invaded the coastal waters throughout the entire Caribbean, including Bonaire. They predate on endemic fish, especially the juvenile population is affected. This exacerbates the all ready diminished fish stocks of Bonaire.
5. *Construction of artificial reefs (AR)* thereby increasing the amount of hard substrate within the coastal waters hereby providing a surface for sessile organisms to manifest themselves. At the same time, artificial reefs provide a safe haven for juvenile fish. The ARs thus create biodiversity hotspots attractive for divers and increase the local abundance and diversity of pelagic and benthic organism.
6. *Active coral recovery through the use of coral nurseries*. The active recovery of specific hard coral species by providing them an *in situ* nursery area. This is done by collecting broken off but still living segments of hard coral. These segments are then brought back to the nursery and cared for. Within the nursery the corals lack competition for resources (e.g. food and space) and thus have the opportunity to thrive. After a year or so these segments are placed back within the natural marine environment.

3.7 Boundaries and limitations

The purpose of this report is to build a dynamic simulation model for illustrating the ecological interactions of Bonaire's nature and to establish their total economic value (TEV). In order to do so, a literature study, surveys and expert interviews with relevant stakeholders were conducted to find out more about the island of Bonaire, its marine and terrestrial ecosystem, the goods and services provided by ecosystems and their economic value.

The area studied represents the total area of Bonaire Island. This study uses the analyses of ecosystems of Bonaire and their biggest threats, followed by the economic benefits gained due to the services they provide, and the calculation of the total value of both the marine ecosystem and terrestrial ecosystem combined. Future scenarios and the baseline will be analyzed for a time period of 30 years. This period is enough for the impacts on ecosystems to show their effects and from an economical point of view it is short enough to make reasonable predictions.

Regarding the data used, not everything is easily available making it difficult to have detailed and complete information necessary to operate the ecological-economic simulation model. Since reliable and long-term ecological data of the island is scarce, providing a well founded baseline has proven to be difficult. Moreover, due to different groups which analyzed these ecosystems over time, some of the data are available only for specific locations of the island and for specific moments in history. To overcome this problem it was necessary at times to extrapolate data.

In some cases, data was unavailable because accessibility was denied. This is often the case when a report is published by the private sector or by specific government departments. Governments are not always willing to share their data. As a result, proxy data from other studies was used. This data was acquired from studies with similar ecosystems on other islands, such as Hawaii, USVI or Guam.

4 The Model

In this chapter a detailed explanation of the steps taken to construct the ecological-economic simulation model will be provided. The model is split in 2 modules i.e. an ecological module and an economical module. The ecological module is separated in a marine system and terrestrial system. For both systems, specific parameters that best exemplify the state of an ecosystem and that were sufficiently available were chosen to be simulated. The main premise of the model is to simulate how the environment influences various ecosystem services on Bonaire. The output of the ecological model is an indicator, which in turn affects specific social-economical processes and the provision ecosystem services. Subsequently, the economical processes influence the state of the environment, this allows for a feedback mechanism between the two systems. The specific dose-response relationships and the underlying data are explained in the following sections.

4.1 Marine Environment Module

The coral reef community in the Caribbean has degraded over time as a result of human activities. In addition to natural stresses such as storms and hurricanes, anthropogenic factors such as overfishing, nutrient loading, sedimentation, deforestation, introduction of invasive species and climate change have had a substantial influence on the marine environment (Newman et al, 2006; Sandin et al, 2008). The model aims at capturing how these stressors impact the marine ecosystem and in turn, how these changes would influence the provision of ecosystem services. It is therefore necessary to use environmental data that 1) can tell something about the state of the environment and 2) inform how changes in specific environmental parameters influence provision of ecosystem services and goods. Keeping these two criteria in mind and having to work with limited data availability the following parameters were chosen as defining the state of the reef: coral cover, coral diversity, fish stock, fish diversity and algae cover.

Coral Cover and Coral Diversity

Out of 2,700 ha of coral reefs on Bonaire, the current coral cover is 28.6% (IUCN, 2011) comprising both soft (8.8%) and hard corals (19.8%). Coral cover depends on different factors that contribute to their increase or decrease. However, this expansion is limited by a maximum coral cover. Using the result from a report by IUCN (2011) in different locations of the island, the maximum cover found was 60%. We assume this value will not increase beyond this level. The present number of coral species is 65 according to IUCN (2011).

When no external factors are present the maximum expansion rate of coral cover, which also includes their resilience property, is 50% per year (Tanner, 1995; Bak et al 2009). Factors that contribute to the decline of coral cover are physical destruction, influenced by the number of stay-over tourists, the rate of sedimentation, amount of nutrients loaded into the water, a change in the temperature and the increase of algae cover which overgrow the corals and kill them. Factors that contribute to a decline in the number of coral species are considered to be the rate of sedimentation, the concentration of nutrients loaded into the sea and the amount of algae present.

Algae cover

Algae are an important part of the benthic community. The current algae cover is 42.4% of the benthic cover. It is formed mainly by turf algae which cover death corals (38.2%) and macro-algae (4.2%). Algae are an important regulator of coral cover. Their growth is sensitive to water temperature and is influenced by a decrease in coral cover, while their only major threat is considered to be herbivore fish. However, their growth is also limited by their carrying capacity calculated to be 82.2%, the maximum algae cover determined by IUCN (2011).

There is a direct competition between corals and algae for the same space (Vermeij et al, 2010; McClanahan-1995; Edwards et al, 2010). After years of research, evidence was gathered to prove that corals are affected by algae as a result of algal release of organic carbon that increases the local activity of microbes (Barott et al, 2011). Algae abundance threatens the existence of coral reefs due to their capacity to overgrow or block corals space, hampering their growth and expansion. According to Box & Mumby (2007), macro-algae and turf algae cause hypoxia on coral tissues, reduce coral fecundity and inhibit larval settlement.

Fish Stock and Fish Diversity

Fish biomass and fish diversity are important parameters for Bonaire's inhabitants as they represent a source of food and income. In the model a distinction is made between herbivore and predator fish species. A study conducted by IUCN (2011) in different locations in Bonaire discovered the average biomass for herbivore fish to be 7,319 g/100m^2, while for predators this was 5,290 g/100m^2. The total biomass of fish was calculated to be 3,404 tonnes, which is the sum of predator and herbivore species extrapolated over the entire BNMP.

The increase in fish stock is influenced by their reproduction rate. Myers et al (1999) calculated the maximum reproduction rate of different fish species to vary between 1 and 7. Because Bonaire has approximately 450 fish species, the maximum reproduction rate was considered to be 0.35. The maximum carrying capacity for the fish stock was calculated to be equal with 5,600 tonnes, by taking into account the maximum amount of fish found by IUCN (2011) in diverse locations of Bonaire.

An assumption was made for the maximum increase in fish diversity. As no data were found about this subject it was assumed that if no external factors are present the rate of expansion in the number of species is 0.04% per year. However, like the other indicators the fish stock is also threatened by diverse factors, such as overfishing and the presence of lionfish (Dew, 2001).

Herbivore fish are a threat to algae cover due to their diet formed by algae and sea grass. In one of their studies, Newman et al (2006) and Edwards et al (2010) demonstrated that there is a negative and linear relation between herbivorous fish biomass and algae biomass. Mumby et al (2006) revealed that parrotfish can graze a maximum of 30% of the seabed in 6 months, meaning a maximum of 60% of algae being grazed in one year. As the herbivore fish present in the water of Bonaire are not just parrotfish, the maximum algae decrease at the carrying capacity for herbivore fish was established to be 50%.

Nutrients

Groundwater represents the source of nutrients loaded into the sea. The major causes of nutrient enrichment in Bonaire are improper land use such as uncontrolled coastal development and the lack of a sewage treatment plant (Slijkerman et al, 2011). Untreated sewage consists of high amounts of nitrogen and phosphorus, important nutrients which contribute to sea water eutrophication[6], coral reefs degradation and a decrease in coral species (Kekem et al, 2006). The average concentration of inorganic nitrogen (NH4+NO3 + NO2) in Bonaire waters was measured by Slijkerman et al (2011) to have a value of 1.51±1.36µM. In their study about economic valuation of Hawaiian reefs, Cesar et al (2002) revealed the following equations for calculating the total decrease of coral cover and coral diversity due to the concentration of nutrients loaded.

$$(1) \qquad Coral\ Cover\ decrease\ due\ to\ Nutrients = \frac{15.5 * Nutrients\ (\mu M)}{30}$$

$$(2) \qquad Coral\ Diversity\ decrease\ due\ to\ Nutrients = \frac{10.8 * Nutrients\ (\mu M)}{30}$$

[6] Eutrophication represents the process by which a high concentration of nutrients is loaded in water and produce excessive growth of algae. Once the algae decompose they occupy the surface of the water depleting the oxygen and causing death to other organisms such as corals and fish (Art, 1993).

A variation in the amount of nutrients loaded into the water depends on the number of tourists. An increase in the number of tourists increases the concentration of N loaded. For the current state, it was considered that the amount of nutrients loaded is 1.51µM as it was determined by Slijkerman et al (2011). See Annex A for more information.

Sediments

High levels of sediments can bury the corals or damage them, making them vulnerable to diseases. Not only coral cover is threatened, but coral biodiversity as well. Bak et al (2005) mention that sediments are an important factor that contribute to coral mortality, considering that most coral species don't have the capacity to remove the sediments brought by hurricanes and other causes. Furthermore, there is a direct relationship between the amount of mature dry forest on the island and the amount of sediment runoff. As the amount of mature forests decrease on the island, so does there capacity to hinder erosion. This means that the sedimentation rate increases as the amount of dry forest cover decreases. A study done to represent the influence of sedimentation on coral cover and coral diversity in Hawaii, Cesar et al (2002) used the following relations to represent the effect of sedimentation on coral cover and diversity. See Annex B for more information.

$$(3) \qquad \ln Coral\ Cover\ (\%) = 3.17 - 0.013 * Sedimentation\ Rate\ (\frac{mg}{cm^2} * day)$$

$$(4) \qquad \ln Coral\ Species = 4.97 - 0.018 * Sedimentation\ Rate\ (\frac{mg}{cm^2} * day)$$

Physical destruction

Tourists and the activities they perform have a direct negative impact on corals. Through diving and snorkeling they are tempted to touch the corals, which decrease their resistance to diseases and make them more vulnerable to bleaching and possible death (WRI.com). According to De Meyer (1998), recreational activities such as diving and snorkeling, damage approximately 2.7% of coral reefs per year (see Annex C).

Fishing rate

Fishing rate contributes directly to the decline of reef fish stock. Overfishing can result in a tropic shift (Bruckner et al, 2010). Less grazing fish will allow algae grow unhindered. Without grazers, algae have a competitive advantage over coral, hereby reducing the coral cover. In contrast, a lower fishing rate will increase fish biomass and can also alter the ecosystem balance. For Bonaire, the total reef dependent fish catch represents 25% of the total catch (Schep et al, 2012a). See Annex D.

Lionfish

Lionfish are an invasive species which represent a big threat to the fish stock and fish diversity due to their appetite for small-bodied and juvenile reef fish. As a result of their ability to invade multiple habitats and reproduce very fast they are considered very devastating. Furthermore, due to their venomous spines they are protected against predators (Mumby et al, 2011). A study done by Vermeij (2012) revealed that at a depth of 15m the presence of lionfish in Bonaire is of 3 g/m^2. They are mainly present on the leeward side of the island. By multiplying this value with the approximately half of total surface of the BNMP, the current biomass of lionfish is equal to 40 tonnes. Cote & Maljkovic (2010) estimated that an adult lionfish (350g) consumes 8.5g fish per day. For Bonaire as a whole this results in 21% of fish stock decrease at the current biomass of lionfish. It is assumed that this decrease is linear with the increase in lionfish stock. According to Albins (2011) the maximum decrease in fish diversity due to lionfish is 1% per year.

Marine Indicator

An important relation in this model is the balance between corals, algae and uncovered substrate (e.g. rubble and sand). Together these three components comprise the surface area of the coastal waters of Bonaire, i.e. 2,700 ha (Holmes & Johnstone, 2010). A decline in coral cover gives the opportunity for algae to expand their surface. All the above mentioned parameters are collected and transformed into one indicator which reflects the general state of the coral reef ecosystem. To calculate the state of the coral reef ecosystem, a common indicator with values between 0 (fully degraded ecosystem) and 1 (pristine ecosystem) was built following the next three steps:

- First, for every indicator the rate of their presence was calculated by dividing their present amount (t or %) to their maximum possible amount.
- Second, according to their importance, each indicator was assigned with different scores, as follows: coral cover-0.3, coral biodiversity-0.3, fish stock-0.15, fish biodiversity-0.15, and algae cover-0.1. The scores were taken from Cesar et al (2002), and are based on expert opinions.
- Finally, in order to calculate the state of the reef, each score was multiplied with their corresponding existence rate and the results were summed up. The total sum was divided by the sum of the scores, which in this case is equal with 1, obtaining the value of coral reef quality. Once the state of the reef is defined, the next step is to calculate the total economic value of the coral reef which is in part influenced by the reef's quality.

4.2 Terrestrial Environment Module

Little is known about the tropical dry forest (TDF) ecology on Bonaire. Studies of abandoned lands have shown that the tropical dry forest regenerates fast, reaching a maximum of basal area of $25m^2$ per ha and the maximum of 25 species per plot between 30 and 40 years (Figure 4.1). Coppicing from stumps and roots remaining after disturbance is considered as the primary regeneration mechanism of disturbed tropical dry sites (Quesada et al 2009). Seed dispersal by wind is important for regeneration. The pioneer species within abandoned lands are wind dispersers plants like the Yellow Poui and Kapok, that function as nursery trees for other species accelerating the transition to primary forest (Aide et al 2000).

Plant-animal interactions in the tropical dry forest are extremely important for conserving the plants, animals and genetic diversity of the forest. It is estimated that 54–80% of the tropical dry forest plant species rely on animal vectors for its pollination, such as bats and the hummingbirds (Machado and Lopes, 2004). This interaction is crucial for the cacti Kadushi, Agave, Kalbas and Kapok which are important diets of the aforementioned animals. Seed dispersal by birds and bats is also important. Animal dispersed species in the dry forest are estimated between 43%-64% of the plant species within the forest (Quesada et al 2009). Changes in animal communities during succession undoubtedly affect seeds arriving to areas undergoing succession and ultimately the emerging succession of the forest, making seed dispersal important for plants such as the medicinal tree Wayaká and the Cactus Kadushi. Absence of such interactions may trigger a cascading effect, affecting plant density and reducing pollination (Traveset and Richardson 2006, Anderson et al 2011).

The ecological module aims to reflect the mentioned ecological characteristics of the tropical dry forest including a fast regeneration, early maturity, and facilitated regeneration by seed dispersal and pollination by bats and birds. For the terrestrial environment there are 3 ecological indicators taken into account, i.e. Animal occurrence, Plant diversity and Plant Cover. These are explained in the following.

Mature & Degraded Forests

According to De Freitas et al (2005) little remains of the original mature forest on Bonaire because the extremely degraded state and the unknown composition of the flora before colonial times. Different vegetation types are mentioned by De Freitas et al (2005) suggesting the existence of at least two different types of ecosystems not taken into account before: the dry evergreen formations and seasonal dry forest. Considering the mentioned generalizations about the regeneration of the dry forest an annual regeneration rate of 1% is assumed, which an approximation of the regeneration rates presented in Figure 4.1. The importance of plant-animal interactions is reflected in the model by including the parameter "facilitated regeneration". This value is estimated1% considering that not all pollinated flowers become a viable seeds and not all seeds survive to become trees. This additional regeneration is activated in the model when there is a high animal richness.

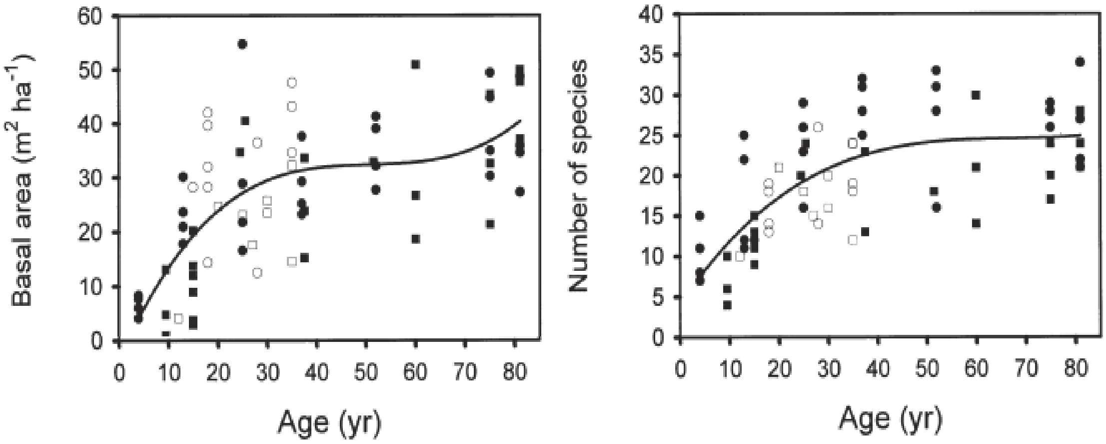

Figure 4.1 Recovery of the tropical dry forest with the age of abandonment

Source: Aide et al. 2000

Plant Richness

As the animal species, plant species composition resembles the flora of the Caribbean region. The flora consist of 387 vascular species including 36 introduced and naturalized species having so far discovered only one endemic plant species (De Freitas et al 2005). The amount of plants is affected by how much livestock is roaming freely. In the model there is a negative linear relationship between the amount of livestock and plant richness. Plant richness is decreases by half when the amount of free roaming livestock is at its peak.

Animal Richness

The parameter 'Animal Richness' reflects the response of the fauna to the mature forest cover. The general idea is that more mature forest extension correlates with more species of birds and bats. Again, it is difficult to make generalizations about how birds and bats abundance and richness are influenced by forest cover and human disturbances. This is mainly due to the fact that different species respond differently to these environmental changes (Lasky & Keit, 2010, see Chettri et al 2000 and Trzcinski et al 1999). For example, abundance of insectivorous birds is favoured by low forest cover but in contrast cavity-nester birds such as the Lora are very limited by availability of tree holes. This is essential for their reproduction but only available in mature forests as seen with other parrots (Cockle et al 2010, Lasky & Keit, 2010). A linear relationship is assumed with a maximum of 60 species.

Disturbance can be caused for example by habitat alteration when crowds of people approach to feeding and nesting places. There is evidence that tourist disturbances can increase nest abandonment and increase bird stress, reducing the successful reproduction, foraging and reducing the frequency of sight (Rochelle et al 2011, Fernandez 2000, Velando & Munilla 2011). The disturbance function is also difficult to determine; there is no doses-response research to tell us how much affected the animals are when different sizes of crowds of tourists approach. Furthermore, the impact of tourists varies in different systems depending on the resilience of the attraction after disturbance. For example, recovery from visitor impacts, might be relatively quick for tracks in some forest compared to the more sustained and permanent alterations of visitor impacts in low energy cave ecosystems (Cigna, 1993).

Animal Occurrence

A function was included that represents the amount of animals occurring on the island. This was done in order to account for the negative effect of tourism and other potential disturbances on the abundance of birds. Occurrence is an independent parameter that refers how often a bird is sighted regardless the species. Unfortunately the lack of ornithological surveys leave to raw assumptions the number of individuals per species of birds or bats found on a period of time. A linear relationship is assumed with a maximum of 80 species. If tourism numbers exceed 650,000 a minimum of 1 is reached. At the current amount of tourists, the amount of species is 80. In order to represent the threat of grazing on the plant richness a function is included that when grazing is bigger than Bonaire's carrying capacity the plant richness drops substantially.

Terrestrial Indicator

The sub-indicators 'plant richness', 'animal occurrence' and 'animal richness' jointly constitute the terrestrial indicator. The indicator summarizes the quality of the terrestrial environment of Bonaire and indirectly also affects the visitation of tourists to Bonaire. The indicator ranges from 0 (highly degraded) to 1(very good state). All three components of the "Terrestrial State Indicator" are normalized in order to sum all of them (i.e. for each of the functions the minimum is converted to 0 and the maximum to 1). To show how different environmental attributes influence the environmental quality and hence the "Terrestrial State Indicator" in different ways, a weight was applied to each normalized value.

4.3 Environmental economic module

Economic module consist of tourism, non-use, biodiversity, fisheries, amenity and coastal protection sub-modules. These main sectors of the economic sub-model are explained in the following.

Tourism and recreation

Tourists are the major contributors to the total recreational value of coral reefs, followed in a small part by locals. Tourism industry represents the main pillar of economic development of Bonaire (Groenenboom & Krul, 2009). There are three types of tourists: stay-over tourists (SOT) and cruise tourists (CT) and yacht tourists. All three types contribute to the recreational value of coral reefs. SOTs arrive by planes or yachts and spend a few days on the island, and CT arrive by cruise boat and spend just a few hours on the island. SOT tourists generate direct revenues by paying the fee for recreational activities such as diving and snorkeling, and indirect revenues through hotel and food costs. CT only produce direct revenues through expenses they have on diverse activities and souvenirs they buy. Moreover, locals contribute as well to the TEV of coral reef by providing direct revenues through recreational activities they undertake such as snorkeling and diving. However, tourism has not only positive sides. It represents also a threat though waste disposal and physical destruction of corals while diving. For more information on the international tourism value of nature on Bonaire, read Schep et al. (2012a). The local recreational values are estimated in Lacle et al. (2012).

In the last 10 years the number of tourists has increased considerably. In 2011, the number of SOT is 74,342 and CT are 229,228 (CTO, 2011). Through the expenses on the island they contribute to the economy of Bonaire. SOTs bring revenues to the island as a result of their direct and indirect expenditures[7]. SOTs have an average stay of 11.2 days and an average daily expenditure at $200. CTs only stay on the island for one day, and have an average daily expenditure of $140.

The principal factors that contribute to the growth rate of tourists are considered to be the growth of the global economy and the state of the reef. To represent the global economy, a growth in GDP of approximately 4% was adopted from the World Bank (2012), and the state of the reefs is determined in the ecological module. The growth rate of SOT, influenced by the change in GDP, is currently 0.8%, and based on past trends it is assumed it can increase or decrease with a maximum of 2% when the growth in GDP varies between -1 and 6%. The present growth rate of CT is double than of SOT (i.e. 1.6%) and is assumed to increase or decrease with a maximum of 4% when the growth rate of GDP varies between -1 and 6%.

The same trend is used to express the influence of the state of the reef on the growth rate of SOT and CT. These numbers are based on the tourist survey described in Schep et al. (2012b).The state of the reef ranges between 0 and 1. At its minimum value of 0, SOT will experience a drastic decline by 80% per year, while at its maximum growth rates will increase by 1%. A similar trend is observed for CT. Cruise tourists will experience a decline of 70% at the minimum qualitative state of the reef and an increase of 2% at its peak. The decline or increase of the tourist growth rates reflects their willingness to visit the island in relation to the state of the reef. The premise being that a totally degraded reef will no longer attract any tourists. There is however a minimum amount of tourists that visits per year, i.e. 1% of the initial starting values at the year 2012. The ST will never drop below the amount 7,500 and CT 25,000.

Besides growth in GDP and the state of the reef, tourist numbers may also be negatively affected by crowding. The tourist survey revealed that a high number of CT negatively affects the average stay of SOT, as the last group will not enjoy the beauty of the island due to a high number of cruise ships and visitors (Schep et al, 2012b). The survey revealed that an increase of 1% in the number of CT on the island leads to a decline of 0.25% the average days spent by SOT on the island. Moreover, the likelihood that SOT will revisit Bonaire in the near future is also substantially reduced with negative experience of over-crowding.

Besides tourists, Bonaire inhabitants contribute to the total recreational value of coral reefs. Their contribution is not so much compared with the tourism industry, but it is still an important factor when calculating the total recreational value of the coral reefs. A growth rate of 1.6% per year was considered in the number of Bonaire inhabitants, because local government presented their plans to increase their number from 15,666 to 25,000 in the coming 25 years. To calculate their total expenses on activities attached to corals it was considered that inhabitants pay a fee of $25 per year for scuba diving or other recreational activities that contribute to the TEV of coral reefs.

Non-use value

Non–values are included in this section by determining the WTP of locals and Dutch population for improving or conserving the Bonaire's nature. Two different methods were used to determine the Non-use value of Bonaire. First, the contingent valuation technique, in which 803 individuals took part. Second, a choice experiment was conducted, in which 512 individuals took part. For information on the non-use value of nature in the Caribbean Netherlands, please check Van Beukering et al. (2012).

Commercial and recreational fisheries

[7] Direct expenditures are due to the fees they pay to enjoy recreational activities directly linked with corals, such as diving or snorkelling. The entire bulk of the direct expenditures can be attributed to the coral reefs. Indirect expenditures, represented by food, kite surfing, boating or hotel payments, also contribute to the value of coral reefs. For both SOT and CT 70.5% of the indirect expenditures can be ascribed to the coral reefs. 85.75% of all expenditures of the ST are Indirect, for the CT this is 68.5%.

Fishing activities are defined by their purpose: commercial fishing, subsistence fishing, aquarium fishing, and recreational fishing. For the Bonaire case study, only commercial and recreational fishing are incorporated in the analysis because there is hardly any information about subsistence and aquarium fishing. To calculate the fishery related values of coral reefs of Bonaire the following steps were followed. First, the total revenues from fish sales are calculated. The fishing rate on Bonaire was established to be 1.8% of the total stock (Schep et al, 2012a). This number represents the fishing rate on reef related fish. Fish are sold at an average price of 6.95 USD per kg. Although only about 70% of the fish caught are sold, all fish caught are ascribed to the fisheries value. This is because the remaining 30% of the fish caught are consumed by the fishermen and therefore also add to the welfare of Bonaire citizens. Second, the cost for fishing was calculated by multiplying the average cost for boat maintenance with the total number of boats (18 small boats and 12 larger boats). Third, the profit of fishermen was calculated by extracting the costs from the total revenues. Finally, the entire value is ascribed to the corals as all fishing takes place within the reef habitat.

Amenity value

The presence of houses on the coast near coral reefs is a factor that contributes to the value of the house. A beautiful view with healthy coral reefs increases the price of the house with a certain percentage. On the opposite, damaged corals will decrease house price, as the amount of algae is going to increase and with their disintegration will trigger a bad smelling and unpleasant view. As a result, the value added to the average house price as a result of coral reefs existence represents the value placed on their existence and it is called the amenity value (Bervoets et al, 2010). For more information on this amenity ecosystem services of nature for the real estate industry of Bonaire, read Van Beukering and Wolfs (2012).

The amenity value is calculated on the basis of the number of houses sold and the nature-related value of houses, which is partly are influenced by the state of the marine and terrestrial environment. The average price of a house and the average number of houses sold influenced by the world GDP, were defined by analyzing their values in the last 8 years and establishing the following relations between them. The influence of reefs' quality on the price of a house was defined by Cesar et al (2002) who mention that a presence of coral cover between 10 to 50% change the house price with 1.3% following a S shape. As mentioned in Cesar et al (2002), from the total value of houses sold, around 1.5% is considered to be the amenity value of coral reefs.

Coastal protection

Coral reefs provide coastal protection and as a result they have the capacity to dissipate wave energy and protect the shoreline against storms, hurricanes and erosion. Healthy coral reefs prevent damage to the infrastructure and houses developed on coast during extreme events. Due to their structure, corals act as wave breakers mitigating the impact of waves and protecting the properties placed on the coast. Coral reefs protect the shoreline within 2,000m and represent 29% of the Caribbean coastline (van Beukering et al, 2007b).

To value this function it is necessary to know how the absence of coral reefs will influence the value of houses and infrastructure and the necessary costs to prevent shoreline damage, by constructing wave breakers or providing coastal nourishment. In this simulation model the coastal protection value of coral reefs is calculated through assessing the damage cost avoided to the properties close to the coast in case of no coral existence. For more information on the coastal protection value of coral reefs of Bonaire, check Van Zanten et al. (2012).

Livestock and Agricultural Value

The terrestrial value of nature on Bonaire is partly determined by the benefits that the agricultural sector gains from the presence of abundant vegetation. The structure of the livestock and agricultural module is pretty straightforward. The annual value of livestock is the sum of the amount of sheep and goat meat sold. Each goat slaughtered provides an average of 5.6 Kg per animal. The annual number of slaughtered animals in Bonaire is estimated to be 4,974. This number reflects the legal and illegal slaughters. The price per kg of meat is maintained constant throughout the 30 years of simulation.

Agriculture on Bonaire takes place on the Kunukus. At present, there are around 300 Kunukus remaining on the island, growing bananas, corn, fruits such as the papaya. Production supplies in a limited way local consumption, but still relies on imports. Agriculture also has a low production due to low economic input, unsuccessful water management, low efficiency of soil, erosion and loss of nutrients. The only evidence of intensive agriculture on Bonaire was Aloe around 50 years ago, today the crops are not intensively managed and some of these are abandoned. (Van Beuzowen et al 2009). There is scarce information available for the agrarian production on Bonaire. Right now the estimates of an expert (i.e. Jan Jaap van Almenkerk) are that the local production of household vegetables and fruit is circa 10,000 USD per year for around 7,000 ha. (Esther Wolfs pers. Com. 2012). In addition, goats, sheep, pigs and cows are raised on Bonaire. Goats are especially part of Bonaire's culture and are the main component of many dishes. For more information on the agricultural and cultural value of Bonaire's nature, verify Lacle et al. (2012). Relevant agricultural information can also be found in Van Beukering and Wolfs (2012).

Medicinal and pharmaceutical values

The quality of Bonaire coral reefs is known as being the highest in the Caribbean. Until now, there are about 65 species of corals and 450 species of fish known in the waters of Bonaire, which generate economic benefits (IUCN, 2011). These benefits of high levels of marine and terrestrial biodiversity are accounted for in the model in the context of research, medicinal and pharmaceutical uses. This total biodiversity value is formed by the sum of the research values, bio-prospecting values and the terrestrial medicinal plants values.

Nature in Bonaire provides important services for research and education. The marine and terrestrial environment of Bonaire is subject for a large group of academics conducting and publishing innovative research based on these unique and easily accessible ecosystems. Without the presence of healthy ecosystems, Bonaire would not attract large numbers of researchers nor would Bonaire's nature be a source of inspiration for many educational activities on the island and beyond. This study made an inventory of all ecosystem related research expenditures funded by governmental and non-governmental organizations for Bonaire. For more information on this research value of nature of Bonaire, read Van Beukering and Wolfs (2012).

Medicinal plants play important roles in many traditional societies. The healing properties of herbal medicines have been recognized in cultures thousands of years ago. A large part of the population in Bonaire is found to regularly collect and use local herbs and other medicinal plants for medical treatment. Two-third of the inhabitants who were surveyed made use of local plants as an alternative to modern medicine or prescription drugs. For more information on this medicinal and pharmaceutical ecosystem services of nature of Bonaire, read Van Beukering and Wolfs (2012).

Besides these local benefits, biodiversity is important for the development of pharmaceutical treatments and drugs. Bio-prospecting value refers to the revenues that pharmaceutical companies can obtain as a result of discovering important drugs that can be obtained using molecules from corals. It is calculated by multiplying the probability of discovery with coral diversity, total surface, probability of discovery and the value placed per species (Brock et al, 2011).

Carbon sequestration

The ecosystem service of climate regulation of Bonaire deals with greenhouse gas emissions and how ecosystems can mitigate such effects. Bonaire has six ecosystems that provide carbon-sequestering properties: salinas, dry forest, coral reefs, sea-grass, mangroves and open-ocean. In this study, we aim to value the climate regulation potential of Bonaire. This desk study has made a rough attempt to estimate the carbon sequestration value of the main ecosystems of Bonaire on the basis of actual carbon market prices. For more information on carbon sequestering ecosystem services of nature of Bonaire, read Van Beukering and Wolfs (2012).

Art value

Artists are inspired by their surroundings. Such is also the case on Bonaire, where the natural scenery of the island stimulates artists to use components of nature in their work. Clearly, nature plays a crucial role in the production process of art on Bonaire. The demand for art consists of the thousands and thousands of tourists visiting the island, who are keen to bring home a piece of art to remember the beauty of the island upon their return. Moreover, the beautiful photographs and books produced on Bonaire are distributed to clients across the world. Given the explicit demand and supply of art on Bonaire and its strong dependence on nature, the art sector on Bonaire plays an important role in the overall economy and provides an additional reason to manage nature well on the island. For more information on this art value of nature of Bonaire, read Van Beukering and Wolfs (2012).

5 Results

In this chapter the baseline scenario and three hypothetical management interventions are analyzed. The three management scenarios will focus on the combined ecological and environmental effects of different interventions. As described in Section 3.5, these three management interventions encompass the following:

- Scenario 1: Restoration which involves restoration of corals in a coral nursery, the construction of artificial reefs, active reforestation;
- Scenario 2: Eradicating invasive which involves the continuous effort to remove lionfish in the marine environment and the control of a large proportion of free-roaming goats, sheep, and donkeys.
- Scenario 3: Sewage improvement which simulates the effects of the sewage treatment plant.

The purpose of this analysis is twofold. First, the aim is to determine how the ecological state of the marine and terrestrial environment is possibly going to be affected over the 30 years by the proposed interventions. Second, since the ecological and economical systems are interlinked, the model will simulate how in turn the different various ecosystem services are influenced by the interventions. By doing so, it is possible to calculate the TEV for each year during the 30 year simulation. Moreover, in order to establish a more accurate indicator of what the benefits for society are, a sensitivity analysis is conducted using different discount rates. Before the three management interventions are analysed, the baseline developments are simulated in the following Section.

5.1 Baseline scenario

The baseline scenario assumes that no changes are going to happen in the selected time framework and that the current indicators and interactions between them will remain constant.

5.1.1 Ecological model

The different ecosystems were split into different modules and each has their own ecological state and influence different economical parameters within the model (i.e. Marine and Terrestrial Indicator). The state of the reef is defined by the five ecological factors: fish stock, fish diversity, coral cover, coral diversity and algae cover. The main threats identified for the marine ecosystem are the presence of lionfish, fishing rate, amount of nutrients released into the water, accumulation of sediments and physical destruction. The terrestrial state is defined by: the amount of mature forest present and diversity of flora and fauna present.

If the present state remains unchanged over the next 30 years the fish stock is expected to decrease. As presented in Figure 5.1, the first 10 years is characterized by decline in fish stock. By 2042 to total amount of fish will have declined from 3400 tons to 1427 tons. The major contributor to this decline is the presence of lionfish. Lionfish consume juvenile fish (Albins & Hixon, 2011) which prevents the local fish populations to replenish and maintain a steady stock. Furthermore, in the absence of a fish population that has the capacity to propagate, the effects of fishing are all the more visible. In a similar fashion the lionfish will affect the amount of fish species. It is expected that by 2042, there will be 387 species left of the initial 450 (Figure 5.1).

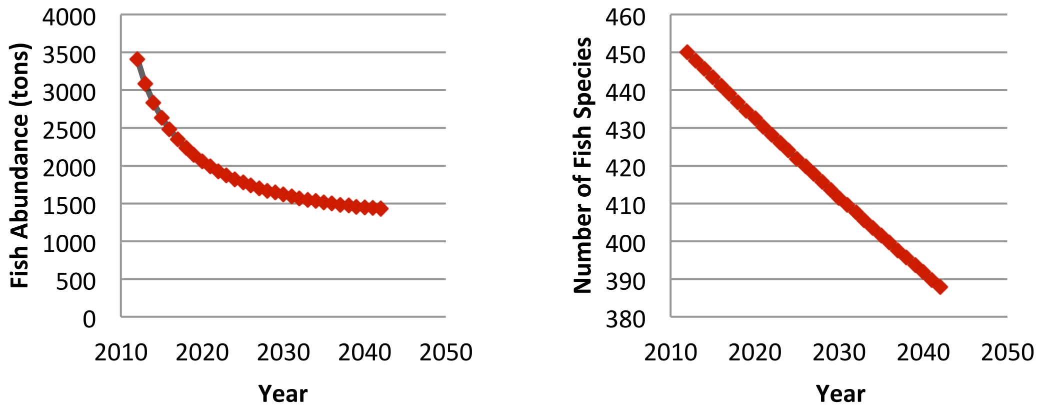

Figure 5.1 Fish stock (t) and fish diversity variation in time for the baseline scenario

As of 2012, the benthic cover is dominated by algae with 42.6% coverage, followed by corals, 28.6% (IUCN, 2011). If all values and their interaction will suffer no change in the following 30 years, the benthic cover will experience a drastic decline, as shown in Figure 5.2. The algae cover will constantly expand and remain constant at around 62%. After 10 years, all of the coral present will have declined to a minimum of 5%. This minimum represents the cover of corals in the deep reefs (20-40m). At these depths corals are less influenced by the harmful disturbances such as pollutants or physical destruction. By the year 2023, 5% of benthic cover is occupied by corals. The reason for such a rapid decline is in part due to a decline of herbivorous fish. Such fish are important in keeping algae populations from over growing fringing corals. Also ambient nutrient concentrations in the water provide excellent conditions for algae to flourish. This means that algae have a competitive advantage when competing for space with corals. The algae have no grazers preventing them from overgrowing the algae, and the high nutrient concentrations in the water elevate growth rates. The result, coastal waters that are almost completely devoid of coral cover.

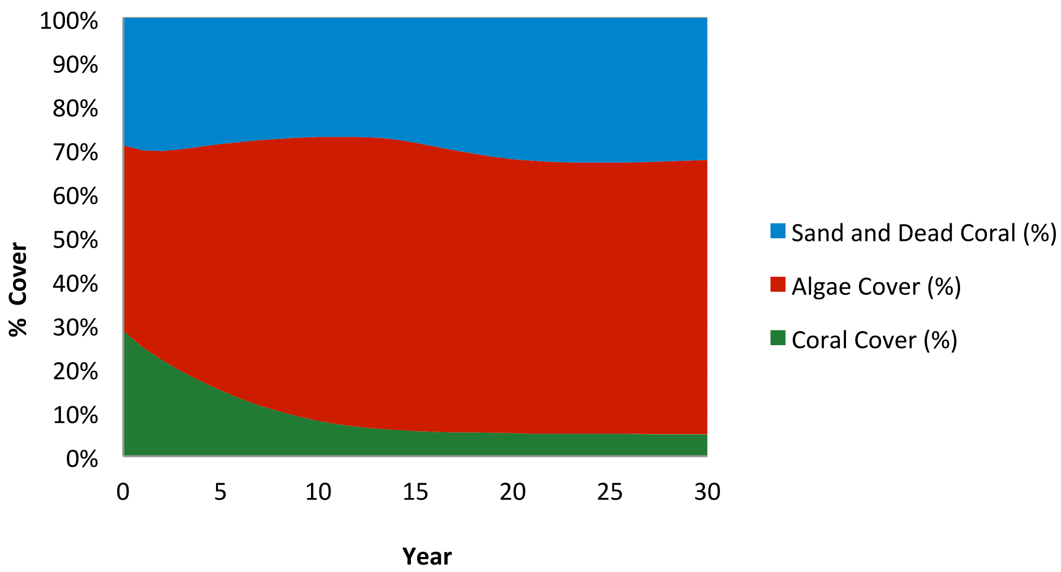

Figure 5.2 Variation in benthic cover for the baseline scenario

All the marine-related ecological indicators define the general state of the reef. As shown in the left side of Figure 5.3, the initial value of the marine indicator was 0.66, which place the corals at a higher than medium quality. The effect of the current threats on the state of the reef will cause a rapid decline in the first 10 years, as it slowly levels out to a value of 0.29 after 30 years. With a value of 0.29 the qualitative state of the reef is low. This will in turn influence different economical systems accordingly.

The initial state of the Terrestrial Indicator is 0.22 and will eventually level out at 0.26 (see right side of Figure 5.3). Although there is a small increase in the qualitative state of the forest 0.26 remains a low value. After 2020, the reason for the increase is that due to an annual decline in tourist visitation. This in turn affects the physical disturbances on the reef and the forest, meaning that the animal populations in the forest have the opportunity to spread without the continual disturbances of tourists. It is for this reason that the quality of the forest slowly starts to increase again after 10 years. Although animal populations have the possibility to recover to a certain degree, mature forests are no longer present on the island. This is caused by the presence of 30,000 goats that roam free on the island.

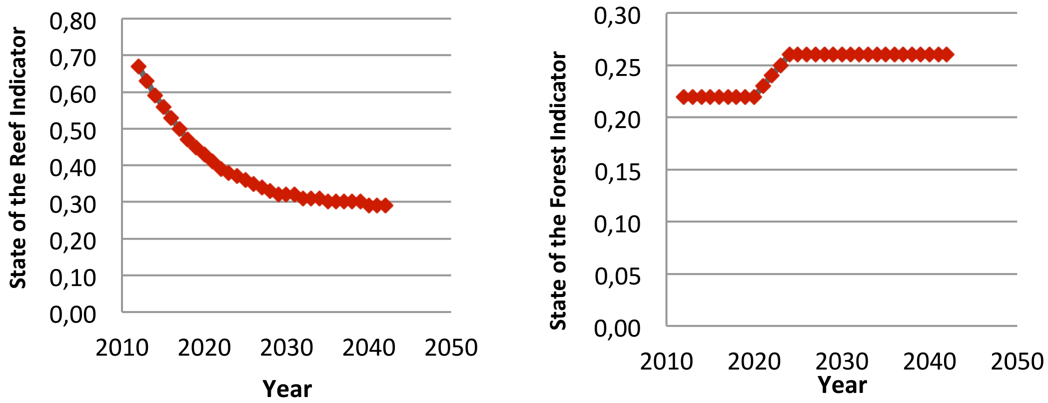

Figure 5.3 Marine indicator (left) and the terrestrial indicator (right) for baseline scenario

5.1.2 Economic module

The total expenses of tourists and locals on activities related to corals and forest contribute to the definition of the tourism and recreational value. The state of the reef, the state of the forest and the change in world GDP determine a growth in tourism and the average amount of money that tourists spend once they arrive on the island. As seen in Figure 5.4 cruise tourists show an increase in the first 10 years reaching a maximum of 268,000. After 10 years the amount of annual visiting cruise tourists shows a drastic decline. Stay over tourists decline from the onset. Eventually the degrading qualitative state of the forest and the reef exacts its toll. Tourists are not as likely to visit or return to Bonaire if the nature it has to offer is in a degraded state. Such declines in the tourist populations entering Bonaire has negative influence on the total revenue produced by the tourism sector.

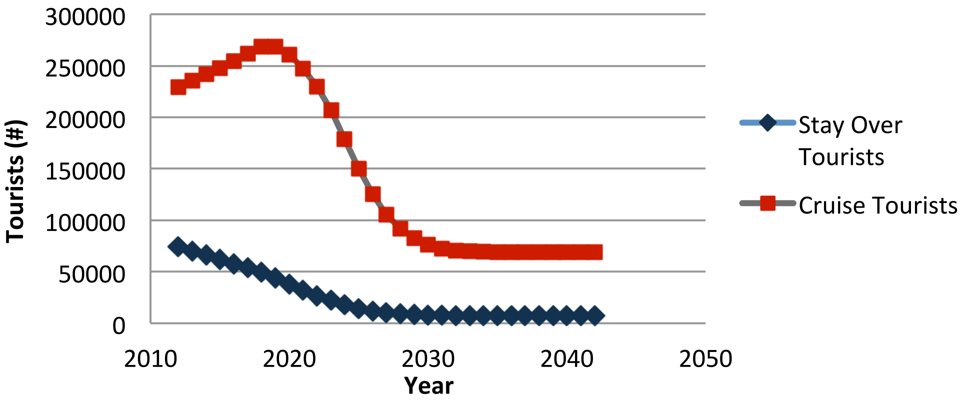

Figure 5.4 Change in the number of tourists for the baseline scenario

The biodiversity value is determined by summing the research, medicinal and bio-prospecting value. Figure 5.5 illustrates the variation of these values over the studied time period. There are different factors that contribute to the variation in the biodiversity value. First, due to a decrease in coral cover and mature forest, the chances of finding medicinal plants declines. However, since the population size of Bonaire is expected to increase so will the number of people benefiting from the biodiversity services. Therefore, the medicinal value of the forests increases as the population increases. This value is calculated by the amount of money is saved by not visiting a doctor and instead using medicinal plants found in the forest.

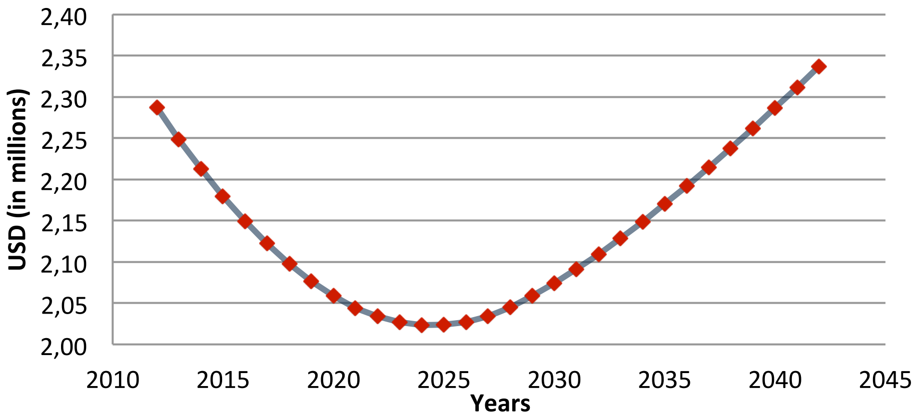

Figure 5.5 Change of the Biodiversity Value for the baseline scenario

The existence of a high diversity of flora and fauna in both marine and terrestrial ecosystems also generates economic benefits by means of non use values. The aesthetic values of the ecosystems are part of non use benefits people obtain through spiritual enrichment, cognitive development, reflection and aesthetic experiences. Many people find beauty or aesthetic value in various aspects of ecosystems, as reflected in the support for parks, scenic drives, and the selection of housing locations and enjoyment of scenery. The WTP of Dutch households to preserve nature declines as the state of the reef and forest declines. In a report by van Beukering *et al* (2012) the willingness to pay for the preservation of Bonaire's nature is presented. During the next 30 years as a result of coral reef and forest degradation it is assumed that the WTP of Dutch citizens to improve Bonaires nature quality will decrease. After 30 years the WTP of Dutch households will have decreased from $61 million to $26 million per year (Figure 5.6). The WTP for locals decreases as this value also depends on the state of the reef and forest.

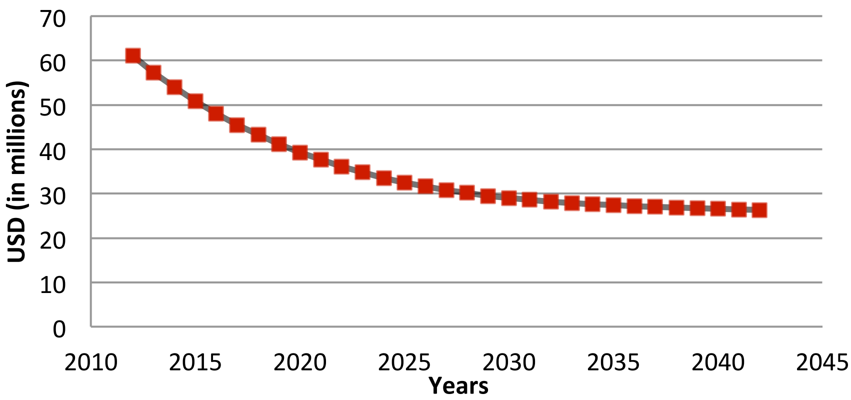

Figure 5.6 Change of the Non-use Value for the baseline scenario

To determine the fisheries value it is necessary to know the amount of fish caught for sale. The total amount of fish catch in the commercial industry varies between 26 tons to 64 tons. The negative influence in the Baseline Scenario is the presence of Lion fish, which decrease the capacity of fish populations to replenish their stocks. The total fisheries value thus decreases accordingly. The initial value of $0.93 million eventually declines to a value of $0.28 million (Figure 5.7). After 25 years the fish caught in the commercial fisheries will be too low for commercial catch (i.e. costs exceed the benefits). The only reason why the fisheries value as a whole does not become negative is because the recreational fisheries still has a large value. The initial value of recreational fisheries is $0.67 million and declines to $0.28 million after 30 years.

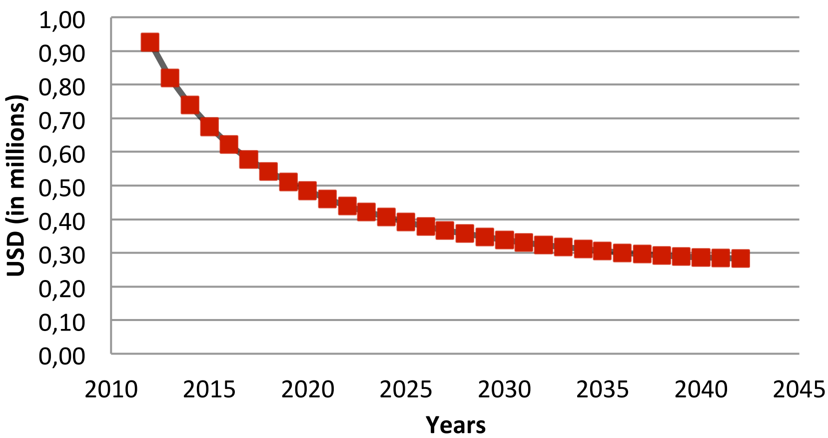

Figure 5.7 Variation of the Fisheries value for the baseline scenario

The value of the agriculture and livestock is not influenced by the state of either the forest or the reef. It is simply a question how much of the land cover on the island is allocated to agricultural use or to the development of mature forest. It is for this reason that the total value remains constant at $0.3 million.

According to Cesar et al, (2002), it is the state of the reef that influences the variation in the average house price by ± 1.3%. Beside this, the global GDP is the major contributor that defines the price of a house and thus the amenity value. Currently, the average price of a house was calculated to be $0.15 million, and will decrease slightly to a value of $0.14 million. The decline can be ascribed to the fact that the value of real estate close to the beaches will drop as these locations become less favourable due to a decline in coral cover and overall aesthetics.

The ability of Bonaire's coral reef to protect the coast is monetized by calculating the avoided damage of properties close to the coast in case of a hurricane in the absence of coral reef. The value is calculated by assessing the damage costs avoided during the event of a hurricane by the presence of corals. With a once in every 30 years hurricane frequency and a damage of 16.6% to the total house value, the coastal protection value of coral reefs will decline from $2.93 million to $2.85 million in 30 years. The reason for this decrease is the decline in the general state of the reef. Since the overall capacity for corals in Bonaire to mitigate damage on coastal infrastructure and housing small, the coastal protection value declines very little. Furthermore, it is also the dead coral that buffer incoming waves and thus help mitigate damage to coastal infrastructure.

5.1.3 Total economic value

In order to define the TEV of Bonaire's coral reef, the final step is to sum all the benefits which were determined in the previous steps. Figure 5.8 show the decline of the TEV, from 105 million USD to 37 million USD in 30 years. The TEV eventually levels out at the 37 million marker, as this is where the reef and forest state reach their collective minima. Figure 5.9 shows the composition of the TEV of nature of Bonaire. The non-use value contributes the most (63%) followed by the tourism and recreational value (265).

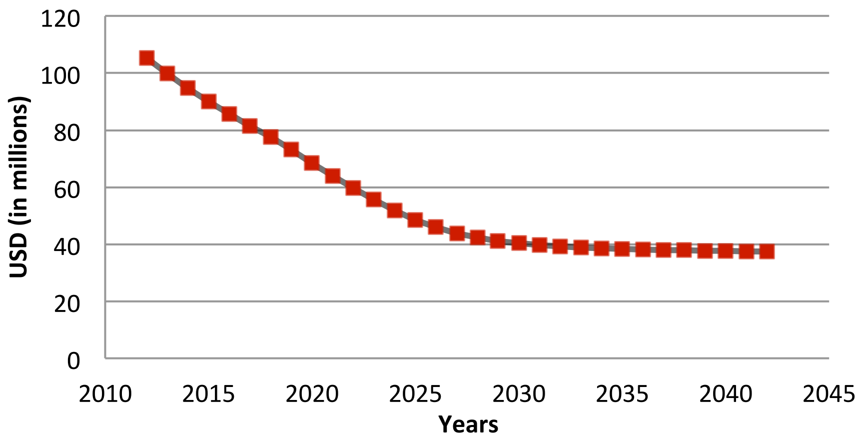

Figure 5.8 The Total Economic Value of the Bonaire's ecosystem services

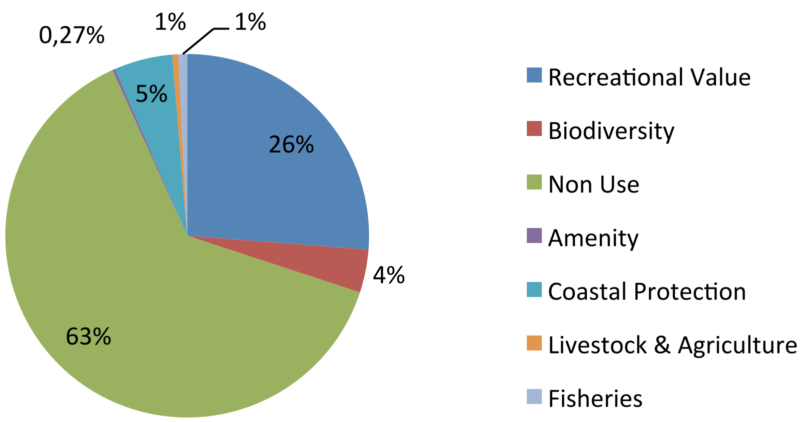

Figure 5.9 The share of each value contributing to the TEV in the baseline scenario

All the aforementioned values are presented without taking a discount rate into account. Figure 5.10 depicts the net present values of Bonaire's nature at different discount rates, ranging from 0% to 15%. The NPV varies between $1.7 billion and $626 million over a 30-year period.

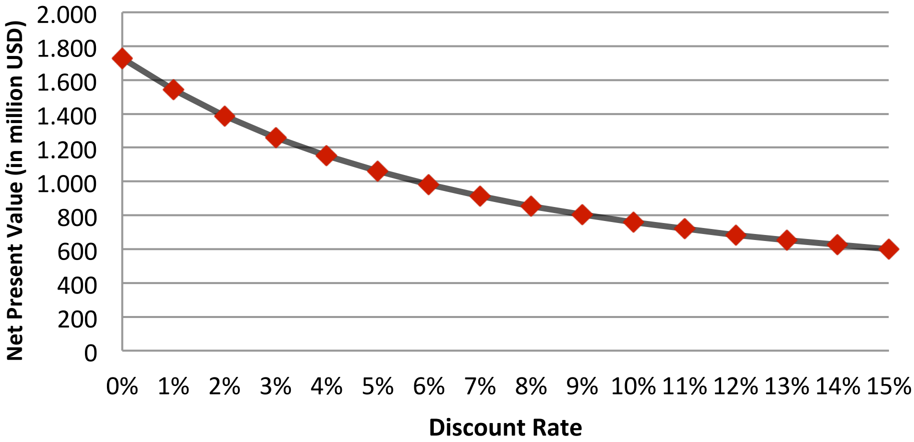

Figure 5.10 The Net Present Value at different discount rates.

5.2 Scenario 1 – Restoration

This Restoration Scenario analysis the effects of various ecological restoration options on the general quality of Bonaire's forests and coral reefs, and thus on the subsequent TEV. These include the following management interventions:

1) *The change of land use from agriculture to mature forest and active reforestation:* An increase in the total cover of mature forests will decrease the total amount of sediment entering coastal waters. In turbid waters, corals are put under a lot of pressure. In part because the sediments stick to the corals and their capacity to rinse themselves is not fast enough. Also because turbid waters decrease the total amount of light penetrating the water. Thus the photosynthetic capacity of the zooxanthellae symbionts decreases.
2) *The construction of artificial reefs:* The construction of artificial reefs will provide extra hard substrate within coastal waters. This hard substrate allows sessile organisms a surface on which they can attach themselves. Such resources of space are rare in coastal waters, especially in coastal waters that are dominated by macro and turf algae. Furthermore the artificial reefs provide a safe haven for juvenile fish, protecting them from predation. This helps increase the rate at which fish stocks replenishes and creates extra habitat for fish thus increasing the carrying capacity.
3) *The use of a coral nursery*: The coral nursery will help the rate at which corals recover. The aim of this activity is to nurse broken off pieces of live coral to a healthy state and a reasonable size ($200cm^2$) and "planting" them back in the wild. The nursing of the corals is all done *in situ*.

All these different management options aim to increase the state of the forest and reef. Eventually a healthier forest and reef can contribute to a higher TEV of Bonaire's nature. At the end of this Section the total costs and benefits are presented, hereby giving an overview if the interventions are worth the investment.

5.2.1 Ecological model

Figure 5.11 depicts the benthic cover of Bonairean coastal waters in Scenario 1 - Restoration. The first thing that becomes obvious from this figure is that it is very similar as seen in Figure 5.2. There is little difference in benthic cover when comparing the restoration scenario with the baseline scenario, despite active restoration efforts. A lack in contrast can be seen for almost all the other parameters and values between the two scenarios.

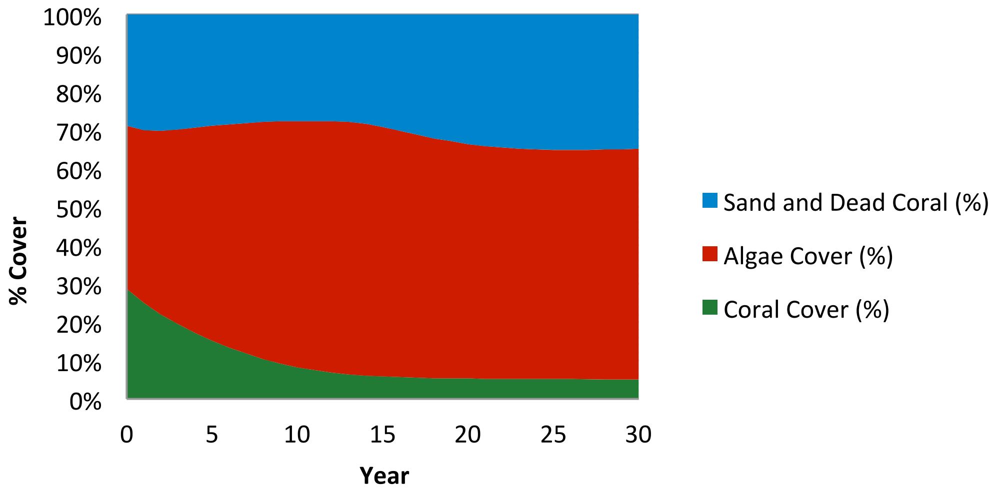

Figure 5.11 Variation in Benthic cover for the restoration scenario

After 30 years the fish abundance for the interventions resulted in a fish stock of 1,571 tonnes, this is 150 tons more than in the baseline scenario. The artificial reefs provide some extra habitat, and a place where fish can avoid predation. Despite these efforts however fish abundance declines because lionfish still predominate throughout coastal waters. This results in a decline of fish abundance. The same can be said about fish biodiversity.

The most distinct difference between the two scenarios is in the level of forest cover (see Figure 5.12). Active reforestation is causing the mature forests to decline at a slower rate when compared to the baseline scenario. Despite the intervention, after 20 years, mature forest cover will be completely absent on the island. Free roaming livestock are the main reason for this decline.

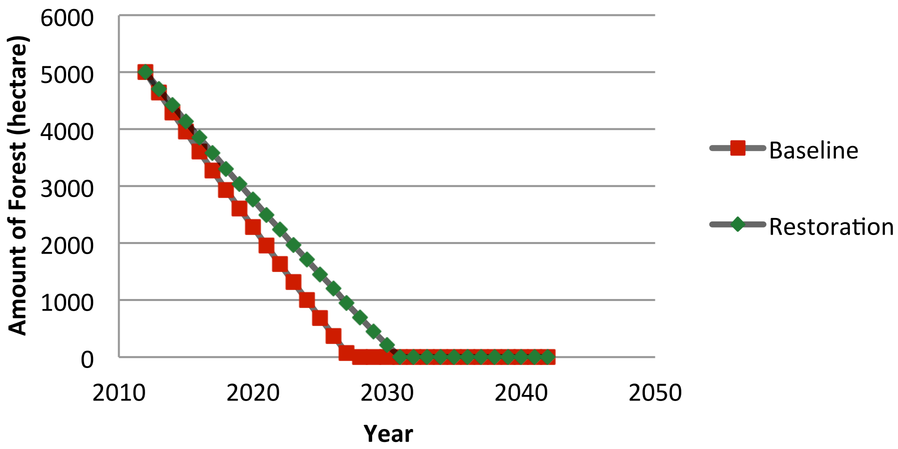

Figure 5.12 Comparison in forest cover for the baseline and restoration scenario

As shown in Figure 5.13, the interventions that were proposed in the restoration scenario have little effect. The indicator follows the same trend as in the baseline scenario. The marine indicator follows the same trend as in the baseline scenario. The interventions had a marginal influence on the qualitative state of the reef. Similarly, the terrestrial indicator follows the same trend, except that its value is slightly higher. This is because from the onset, in the model it was presumed that more land use was allocated to mature forest cover. In total 4,000 ha that was previously used for agriculture is now presumed to be used for natural forest growth. This means that the value of the forest indicator has a higher value, but follows the same trend as in the baseline scenario. Despite restoration efforts for both environments, the indicators follow the same trend as in the baseline scenario It seems that there are greater forces at work. Despite the presumed increased survival rate of fish, and increased coral growth other factors such as the presence of Lionfish and free roaming goats predominated. As a result, the interventions had little effect.

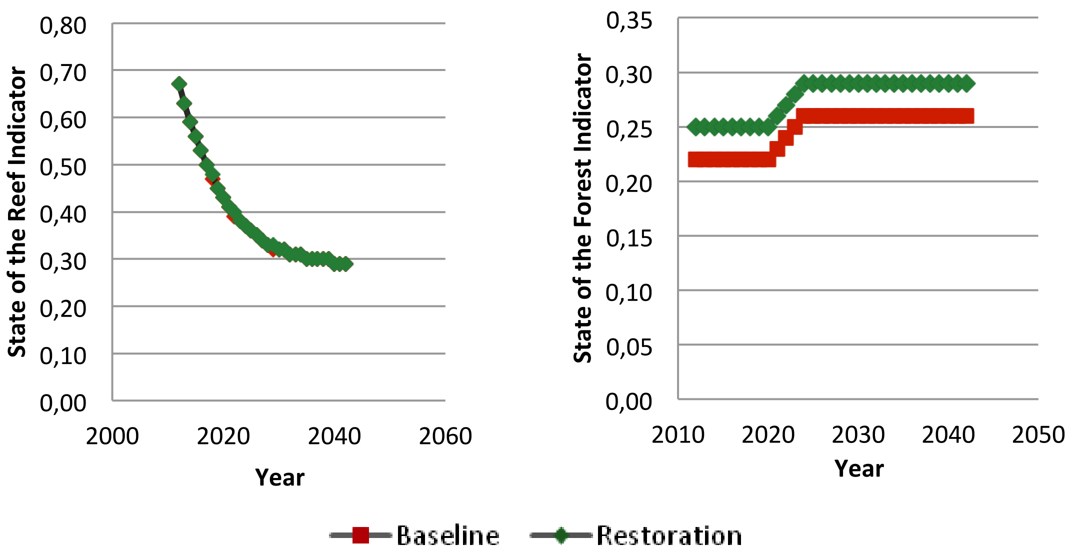

Figure 5.13 Comparison in the marine indicator (left) and the terrestrial indicator (right) for the baseline and restoration scenario

5.2.2 Economic model

Because of the limited impact of the restoration measures on the health of ecosystems of Bonaire, little economic impacts are recorded in terms of substantial increases in value of ecosystem services. The cruise and stay over tourists depict a similar trend as seen in the baseline scenario. Initially the annual visiting cruise tourists increase to a maximum of 270,000. After 10 years a rapid decline occurs. The amount of cruise tourists that visit annually drops and levels out at a value of 70,000. Stay over tourists on the other hand start to decline from the onset. Starting at 75,000 and levelling out at 7,500. These declines can be ascribed to the degrading state of both the terrestrial and marine environment. As the tourist numbers decline, so does the recreational value. The value of Bonaire's tourism industry starts off at $38 million and levels out at $5.2 million (see Figure 5.14).

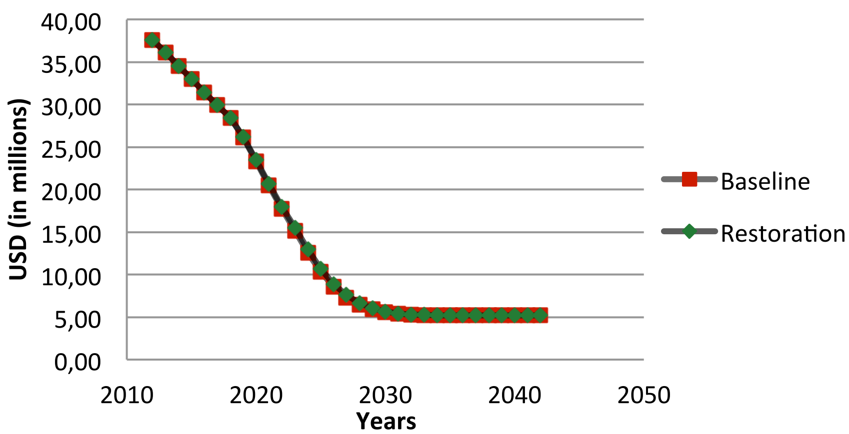

Figure 5.14 Variation in tourism and recreational Value for the restoration and the baseline scenario

The sum of all the different values in this scenario reflects the same trend as seen in the baseline scenario. It starts off at 105 million USD and drops to a value of $38 million (Figure 5.15). Despite investments in the restoration, the net benefits are barely visible. This is because the restoration efforts were inept at reversing the declining quality of the environment.

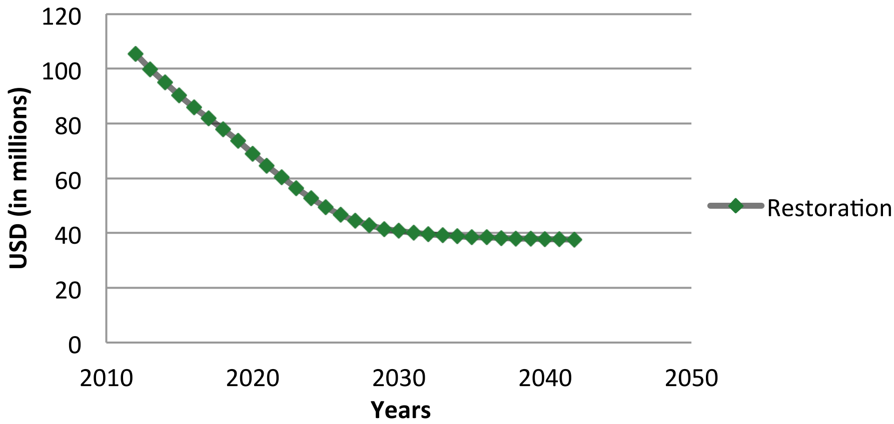

Figure 5.15 Variation in the TEV for the restoration scenario over time

When a closer look is taken at the intervention costs and the economic benefits, it becomes apparent that what at first glance appears like an investment with a very low yield, actually resulted in some profit. The costs of the interventions amount to $140,000 annually for the first two years. After the initial investments, the maintenance costs of the artificial reefs, continuous reforestation efforts and the coral nursery program sum up to an annual costs of $98,000. The net benefits were calculated for the restoration scenario with respect to the baseline scenario. The net benefits represent the sum of TEV over a 30 year period for different discount rates. In order to obtain the cost benefit ratio, the net benefits were then divided by the intervention costs (Figure 5.16). Overall, the benefits exceed the costs. Only at a discount rate of 15% do the costs exceed the benefits. The interventions that were simulated in this scenario are profitable. This can mainly be ascribed to a non-use value worth $7 million more and the tourism and recreational value worth $2 million more than in the baseline scenario.

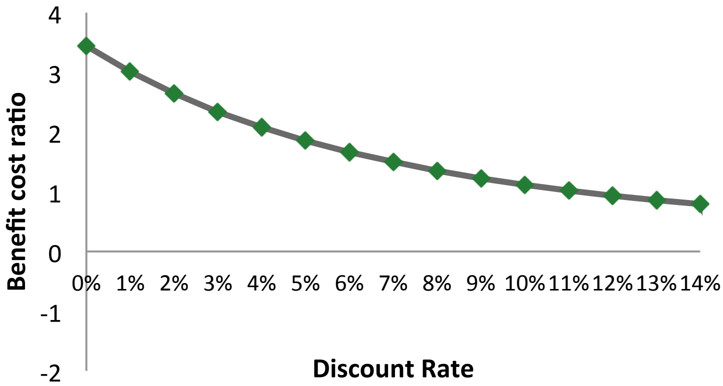

Figure 5.16 The cost benefit ratio for the restoration scenario.

5.3 Scenario 2 – Conservation

The environment of Bonaire is highly threatened by the presence of lionfish and free roaming livestock. This Conservation Scenario aims at solving this problem of invasive species by.

1) *Controlling livestock on the island:* In this conservation scenario it is assumed that of the 30,000 goats that still roam free in the baseline scenario, only 1,000 still do after 30 years. The remaining livestock is fenced and do not have the opportunity to forage on young flora.

The removal of free roaming livestock and fencing them will decrease the pressure exerted on the forests. This allows the forest flora to recover and flourish and eventually to grow into a mature forest hereby increasing the rate at which forest become mature. Furthermore, a more efficient means of managing the livestock is presumed, hereby increasing the per capita value of livestock.

2) *Eradicating lionfish:* The process of lionfish eradication involves workshops and projects to make tourists aware of their existence and to train divers to catch them. It is important to mention that there are a lot of volunteer divers that could help to catch lionfish as they understand the threat posed on marine ecosystem. The eradication of lionfish will have the biggest influence on fish stock and fish diversity as they are the first affected by their presence. Effective management of the lionfish population can results in a 2-8 times decrease. In this scenario the maximum decrease is presumed.

5.3.1 Ecological model

The effects of removing the lionfish are most apparent when looking at the fish abundance. At the onset of the simulation, the trend of the conservation scenario diverges from the baseline scenario. Removing the lionfish threat allows the fish stock to replenish (see left side of Figure 5.17). The fish population increases rapidly the first 10 years and starts to level out at 6000 tonnes. This is more than 4000 tons difference when compared to the baseline scenario.

Lionfish removal also has a positive effect on the number of fish species present (see right side of Figure 5.17). In the baseline scenario, 62 species disappear. With the lionfish have diminished in numbers in the conservation scenario, the amount of fish species only decreases very little. According to the results of the simulation only 8 species disappear).

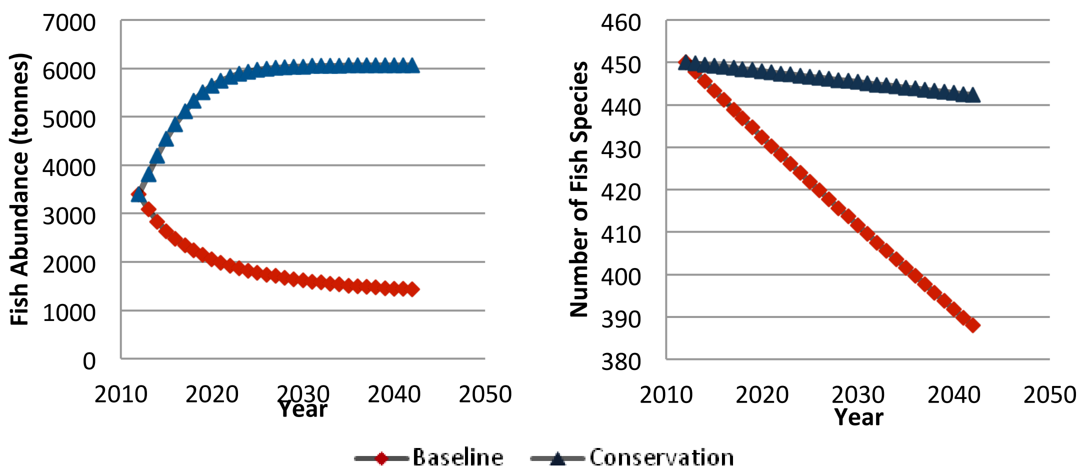

Figure 5.17 Comparison in fish abundance (left) and the number of fish species (right) for the baseline and conservation scenario

Because the fish population has the capacity to grow many herbivorous fish species also increase. These herbivorous fish species forage on algae and can now regulate there abundance. This is apparent when looking at Figure 5.18. Algae cover decreases the first ten years and levels out at around the same moment when the fish abundance reaches it maximum. A decrease in algae cover provides more space for corals to expand. At first coral cover continues to decrease, but after 5 years the decline halters and eventually corals have the space to expand. Coral cover finally reaches a maximum of 27% cover after 25 years, almost recovering to initial values. Algae cover still remains because of high ambient nutrient conditions.

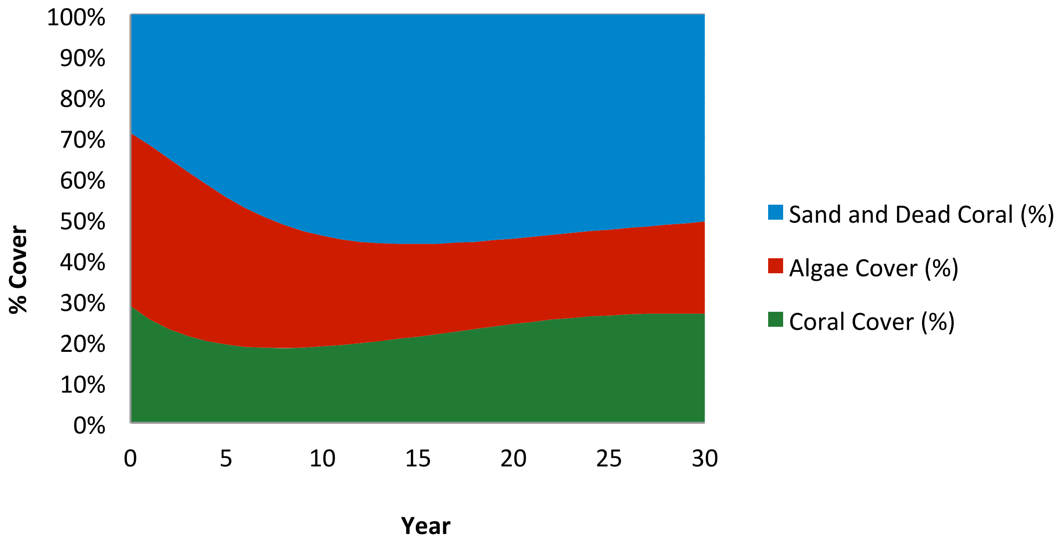

Figure 5.18 Variation in benthic cover for the conservation scenario

The effects of fencing the free roaming livestock are most apparent when looking at the forest cover (Figure 5.19). The results of the simulation show that in this scenario, the amount of mature forest cover reaches high numbers. Goat removal was implemented in stages in the model, hence there first is a small decline forest cover as free roaming livestock still predominate throughout the island. However, after 4 years goat numbers decrease to such a level that forests have the chance to mature. From 2017 and onwards, the mature forest cover keeps on increasing. After 30 years mature forest cover amounts to 7,500 hectare.

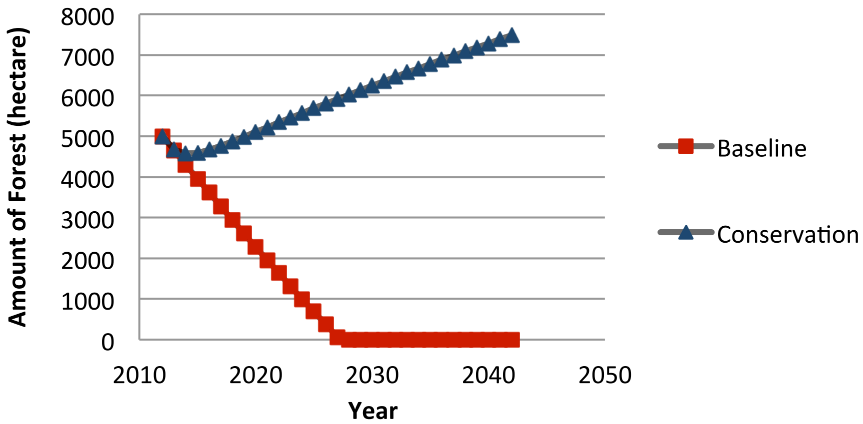

Figure 5.19 Amount of mature forest cover for the baseline and conservation scenario

Removing free roaming livestock not only affects the terrestrial environment but also the marine environment. An increase in more mature forest cover causes a decrease in sediment runoff. This means that the overall stress experienced by the corals decrease. This is visible in Figure 5.20. Coral diversity still decreases, yet at a slower rate compared to the baseline scenario.

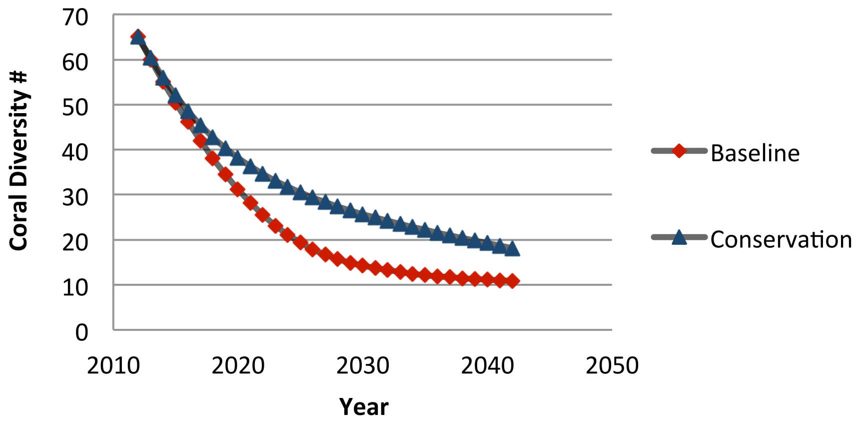

Figure 5.20 Number of coral species present for the baseline and conservation scenario

All the aforementioned changes in the environment result in a higher marine and terrestrial indicator. After 10 years the marine indicator levels out at a value of 0.5. The terrestrial indicator experiences a rapid growth the first 5 years and levels out at 0.74. In this scenario the interventions had a very positive effect on the environment. For both environments it can now be said that they are no longer in a degraded state. The effects of which are also apparent in the economical model.

5.3.2 Economic model

Despite an increase in the quality of the terrestrial and marine ecosystem, there is still a visible decline in the amount of stay over tourists. The reduction in stay over tourist can be attributed to the vast expansion of the cruise tourists. The negative feedback mechanism in the model presumes that as cruise tourists increase, stay over tourists are less likely to visit. After 30 years the cruise tourists reach a level of 547,000 visitors (Figure 5.21). The stay over tourist group reaches a minimum of 7,500 visitors per year after 30 years.

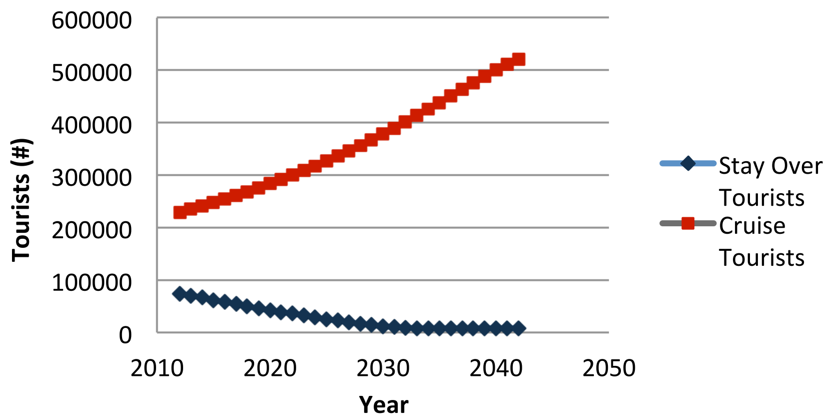

Figure 5.21 Change in the number of tourists for the conservation scenario

Although only the cruise tourists increase in numbers, there is a contrast in the recreational value between the conservation scenario and the baseline scenario (see Figure 5.22). At first, the two scenarios follow the same trend, i.e. a steady and slow decline. However, as the cruise tourists continue to increase in numbers, the recreational value diverges from its path. Eventually the value even starts to increase ending at a value of $18.5 million.

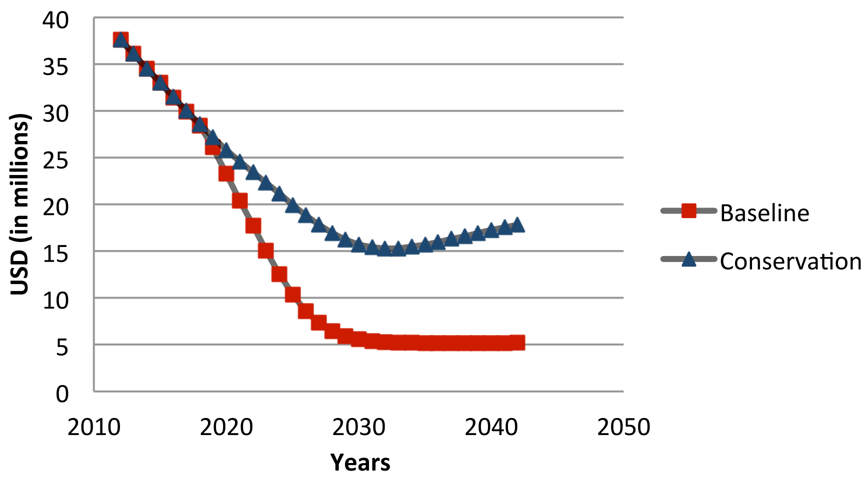

Figure 5.22 Comparison of the recreational value for the baseline and conservation scenario

The value of fisheries depends on the amount of fish caught for commercial and recreational fisheries. Since fish populations have grown in numbers, the catch per unit effort increases. This is reflected in the fact that the fisheries value increases substantially. After a period of 30 years, the value of fisheries levels out at $1.8 million.

Since the coral cover is increasing and the amount of mature forest is expanding the chances of encountering medicinal plants or extracting substances that can be of medicinal use, the biodiversity value increases. The model simulates that after 30 years the value in the conservation scenario will be $2.47 million.

Another value that differs substantially from the baseline scenario is the non-use value. The WTP of Dutch and Bonairean depends on the qualitative state of the environment. In the baseline scenario the Non-use value drops by more than $30 million. In the conservation scenario, however, the value stabilises at $44 million (see Figure 5.23).

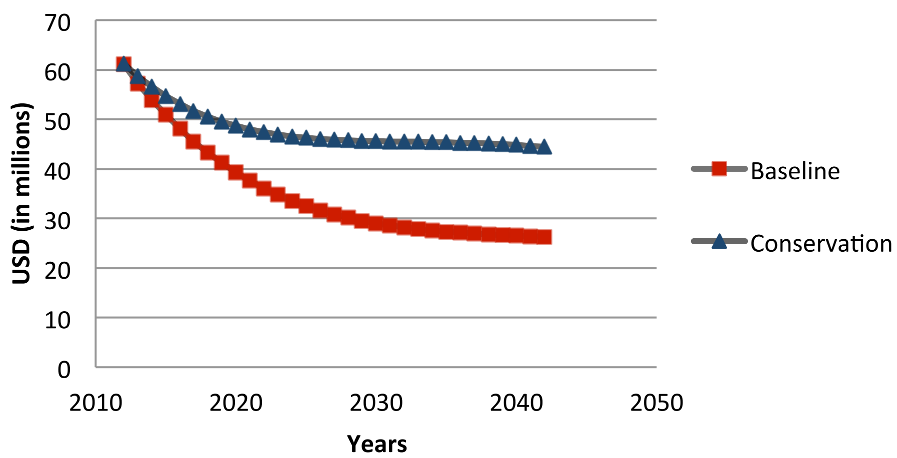

Figure 5.23 Comparison of the Non-Use value for the baseline and conservation scenario

The interventions in the conservation scenario have a substantial positive influence on the different ecosystem services. As a result the TEV increases compared to the baseline scenario: the TEV decreases just as in the baseline scenario, however in the conservation scenario the trend diverges. After 30 years the value reaches $70 million, compared to a TEV of $38 million after 30 years in the baseline scenario (Figure 5.15). As a result higher NPV is also emerging (Figure 5.24). The difference between the scenarios decreases as the discount rate increase, but there remains to be a pronounced difference.

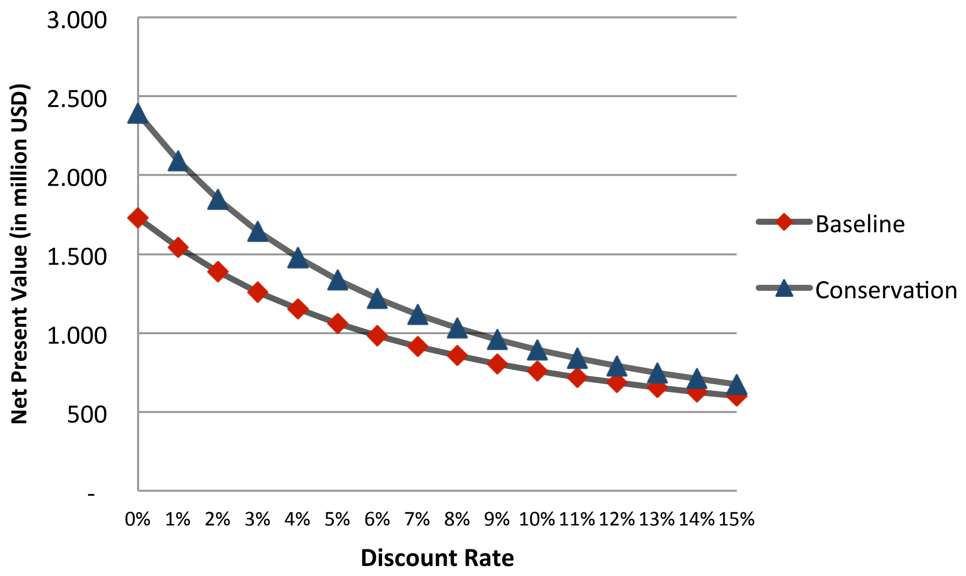

Figure 5.24 Comparison of the NPV for the baseline and conservation scenario

Finally the benefit cost ratio is a good indicator whether or not the interventions were worth the investment. The most expensive part of the interventions are catching and fencing the goats. It is presumed that this costs $231,000 the first 10 years and $10,000 per year for maintenance for the remaining period. Removal of the lionfish is $40,000 for the first 3 years and $10,000 per year for the remaining period. At its highest the cost benefit ratio is 227 at its lowest 26. Despite the discount rate, the investment has a very high yield.

5.4 Scenario 3 – Sewage treatment plant

To reduce the amount of nutrients entering coastal waters, there are plans to construct a waste water treatment plant in Bonaire near Kralendijk. Slijkerman et al (2011) determined that the amount dissolved organic nitrogen present in coastal waters is on average 1.51µM. Once the sewage system is operational this concentration will not exceed the threshold of 0.5 µM. Some nutrient loading will remain as the nutrients will continue to load into the sea from agricultural activities through groundwater. This scenario will analyze the ecological and economic effects of establishing a sewage plant on Bonaire.

5.4.1 Ecological model

The most pronounced effect of nutrient reduction is on coral cover and coral diversity. One of the greatest stressors corals endure is the high ambient nutrient conditions. Not only does it hamper their growth, it creates favourable conditions for algae to outcompete the corals. The number of coral species that disappear due to nutrients is considerably less in this scenario. Also, in no other scenario does the coral cover reach such high levels. After 20 years the total coral cover is 47%, while algae cover is reduced to a meagre 6% (Figure 5.25).

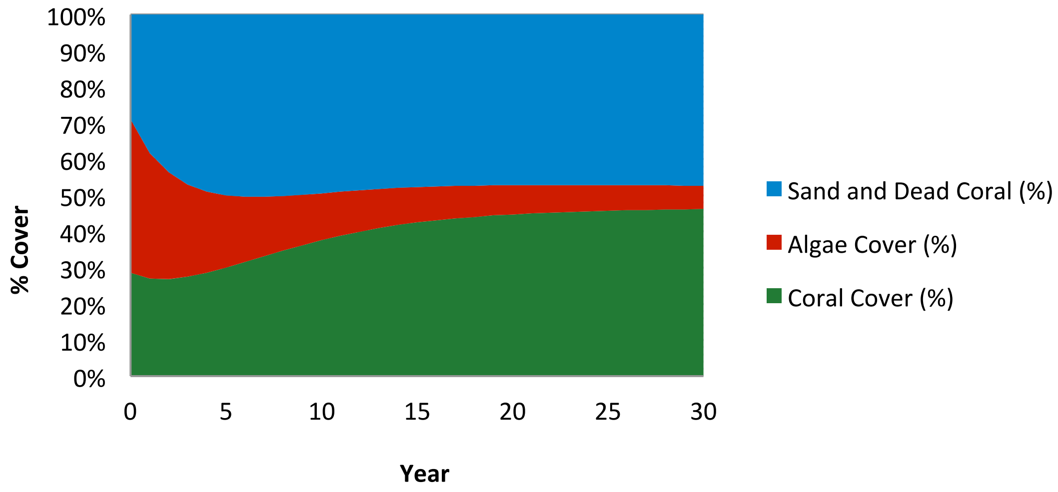

Figure 5.25 Variation in the benthic cover for the sewage treatment scenario

The other ecological parameters indicate little to no improvement in the model. Fish diversity and abundance follows the same trend as in the baseline scenario. The terrestrial parameters are also not affected. No negative effects by nutrient loading on the terrestrial ecosystem are assumed.

Despite the fact that the implementation of a sewage treatment plant only seems to affect coral and algae cover, the marine indicator reaches its highest level of 0.59 (left side of Figure 5.26). The main reason is that the coral cover is one of the parameters that has the heaviest weight in determining the marine indicator.

The terrestrial indicator actually reaches a lower level than in the baseline scenario (right side of Figure 5.26). The reason for this is twofold. First, there is nothing done to improve the qualitative state of the reef, and second the amount of pressure exerted by tourists increases as the tourists increase (see next section).

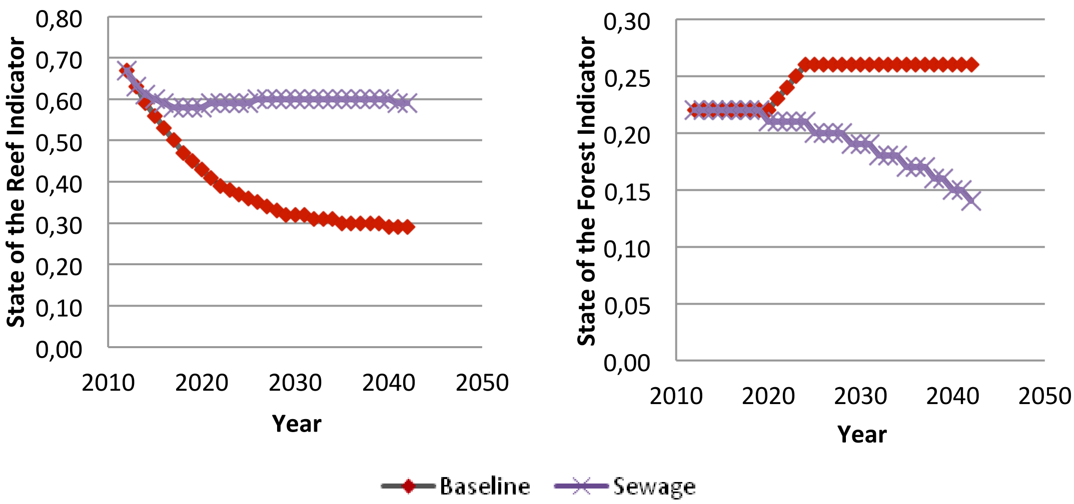

Figure 5.26 Comparison of the Marine Indicator (left) and the Terrestrial Indicator (right) for the baseline and sewage scenario

5.4.2 Economic model

Because the quality of the reef is increasing and most tourists that visit the island come for the aesthetic value of the coral reefs, there is an expansion in tourists visiting annually. Once again, it is the cruise tourists that increase substantially, from 230,000 to 550,000 in the 30 year period (see Figure 5.27). Such an increase in the cruise tourists has a negative influence on the visitation rate of stay over tourists. As a result, instead of seeing an increase in stay over tourists as one would expect when the quality of the reef improves, a steady decline is visible. The stay-over tourists reach a minimum of 7,500. Nonetheless, there is an increase in the tourism and recreational value (Figure 5.28). It follows the same trend as in the baseline scenario for the first 8 years, upon which it diverges and decreases at a lower rate. Finally, as the cruise tourists reach such high levels, the tourism and recreational value also starts to increase.

Another value that is influenced substantially is the coastal protection value. Because the coral cover does not decrease as much as in the other scenarios, the capacity of the coral ecosystem to protect the mainland against storms increases.

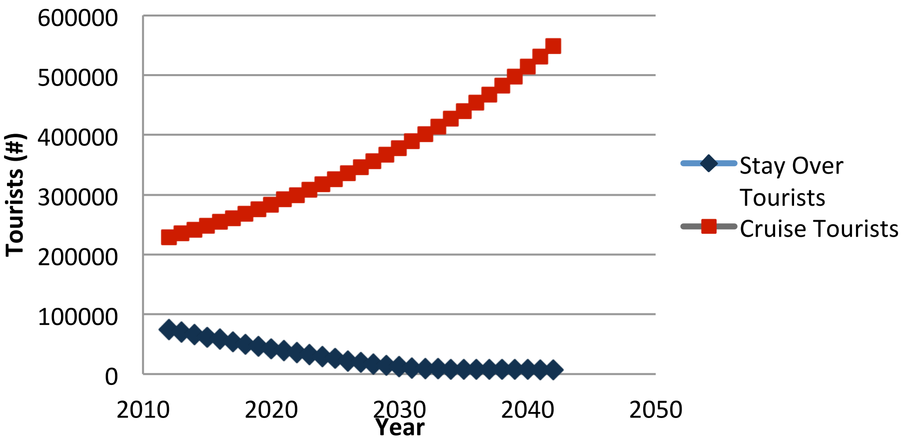

Figure 5.27 Change in the number of tourists for the sewage treatment scenario

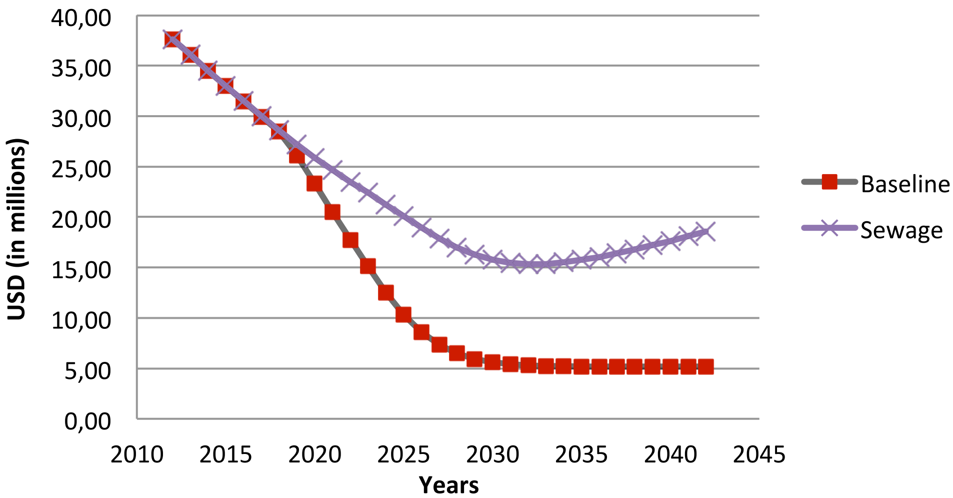

Figure 5.28 Comparison in the tourism and recreational value for the baseline and sewage scenario

The results of the model indicated that the marine ecosystem remains within a healthy state, the main reason for this being high coral cover. The WTP of Dutch and Caribbean citizens therefore also remain high (Figure 5.29). The Non Use value reaches the highest levels in this scenario. After 30 years the value ends up at $47 million.

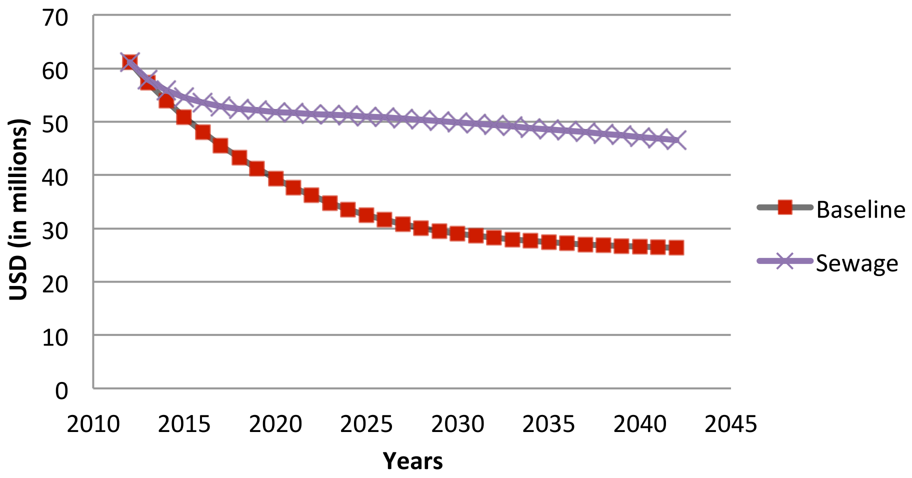

Figure 5.29 Comparison of the non-use value between the baseline and sewage scenario

Other values that contribute also increase are the amenity and the biodiversity value. All caused by an increase in coral cover. The biodiversity value of the corals increases with 7 million USD. After 30 years, the amenity value has increased by $70,000 when compared to the baseline scenario.

The creation of the sewage treatment plant is the most expensive intervention proposed within this report. However, it has also proven to be the scenario with the highest TEV. After the 30 year simulation period the TEV reaches a value of $71 million (Figure 5.30). The sewage treatment plant resulted in a TEV that is two million higher than the conservation scenario and over $33 million more than the baseline scenario.

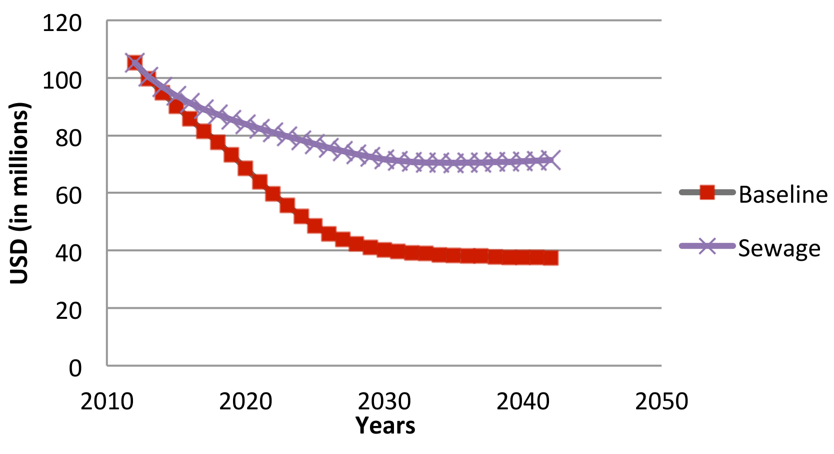

Figure 5.30 Comparison of the TEV between the baseline and sewage scenario

The allocation of the different benefits is similar to the ones in the other scenarios: the non-use and the tourism and recreational value contribute the most to the TEV. The net present value is the highest at each discount rate compared to all other scenarios. At the lowest discount rate the NPV is $2,450 million and at the highest discount rate the NPV is $680 million (Figure 5.31). Despite having the highest TEV, the cost benefit ratio is not as good as in the conservation scenario. With a discount rate of 0% the highest ratio value is 7.45 and at a discount rate as high as 15% the benefit cost ratio still exceeds 1 (sees Figure 5.32). This proves that the investment in the sewage treatment plant pays off, even at high discount rates.

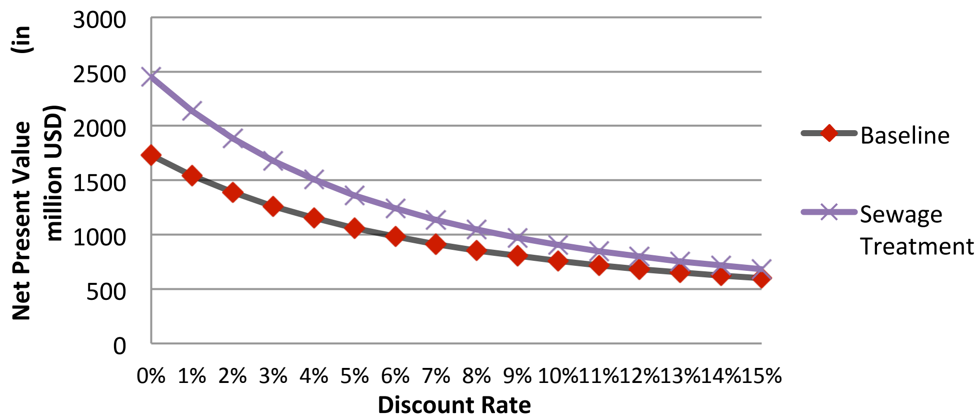

Figure 5.31 Comparison of the NPV between the baseline and sewage scenario

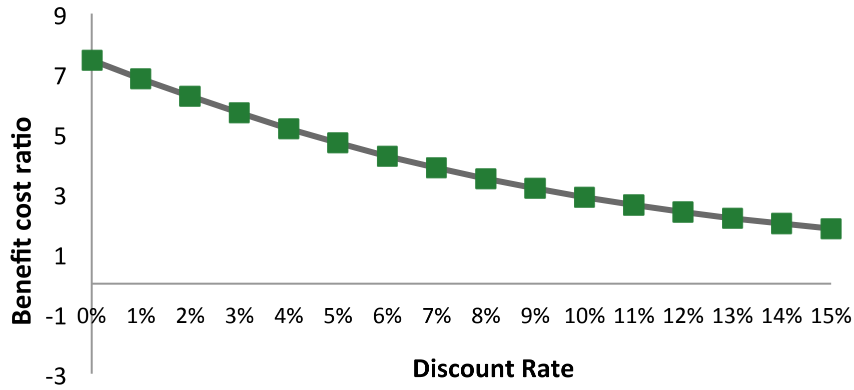

Figure 5.32 Cost benefit ratio at different discount rates for the sewage scenario

6 Conclusions & Recommendations

The analysis of the wide range of ecosystem services provided by the natural environment of Bonaire generates many opportunities for decision makers to improve economic and environmental policies on the island. To deliver information to decision makers the first step is to calculate the Total Economic Value (TEV) and to grasp the role of nature in the Bonairean economy. Next, an extended cost-benefit analysis of alternative future scenarios provides an objective means of deciding which interventions in the economy and environment generate the highest yield. Such an integral approach is intended to ensure the betterment of Bonaire's environment while at the same time warranting sustainable economic development.

6.1 Validity of the model and lessons to be learned

The mutual relationship between ecological and economic processes of coral reef ecosystems is strong. Therefore, a multidisciplinary approach is essential in tackling the multiple threats that currently face the fragile ecosystems of Bonaire. The ecological economic simulation model is nothing but a first attempt to provide a platform for ecologists and economists to exchange knowledge on the degradation and management of the nature of Bonaire. Some will consider the simplifications that have been made in building the model and in conducting the management scenarios unacceptable. Indeed, we acknowledge that this is the weaker aspect of the approach followed. Yet, although the model simulations are far from accurate and sometimes lack the desired level of comprehension, they do provide a representation of the current state of the scientific knowledge available in the literature. Moreover, unlike most mono-disciplinary studies, the simulation model contains the main elements required to oversee the full picture of nature reef management on Bonaire and thereby enables scientist and managers to evaluate ecological and economic impacts effectively. Finally, the model does provide insights into the economic importance of nature for Bonaire and also sheds light on the effectiveness of various potential management interventions, although these lessons are surrounded by great uncertainty. Some of the main lessons of the study are summarised below.

6.2 Total Economic Value versus Total Financial Value

By summing up the worth of the range of valued ecosystem services, the TEV of the natural environment of Bonaire is estimated to be more than $105 million annually. To get an idea of the importance of nature for Bonaire, it is useful to compare the TEV of nature to the Gross Domestic Product (GDP). The GDP of the Bonaire economy was around $224 million in 2008, which means that welfare derived either directly or indirectly from ecosystems is almost half of Bonaire's economy. Compared to other island-based studies, this shows that the economy of Bonaire has an extremely high dependence on their natural environment. Although the TEV of nature on Bonaire is very large, this aggregated value is composed of numerous latent welfare-related values that are not necessarily translated into actual monetary flows. For example, the value by Dutch mainland citizens is a genuine economic value, yet, at the same time, this ecosystem service is predominantly a non-financial value (i.e. its value is not [fully] transferred in money terms to the financial economy of Bonaire). Of the TEV of $105 million, only one-third of this amount (i.e. $37 million) is truly traceable in the financial accounts of the economy of Bonaire. The majority of this financial value is captured by the tourism sector.

6.3 Costs and benefits of environmental measures

Through the use of simulation models, scenario development, and cost-benefit analysis the efficiency of various interventions is determined. Out of the extensive analysis of the ecosystem services and the different scenarios one result becomes very clear: an ounce of prevention is worth a pound of cure. In other words, it is more efficient to prevent extensive environmental damage than trying to revitalize the environment while there are still threats at hand. With the current threats unmanaged, the TEV of Bonairean nature will decrease from $105 million today to around $60 million in ten years time and to less than $40 million in 30 years. Therefore, addressing the main threats proves to be very cost effective, e.g. by removing the threat of goats and lionfish, the environment has the possibility to regenerate. Similar conclusions are drawn for the economic efficiency of improved sewage treatment. This demonstrates that interventions and policies need to aim at preventing damage to Bonaire's nature.

6.4 Stakeholder engagement

This project intensively involved stakeholders from the start. Therefore, the local government began the research with a public kick-off, followed by several workshops. Relevant stakeholders from various backgrounds participated in all sessions. These included local and national policy makers, nature conservation organisations, local industry (e.g. tourism industry), but also financial services and waste management companies. One of the objectives was building capacity with a select group of stakeholders educating them on socio-economic valuation and learning from them which ecosystems, ecosystem services and threats are most relevant on Bonaire. During the complete research cycle these stakeholders provided continuous feedback and frequently reviewed research output. A final workshop took place in which stakeholders discussed the results and applied the monetary estimates in extended cost benefit analyses and explained the results from their own perspective in the documentary. This process restated the lesson that raising awareness locally as well as nationally is of crucial importance to generate the necessary support for preserving nature as an important economic source for Bonaire. Moreover, strong societal support is needed to convince local decision makers to apply the recommendations of the study. Examples of recommendations are for the improvement and further development of sustainable financing mechanisms, emergency plans or damage assessments protocols in response to increased hurricane events, and to improve policies that should guide Bonaire towards a sustainable green island economy.

References

Van Zanten, B., Van Beukering, P. and Wolfs E. (2012) on the coastal protection value of coral reefs of Bonaire. Report number R12-XX, Institute for Environmental Studies VU University Amsterdam, the Netherlands, December 2012

Schep S, Brander L, Van Beukering P, Wolfs E (2012a). The touristic value of nature on Bonaire. A multiple valuation techniques approach. Report number R12-XX, Institute for Environmental Studies VU University Amsterdam, the Netherlands, December 2012

Schep S, Johnson AE, Van Beukering P, Wolfs E. (2012b). The fishery value of coral reefs in Bonaire. Applying various valuation techniques. Report number R12-XX, Institute for Environmental Studies VU University Amsterdam, the Netherlands, December 2012

Lacle F, Wolfs E, Van Beukering P, Brander L. (2012). Recreational and cultural value of Bonaire s nature to its inhabitants. Report number R12-XX, Institute for Environmental Studies VU University Amsterdam, the Netherlands, December 2012

Van Beukering P, Botzen W, Wolfs E. (2012). The non-use value of nature ion the Netherlands and the Caribbean Netherlands. Applying and comparing contingent valuation and choice modelling approaches. Report number R12-XX, Institute for Environmental Studies VU University Amsterdam, the Netherlands, December 2012

Van Beukering P. and Wolfs E. (2012). Essays on economic values of nature of Bonaire. A desk study. Report number R12-XX, Institute for Environmental Studies VU University Amsterdam, the Netherlands, December 2012

Albins, M. A. (2011). Effects of the Invasive Pacific Red Lionfish Pterois volitans on Native Atlantic Coral-reef Fish Communities. PhD Thesis, Oregon State University, Department of Zoology, 224 pp.

Albins, M. A. & Hixon, M. A. (2011). Worst case scenario: potential long-term effects of invasive predatory lionfish (Pterois volitans) on Atlantic and Caribbean coral-reef communities. *Environmental Biology of Fishes*. doi:10.1007/s10641-011-9795-1

Alevizon, W. (2009). Caribbean Coral Reefs: Types, Characteristics, *Marine Life*. Retrieved from http://www.coral-reef-info.com/caribbean-coral-reefs.html. Accessed. June 13, 2012

Annual Statistics Report. (2009). *Bonaire Tourism*. Bonaire Report 2008. Tourism Corporation Bonaire, Dutch Caribbean, 35 pp.

Arias-González, J. E., Nuñez-Lara, E., González-Salas, C., & Galzin, R. (2004). Trophic models for investigation of fishing effect on coral reef ecosystems. *Ecological Modelling*, *172*(2-4), 197-212. doi:10.1016/j.ecolmodel.2003.09.007

Art, H. W. (1993). *Eutrophication - A dictionary of ecology and environmental science* (1st edition). New York. Henry Holt and Company, 196 pp.

Bak, M., Nieuwland, G., & Meesters, E. H. (2009). Coral Growth Rates Revisited After 31 Years: What is Causing Lower Extension Rates in Acropora Palmata?. *Buletin of Marine Science, 84*(3), 287-294.

Bak, R. P. M., Nieuwland, G., & Meesters, E. H. (2005). Coral reef crisis in deep and shallow reefs: 30 years of constancy and change in reefs of Curacao and Bonaire. *Coral Reefs*, *24*(3), 475-479. doi:10.1007/s00338-005-0009-1

Baker, A. C., Glynn, P. W., & Riegl, B. (2008). Climate change and coral reef bleaching: An ecological assessment of long-term impacts, recovery trends and future outlook. *Estuarine, Coastal and Shelf Science*, *80*(4), 435-471. Elsevier Ltd. doi:10.1016/j.ecss.2008.09.003

Barott, K. L., Rodriguez-Mueller, B., Youle, M., Marhaver, K. L., Vermeij, M. J. a, Smith, J. E., & Rohwer, F. L. (2012). Microbial to reef scale interactions between the reef-building coral Montastraea annularis and benthic algae. *Proceedings. Biological sciences / The Royal Society*, *279*(1733), 1655-1664. doi:10.1098/rspb.2011.2155

Blackwood, J. C., Hastings, A., & Mumby, P. J. (2011). A model-based approach to determine the long-term effects of multiple interacting stressors on coral reefs. *Ecological* *applications: a publication of the Ecological Society of America*, *21*(7), 2722-33. Retrieved from http://www.ncbi.nlm.nih.gov/pubmed/22073655

Bonaire Department of Physical Planning (2010) Ruimtelijk Ontwikkelingsplan Bonaire http://www.bonairegov.an/attachments/998_Ruimtelijk_Ontwikkelingsplan_Bonaire_vastgesteld.pdf

Botzen, W., van Beukering, P., & Wolfs E. (2012). *The non-use value of nature in the Netherlands and the Netherlands Caribbean.* IVM Report, VU University, Amsterdam.

Box, S., & Mumby, P. (2007). Effect of macroalgal competition on growth and survival of juvenile Caribbean corals. *Marine Ecology Progress Series*, *342*, 139-149. doi:10.3354/meps342139

Brock, A., Ferrer, M., & Scholte, S. (2011). *Economic Valuation of Medicinal Plants for Local and Global Use The Case of Bonaire*. IVM Report, VU University, Amsterdam. 33 pp.

Bruckner, A., Williams, A., & Renaud, P. (2010). *An Assessment of the Health and Resilience of Bonaire ' s Coral Reefs*. Khaled bin Sultan Living Oceans Foundation Report, 51 pp.

Burkepile, D. E., & Hay, M. E. (2008). Coral Reefs. *Encyclopedia of Ecology*, *1*, 784-796.

Cartier, J. R. and C. (1999). *Issues in Applied Coral Reef Biodiversity Valuation: Results for Montego Bay , Jamaica*. Report for World Bank Research Committee. 279 pp.

CBS (Central Bureau of Statistics). (2011). *A million households more by 2045*. Retrieved June 12, 2012 from http://www.cbs.nl/en-gb/menu/themas/bevolking/publicaties/artikelen/archief/2011/2011-3365-wm.htm.

Central Bureau of Statistics. (2001). *Census 2001 Publication Series Demography of the Netherlands Antilles*. Department of Publication and Information, Fort Amsterdam. 95pp.

Central Bureau of Statistics. (2010). *Statistical Yearbook Netherlands Antilles 2010.*Willemstad, Curacao. 120 pp.

Centraal Bureau voor de Statistiek. (2012). *Bevolkingsontwikkeling Caribisch Nederland; geboorte, sterfte, migratie*. Retrieved May 25, 2012 from http://statline.cbs.nl/StatWeb/publication/?DM=SLNL&PA=80539ned&D1=0-1,9-10&D2=a&D3=a&HDR=T&STB=G1,G2&CHARTTYPE=1&VW=T

Cesar, H., Pet-soede, L., & Burke, L. (2003). *The Economics of Worldwide Coral Reef Degradation.* Report for Cesar Environmental Economics Consulting. 24 pp.

Cesar, H., van Beukering, P., Pintz, S., & Dierking, J. (2002). *Economic valuation of the coral reefs of Hawaii*. Report NOAA, 120 pp.

Cesar, H., van Beukering, P., & Romilly, G. de B. (2003). *Mainstreaming Economic Valuation in Decision Making: Coral Reef Examples in selected CARICOM-countries*. Report Arcadis and World Bank. 145 pp.

CTO (Caribbean Tourism Organization). (2011). *Visitor Arrival Summary Anguilla and Bonaire*. Retrieved June 18, 2012, from Visitor Arrival Summary

Costanza, R., & Gottlieb, S. (1998). Modelling ecological and economic systems with STELLA: Part II. *Ecological Modelling*, *112*(2-3), 81-84. doi:10.1016/S0304-3800(98)00073-8

Costanza, R., & Voinov, A. (2001). Modeling ecological and economic systems with STELLA: Part III. *Ecological Modelling*, *143*(1-2), 1-7. doi:10.1016/S0304-3800(01)00358-1

Côté, I., & Maljkovic, a. (2010). Predation rates of Indo-Pacific lionfish on Bahamian coral reefs. *Marine Ecology Progress Series*, *404*, 219-225. doi:10.3354/meps08458

Deza (Department of Economic and Labour Affairs Bonaire, (2008). *The Bonaire Economic Note*. Island Territory of Bonaire

Deza, Affairs, (Department O. E. A. L., & Bonaire, (2005). *The Bonaire Economic Note*. Island Territory of Bonaire

Dailer, M. L., Smith, J. E., & Smith, C. M. (2012). Responses of bloom forming and non-bloom forming macroalgae to nutrient enrichment in Hawai'i, USA. *Harmful Algae*, *17*, 111-125. Elsevier B.V. doi:10.1016/j.hal.2012.03.008

De Meyer K. (1998). Bonaire, Netherlands Antilles. *Coastal region and small island*. Papers 3.

Debrot, A. O., & Bugter, R. (2010). *Climate change effects on the biodiversity of the BES islands*.Altera Report 2081, Wageningen. 40 pp.

Dew, I. M. (2001). Theoretical model of a new fishery under a simple quota management system. *Ecological Modelling*, *143*(1-2), 59-70. doi:10.1016/S0304-3800(01)00356-8

Edwards, H. J., Elliott, I. A., Eakin, C. M., Irikawa, A., Madin, J. S., Mcfield, M., Morgan, J. A., van Woesik R., & Mumby P.J. (2011). How much time can herbivore protection buy for coral reefs under realistic regimes of hurricanes and coral bleaching? *Global Change Biology*, *17*(6), 2033-2048. doi:10.1111/j.1365-2486.2010.02366.x

Evans, G.R. (1997). *Economic Models* - Chapter 1. Retrieved July 17, 2012 from http://www2.hmc.edu/~evans/chap1.pdf

Gardner, T. A, Côté, I. M., Gill, J. A, Grant, A., & Watkinson, A. R. (2003). Long-term region-wide declines in Caribbean corals. *Science (New York, N.Y.)*, *301*(5635), 958-60. doi:10.1126/science.1086050

Groenenboom, W. and K. R. (2009). *Document Bonaire* (p. 23). Leveroij BV. Retrieved June 13, 2012 from http://books.google.nl/books?id=abvJGRo_EuoC&hl=nl&source=gbs_navlinks_s

Hal, C.A.S. & Day, J.W.(1990). *Ecosytem Modeling in Theory and Practice: An Introduction with Case Histories. University Press of Colorado. pp. 7–8. ISBN 0-87081-216-5.*

Harty, M. (2011). Christmas tree worms (Spirobranchus giganteus) as a potential bioindicator species of sedimentation stress in coral reef environments of Bonaire, Dutch Caribbean. *PhysisJournal of marine science.* vol.IX, 31 pp.

Hoegh-Guldberg, O. (1999). Climate change, coral bleaching and the future of the world's coral reefs. *Marine and Freshwater Research*, *50*, 839-66.

Hoegh-Guldberg, O., Mumby, P. J., Hooten, a J., Steneck, R. S., Greenfield, P., Gomez, E., Harvell, C. D., et al. (2007). Coral reefs under rapid climate change and ocean acidification. *Science (New York, N.Y.)*, *318*(5857), 1737-42. doi:10.1126/science.1152509

Holmes, G., & Johnstone, R. W. (2010). Modelling coral reef ecosystems with limited observational data. *Ecological Modelling*, *221*(8), 1173-1183. Elsevier B.V. doi:10.1016/j.ecolmodel.2010.01.010

ICRI (International Coral Reef Initiative). (n.d.). *Status of and threat to coral reefs*. Retrieved June 16, 2012 from http://www.icriforum.org/about-coral-reefs/status-and-threat-coral-reefs

InfoBonaire. (2012). *Activities on Bonaire*. Retrieved June 17, 2012 from http://www.infobonaire.com/activities.html

IUCN. (2011). *Coral Reef Resilience Assessment of the Bonaire National Marine Park , Netherlands Antilles*. Gland, Switzerland. 51 pp.

Lacle, F. A. (2012). *Recreational and cultural value of Bonaire' s nature to its inhabitants*. Research project Msc. Faculty of Earth and Life Sciences, Vrije University Amsterdam. 86 pp.

McClary, M. & (2010). *The Encyclopedia of Earth - Threats to coral reefs*. Retrieved June 10, 2012 from http://www.eoearth.org/article/Threats_to_coral_reefs?topic=49513#gen9

McClanahan, T. R. (1995). A coral reef ecosystem-fisheries model: impacts of fishing intensity and catch selection on reef structure and processes. *Ecological Modelling*, *80*(1), 1-19. doi:10.1016/0304-3800(94)00042-G

Meindertsma, J. D. (n.a.). *Cost Benefit Analysis III: Environmental Perspective - Key Concept.* Learning resource for ICRA. Retrieved July 10, 2012,from http://www.icra-edu.org/objects/anglolearn/Cost_Benefit_Analysis_3-Key_Concepts(new).pdf

Millennium Ecosystem Assessment. (2005). Ecosystems and Human Well-being: Synthesis. Island Press, Washington, DC.

Millspaugh, J. J., & Thompson F. R. (2009). *General Principles for Developing Landscape Models for Wildlife Conservation. Models for planning wildlife conservation in large landscapes*. Academic Press. p. 1. ISBN 978-0-12-373631-4.

MSNA & A (Meteorological Service of the Netherlands Antilles & Aruba). (2008). *Climatological report* (Vol. 46). doi:10.1002/mus.23469

Mumby, P. J., Harborne, A. R., & Brumbaugh, D. R. (2011). Grouper as a natural biocontrol of invasive lionfish. *PloS one*, *6*(6), e21510. doi:10.1371/journal.pone.0021510

Mumby, P. J., Hedley, J. D., Zychaluk, K., Harborne, A. R., & Blackwell, P. G. (2006). Revisiting the catastrophic die-off of the urchin Diadema antillarum on Caribbean coral reefs: Fresh insights on resilience from a simulation model. *Ecological Modelling*, *196*(1-2), 131-148. doi:10.1016/j.ecolmodel.2005.11.035

Mumby, P. J., & Steneck, R. S. (2011). The resilience of coral reefs and its implications for reef management. *Coral Reefs: An Ecosystem in Transition.* (Z. Dubinsky & N. Stambler, Eds.), 509-519. doi:10.1007/978-94-007-0114-4

Myers, R. a, Bowen, K. G., & Barrowman, N. J. (1999). Maximum reproductive rate of fish at low population sizes. *Canadian Journal of Fisheries and Aquatic Sciences*, *56*(12), 2404-2419. doi:10.1139/f99-201

Newman, M. J. H., Paredes, G. a, Sala, E., & Jackson, J. B. C. (2006). Structure of Caribbean coral reef communities across a large gradient of fish biomass. *Ecology letters*, *9*(11), 1216-27. doi:10.1111/j.1461-0248.2006.00976.x

NOAA (National Oceanic and Atmospheric Administration). (2008). *Anthropogenic Threats to Corals*. Retrieved May 18, 2012 from http://oceanservice.noaa.gov/education/kits/corals/coral09_humanthreats.html

NOAA (National Oceanic and Atmospheric Administration). (2011). *NOAA's National Ocean Service: Coral Reefs.* Retrieved May 15, 2012, from http://oceanservice.noaa.gov/oceans/corals/

Parsons, G. R., & Thur, S. M. (2007). Valuing Changes in the Quality of Coral Reef Ecosystems: A Stated Preference Study of SCUBA Diving in the Bonaire National Marine Park. *Environmental and Resource Economics*, *40*(4), 593-608. doi:10.1007/s10640-007-9171-y

Polunin, I D, W. N. V. C. (2001). Large-scale associations between macroalgal cover and grazer biomass on mid-depth reefs in the Caribbean. *Coral Reefs*, *19*, 358-366. doi:10.1007/s003380000121

Prévost, E., Crozier, W. W., & Schön, P.-J. (2005). Static versus dynamic model for forecasting salmon pre-fishery abundance of the River Bush: a Bayesian comparison. *Fisheries Research*, *73*(1-2), 111-122. doi:10.1016/j.fishres.2005.01.002

Rogers, C. S. (1990). Responses of coral reefs and reef organisms to sedimentation. *Marine Ecology Progress Series*, *62*, 185-202.

Sandin Stuart A., Eugenia M. Sampayo, A. M. J. A. V. (2008). Coral reef fish and benthic community structure of Bonaire and Curaçao, Netherlands Antilles. *Caribbean Journal of Science*, *44*(2), 137-144.

Sarkis, S., van Beukering, P. J.H., & McKenzie E. (2010). *Total economic value of Bermuda's coral reefs*. Technical Report, Department of conservation services, Government of Bermuda. 228 pp.

Slijkerman DME, RBJ Peachey, P. H., & Meesters, H. (2011). *Eutrophication status of Lac, Bonaire, Dutch Caribbean Including proposals for measures.* Report Environics Consulting Services, NV, C093/11. 40 pp.

Stinapa, Bonaire. (2008). *Bonaire National Marine Park: Background information.* Retrieved from http://www.bmp.org/. Accessed.June 10, 2012.

Stinapa-WSNP & BNMP. (2012). *Washington Slagbaai National Park.* Retrieved May 18, 2012 from http://www.washingtonparkbonaire.org/index.html

Susan, D. (2011). *Coral Bleaching Alert as Sea Temperatures Rise*. Retrieved May 29, 1BC, from http://www.bonaireinsider.com/index.php/bonaireinsider/coral_bleaching_alert_as_sea_temperatures_rise/

System, I. (2012). *Stella - Systems Thinking for Education and Research*. Retrieved April 15, 2012 from http://www.iseesystems.com/softwares/Education/StellaSoftware.aspx

Systems Management College. (2001). *Systems Engineering Fundamentals.* Supplementary Text Prepared By The Defense Acquisition University Press Fort Belvoir, Virginia 22060-5565. 170 pp

Tanner, J. E. (1995). Competition between scleractinian corals and macroalgae: An experimental investigation of coral growth, survival and reproduction. *Journal of Experimental Marine Biology and Ecology*, *190*(2), 151-168. doi:10.1016/0022-0981(95)00027-O

The Nature Conservancy. (2012). *Bonaire.* Coral reefs-A reef resilience toolkit module. Retrieved July 17, 2012 from http://www.reefresilience.org/Toolkit_Coral/C8_Bonaire.html

Thur, S. M. (2010). User fees as sustainable financing mechanisms for marine protected areas: An application to the Bonaire National Marine Park. *Marine Policy*, *34*(1), 63-69. Elsevier. doi:10.1016/j.marpol.2009.04.008

van Beukering, P., Haider, W., Longland, M., Cesar, H., Sablan, J., Shjegstad, S., Beardmore, B., et al. (2007a). *The economic value of Guam's coral reefs*. Technical Report nb.116. University of Guam Maryne Laboratory. 130 pp

Van Beukering, P., Brander, L., Tompkins, E. and McKenzie, E., (2007b), Valuing the Environment in Small Islands - An Environmental Economics Toolkit, ISBN 978 1 86107 5949

van Beukering, P., Brander, L., van Zanten, B., Verbrugge, E., & Lems, K. (2011). *The Economic Value of the Coral Reef Ecosystems of the United States Virgin Islands*.IVM Report, R-11/06, VU University Amsterdam. 160 pp.

van Kekem, A. J., Roest, C. W. J., & and van der Salm, C. (2006). *Critical review of the proposed irrigation and effluent standards for Bonaire*. Alterra Report nb.1289. Wageningen. 129 pp.

Vermeij, M. J. a, van Moorselaar, I., Engelhard, S., Hörnlein, C., Vonk, S. M., & Visser, P. M. (2010). The effects of nutrient enrichment and herbivore abundance on the ability of turf algae to overgrow coral in the Caribbean. *PloS one*, *5*(12), e14312. doi:10.1371/journal.pone.0014312

Vermeij, M. J. A. (2012). *The current state of Curacao's coral reefs*.Report for Carmabi Foundation/University of Amsterdam. 34 pp.

Watzold, F., Drechsler, M., Armstrong,C. W., Artner, S.F., Grimm,V., Huth,A., Perrings, C., Possingham,H.G., Shogren,J. F., Skonhoft,A., Verboom-Vasiljev ,J. & Wissel, C.(2006). Ecological-Economic Modeling for Biodiversity. *Conservation Biology* Volume 20, No. 4, 1034–1041

Management: Potential, Pitfalls, and Prospects

West Bay Website. (2012). *West Bay Harbour.* Retrieved July 17, 2012, from http://www.westbay.co.uk/harbour/

Wieggers, M. W. (2007). *Impact of Increased Nutrient Input on Coral Reefs on Bonaire and Curacao*. Utrecht University. Report University Utrecht. 66 pp.

Wolfs, E. (2010). *What ' s Bonaire Nature worth?*.Project proposal. 48 pp.

World Bank. (2012). *GDP growth (annual %)*. Retrieved June 18, 2012, from http://data.worldbank.org/indicator/NY.GDP.MKTP.KD.ZG/countries?display=graph

WRI (World Resources Institute). (n.d.). *Reefs at Risk in the Caribbean.Retrieved May 25, 2012 from http://www.wri.org/publication/content/7918*

Annex A Nutrients and eutrophication

The waters surrounding coral reefs are typically oligotrophic or low in concentrations of inorganic nutrients (nitrogen and phosphorous). Under these circumstances, reef building corals usually dominate and fleshy algae are kept in low abundance because of a combination of both low nutrients and high grazing activity from fish and invertebrate species. It has been suggested that by altering either of these factors (nutrient levels or herbivory) that the competitive interaction between corals and algae will shift. Increasing nutrient levels may accelerate algal growth rates to a point where they may overgrow corals and/or by reducing herbivore pressure algae will be able to grow “unchecked”. While there is some evidence that phosphorous may decrease rates of calcification in corals most of the literature suggests that the largest effect of increased nutrients on coral reefs is by the response of the algal community. In some cases herbivores be abundant enough to make up for any differences in algal growth caused by enhanced nutrient supply. Phase shifts from coral to algal dominance are typically believed to be the result of both increased nutrients and reduced grazing pressure as a result of overfishing or disease.

Coral reefs are particularly susceptible to sewage pollution because of the delicate ecological balance maintained among a large number of species. The natural low levels of nutrients in tropical seawater are partly responsible for maintaining that balance. Sewage pollution disturbs that balance by nutrient enrichment, which will favour certain species, usually at the expense of reef corals, and will lead to alteration of community structure (e.g. Marszalek, 1987; Grigg and Dollar, 1990; Maragos et al., 1985). Other effects of sewage pollution include toxicity (from toxic materials or toxic by-products from pesticides, herbicides or heavy metals contained in sewage), sedimentation (suspended solids), high biochemical oxygen demand, and hydrogen sulphide generation (Grigg and Dollar, 1990; Pastorok and Bilyard, 1985). However, most of the impacts from sewage pollution on coral reefs reported in the literature relate to the nutrient enrichment rather than to toxic effects. The literature suggests threshold levels for dissolved inorganic nitrogen (DIN) of 1.0 mM and for soluble reactive phosphorus (SRP) of 0.1 mM (see for example Lapointe et al., 1997).

Other impacts of sewage pollution include a decline in growth rate of corals (Tomascik and Sander 1985), reduced calcification (Kinsey and Davies 1979), reduced planulae production (Tomascik and Sander 1987), reduced fertilization success of gametes (Harrison and Ward 2001) and reduced settlement of coral larvae (Tomascik 1991, Ward and Harrison 1996).

In her survey of coral reef degradation in the Caribbean, Rogers (1985) identified sewage as one of the human-related stresses in 9 of the 25 islands or areas for which information was available.

Bell (1991) describes the impact of wastewater discharges from tourist resorts in the Great Barrier Reef, Australia. Two of the most visited islands, Hamilton and Green Island, have discharged virtually untreated sewage in the sea for quite some time. The coral communities at Green Island have been largely replaced by algae and seagrasses. He also recognizes the impact of discharges of secondary treated sewage and sludge from Townsville on the coral reefs of Magnetic Island, just off Townsville. He expects seepage from sewage on Magnetic Island to be disastrous for the already stressed corals. He concludes that tertiary treatment of sewage will be necessary to achieve acceptable levels of nutrients (after dilution) in the discharged effluent. In comparing these findings with the Caribbean, we must realize that mean background phosphate levels for the waters outside the Great Barrier Reef are higher than those reported for the Caribbean.

Van Woesik et al. (1991) examined the response of coral communities to effluent discharge on Hayman Island, Green Island and John Brewer Reef in the Great Barrier Reef Marine Park. Except in the immediate vicinity of the sewage discharge outfall, they found no impact from discharge of secondary treated sewage from the resort on Hayman Island. On Green Island sewage from septic systems is subject to primary treatment before discharge. They attribute the increase in seagrass beds to nutrient enrichment from sewage discharge. At John Brewer Reef Floating Hotel (removed in 1989) treated sewage was transported and discharged 5 km off the reef and the only effluent discharged was brine from the desalination plant. Overall, coral cover increased in the vicinity of the hotel and the authors conclude that there was no detrimental impact of its placement or operations. They suggest that the impact of sewage discharges on coral communities is mostly dependent on the level and quality of treatment.

There are some field studies in which nutrients were experimentally enhanced. In 72 ammonium and phosphate were added to a patch reef at One Tree Island (GBR). Nutrient addition increased primary production (photosynthesis) (Kinsey and Domm 1974) and reduction of calcification . The experiments were repeated in a broad scale set up with multiple patches with different nutrient additions and measurements of a wide range of variable in 1993-1996 in the ENCORE project. Hoegh-Guldberg et al. (1997) studied the impact of nutrient enrichment in the Great Barrier Reef as part of the ENCORE project (Enrichment of Nutrients on Coral Reefs Experiment). Nitrogen and phosphorus (10 mM NH4 and 2 mM PO4 were added to experimental corals on micro-atolls. Although the results of the experiment were not conclusive, it is suggested that increased levels of phosphorus have a negative effect on coral growth. Strong seasonal variation in calcification rates appears to have masked the impact in the experiment. The final conclusions of ENCORE sum up to: increased coral mortality, decreased coral growth, increased calcification but lower skeletal density (weak structure) and reduced settlement of larvae (Koop, et al. 2001).

Simmons and Associates (1994), in their study of the impact of tourism on the marine environment of the Caribbean, note that "….the impact of liquid waste from yachts has been poorly studied in the Caribbean region. While it is very likely to have an effect on water quality in lagoons and semi-enclosed bays, its impact is probably small or negligible in open bays with adequate flushing." Talge (1992) also touches on the issue of nutrient enrichment by diver activities and boat effluents. She raises the question: "…. But are the amounts significant and do they remain over and around the reef long enough to fertilize reef communities?" These questions have remained unanswered to date.

An interesting quote from the abstracts of the 10th International Coral Reef Symposium in Bali of which the proceedings will hopefully come out shortly: Bucher, D.J. "Ammonium reduced the ability of corals to repair lesions, a result which has implications for the recovery of polluted reefs following physical damage."

The hampering of natural restoration through reproduction is of utmost importance: "The point is, while levels of stress may be sub-lethal to adult coral colonies, they may be sufficient to cause reproductive and recruitment failure on nearby and distant reefs (Richmond 1993). Reefs may still hang in there, but where is a future without new generations?

Nutrients can either lead to reduced coral cover by direct harmful effects or by stimulating algae which then outgrow and out-compete corals. Well known cases of coral-algal phase shifts are Kaneohe Bay in Hawaii (Smith, et al. 1981) and Jamaica (Hughes 1994). For simplicities sake the mechanism of influence is by-passed and a direct relation between nutrients and coral cover is sought, because the problem with algal overgrowth is hugely complex, uncertain ("Competitive outcomes did not support the argument that algae are more successful competitors in more eutrophic conditions" (McCook 2001)) and difficult to quantify (McCook 1999, McCook 2001, McCook, et al. 2001). Field data have been chosen from Barbados (Tomascik and Sander 1985, Tomascik and Sander 1987, Wittenberg and Hunte 1992), Brazil (Costa, et al. 2000), Curaçao (Gast 1992, Gast 1998, Gast, et al. 1998, Gast, et al. 1999), Kaneohe Bay (Smith, et al. 1981, Hunter and Evans 1995) and Reunion (Naim 1993), which are used to distil relationships between nutrients and coral cover and the number of coral species. Figure A.1 includes the data of all these studies and shows that there is a general trend of higher concentrations causing lower cover. Extremely high concentrations are always accompanied by cover close to zero. A few corals always appear to be able to survive with high nutrient concentrations, but one can hardly call these a reef. Cut-off concentrations have therefore been chosen and higher values have been removed from the data set. The relationships of coral cover with phosphate is very week. Apparently, forms of nitrogen have a more linear effect on coral cover (Figure A.2). The final choice is for DIN rather than either nitrate or ammonium, because:

- Ammonium is rapidly converted via nitrite to nitrate by nitrifying bacteria in the reef water column and sediments.
- The conversion of urea to ammonium to nitrite to nitrate also takes place in the sewage system. As these processes are oxygen dependent, mainly ammonium enhanced with direct discharge of untreated sewage, but an unknown mixture of ammonium and nitrate is brought into the water column with the outflow of a sewage treatment plant.
- Groundwater seepage always and mainly leads to enhanced nitrate concentrations, because the long residence times give ample opportunity for the conversion.

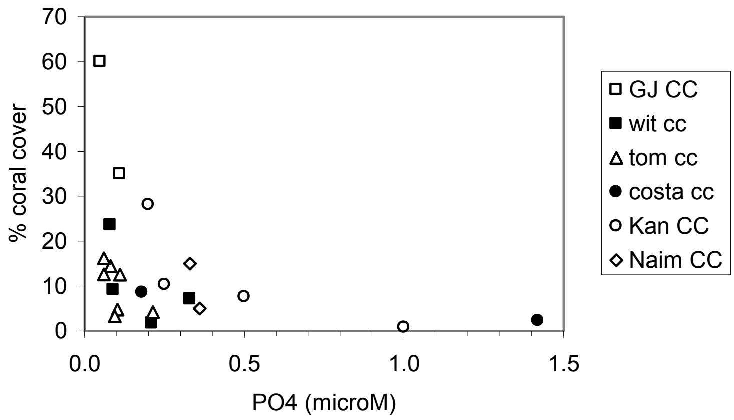

Figure A.1 Decreasing coral cover with increasing phosphate concentration. 'GJ' is data Curaçao, 'wit' and 'tom' are data Barbados, 'costa' is data Brazil, 'Kan' is Kaneohe Bay, 'Naim' is data Reunion. References of these studies are mentioned in the above text. Phosphate is an example; forms of nitrogen show the same pattern.

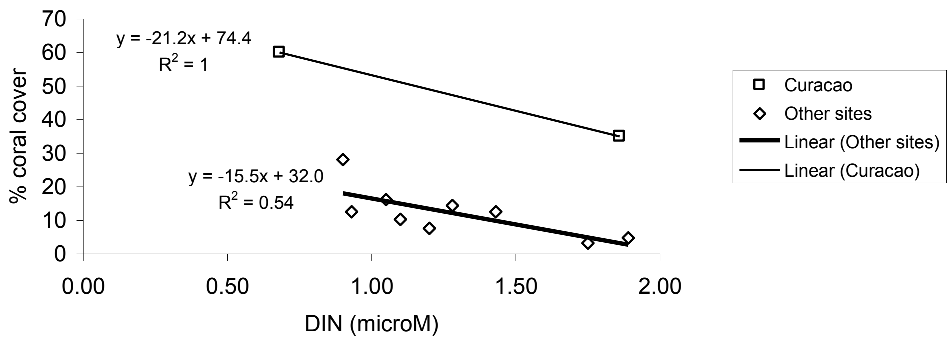

Figure A.2 Decreasing coral cover with increasing DIN values (NH4 + NO2 + NO3). Other sites is collection of refs in legend Fig 1, excluding values > 2. For coral cover the data from Curaçao have been separated from the rest of the data set, because the show a far higher coral cover with comparable nutrient concentrations (the cause could be different local circumstances or a difference in methods with which the data were obtained).

Hence the use of DIN includes bacterial conversion effects and covers both sewage discharge and groundwater seepage. As the slopes differ with a stronger effect of ammonium, one of the other figures could be applied accordingly in areas where the main eutrophication cause is known. The number of scleractinian corals also decreases with eutrophication. DIN is chosen again, because it shows the strongest relation (Fig 3) and to be consistent with the coral cover equation.

Equations:
- % coral cover = 32 – 15*DIN (mM)
- % coral cover = 31 – 30*NH4 (mM)
- % coral cover = 19 – 13*NO3 (mM)
- # coral species = 33 – 11*DIN (mM)
- # coral species = 31 – 22*NH4 (mM)
- # coral species = 33 – 22*NO3 (mM)

Remarks:
- Regressions of # coral species Barbados and Curaçao data
- For coral cover the data from Curaçao have been separated from the rest of the data set, because the show a far higher coral cover with comparable nutrient concentrations (the cause could be different local circumstances or a difference in methods with which the data were obtained).
- Although it is ridiculous to draw a line through two points, this has only been done to show that the slopes of the lines are comparable and that the patterns at Curaçao are comparable to those of all other sites combined regardless of the higher cover.
- Correlation coefficients are included in the excel sheet. However, as the measuring error of nutrients is negligible compared to the uncertainties in establishing coral cover and because there is an a priory accepted effect of the X variable on the Y variable, regression is assumed to be justified.

The final choice in for modelling nutrient pollution is to concentrate on DIN rather than either nitrate or ammonium, because: (1) Ammonium is rapidly converted via nitrite to nitrate by nitrifying bacteria in the reef water column and sediments; (2) The conversion of urea to ammonium to nitrite to nitrate also takes place in the sewage system. As these processes are oxygen dependent, mainly ammonium enhanced with direct discharge of untreated sewage, but an unknown mixture of ammonium and nitrate is brought into the water column with the outflow of a sewage treatment plant; and (3) Groundwater seepage always and mainly leads to enhanced nitrate concentrations, because the long residence times give ample opportunity for the conversion.

Hence the use of DIN includes bacterial conversion effects and covers both sewage discharge and groundwater seepage. As the slopes differ with a stronger effect of ammonium, one of the other figures could be applied accordingly in areas where the main eutrophication cause is known.

The number of scleractinian corals also decreases with eutrophication. DIN is chosen again, because it shows the strongest relation (Figure A.3) and to be consistent with the coral cover equation.

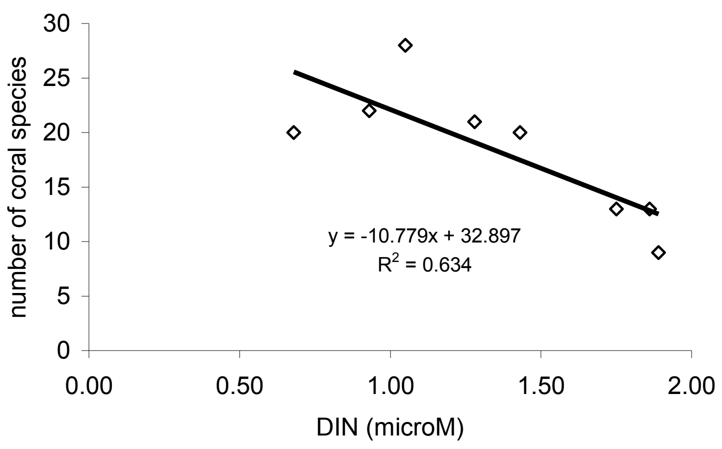

Figure A.3 Decreasing number of stony coral species with increasing DIN concentration. Data from Curaçao and Barbados (see earlier references).

Annex B Sedimentation

Sedimentation resulting from anthropogenic influence occurs almost always concomitant with eutrophication. Major reviews of anthropogenic disturbances on reefs (Pastorok and Bilyard 1985, Grigg and Dollar 1990, Richmond 1993, Dubinsky and Stambler 1996) both address sedimentation and eutrophication. In practice, both are usually the result of urbanization, coastal development and changes in land use (e.g. deforestation), which increase run-off and/or sewage discharge. It is often difficult to seperate the individual effects of the 2 influences (Walker and Ormond 1982, Tomascik and Sander 1985, Tomascik and Sander 1987, Tomascik and Sander 1987, Tomascik 1991, Bak and Nieuwland 1995, Connel 1996). However, it is clear that sedimentation alone is an important threat to the health of coral reefs (see also quote RS Young below).

Grigg and Dollar (1990), in their review of natural and anthropogenic disturbances on coral reefs, state: “The impact of increased sedimentation is probably the most common and serious anthropogenic influence on coral reefs.” Increased sedimentation results primarily from dredging and runoff. Dredging, runoff or siltation were mentioned as one of the human-related stresses on coral reefs in Barbados, Bermuda, Bonaire, Costa Rica, Curaçao, Dominican Republic, Florida Keys, Grenada, Guadeloupe, Jamaica, Panama, St. Lucia, British Virgin Islands and US Virgin Islands (Rogers, 1985). Rogers (1990) associates dredging in the Caribbean with construction of hotels, condominiums, runways, roads, harbors, navigation channels, military installations, and beach replenishment. She states: “Unprecedented development along tropical shorelines is causing severe degradation of coral reefs primarily from increases in sedimentation.”

Background levels of sedimentation on reefs that are not influenced by human activities are between 1 and 10 mg per cm2 per day (Rogers, 1990). She suggests that chronic sedimentation rates above 10 mg per cm2 per day are “high”. The relevance of this threshold is shown in Kenya, where sedimentation values of 1.35 and 4.25 mg/cm2/d did not lead to differences in coral cover (McClanahan and Obura 1997). Sudden exposure to heavy sedimentation may result in burying of corals, expulsion of the symbiotic algae from the coral polyps (“bleaching”), and subsequent death. Other effects of increased sedimentation (varying from 200 to 800 mg per cm2) include: no effect, reduced growth, reduced calcification (33%), decrease in net production, and increase in respiration. Coral species respond differently to heavy sedimentation and some are more efficient in rejecting sediment than others. Bak and Elgershuizen (1976) found that *Acropora palmata*, *A. cervicornis*, *Porites astreoides* and *Agaricia agaricites* were the least efficient and *Colpophyllia natans, Diploria strigosa* and *Madracis mirabilis* were among the most efficient. Rogers (1990) summarizes the results of field and laboratory studies as follows:

- Different species have different capabilities of removing sediment or surviving at lower light levels.
- The coral’s ability to remove sediment depends on the amount and type of sediment, which covers the coral colony.
- Sediment rejection is a function of morphology, orientation and behavior of a coral colony.
- Experimental corals may not behave normally in the laboratory.

Chronic exposure to higher concentrations of sediment can have a variety of negative impacts on corals, many of which can be attributed to reduced light levels. These include (Rogers, 1990):

- Lower species diversity and absence of certain species.
- Less cover by live coral.
- Lower coral growth rates.
- Greater abundance of branching forms.
- Reduced coral recruitment.

- Decreased calcification.
- Decreased net productivity of corals.
- Slower rates of reef accretion.

Marszalek (1981) monitored the impact of a large-scale dredging operation for beach replenishment in Miami, Florida. He distinguished three types of impact: mechanical damage, sediment loading and increased turbidity. A substantial percentage of coral colonies showed signs of stress such as partial bleaching, polyp swelling and excessive mucus secretion. He suggests that sustained increased turbidity was more detrimental than short-term sediment loading.

During a 9 months dredging operation in 1987 in Thailand reefs subject to the resulting sedimentation showed significant decreases of coral cover and coral diversity (Brown 1990). However, these reefs were recovered in 1988 to pre-dredging levels.

Work by Hubbard et al. (1987) in St. John, US Virgin Islands, demonstrates a gradual decrease in growth by 10-20% over the past 100 to 200 years. Their data suggest that this decline may be due to increased sedimentation following the cultivation period of the island.

Van't Hof (1983) describes the effects of dredging and excavation (to construct a canal system and waterfront home sites in a limestone cliff) on the fringing reef in Bonaire. Dredging resulted in sediment loading of almost 100 times the background level and a decrease in percent live coral cover adjacent to the dredge site. The largest change in percent live coral cover was observed on the lower reef slope (35m depth, dominated by Agaricia lamarcki) from 73% cover pre-dredging to 32% post-dredging.

Hodgson (1990) determined that sedimentation inhibited settlement of coral larvae on artificial substrate in a common Indo-Pacific coral species, thus potentially affecting coral recruitment under natural circumstances.

Extensive studies on effects of sedimentation in combination with eutrophication are those done at Barbados by Tomascik & Sander. They measured both suspended particulate matter (SPM), which is an indication of particles in the water column, and total downward flux of SPM (DF-SPM), which shows how many of those particles sink down on corals on the bottom. Comparisons of the sampling stations show that these two variables do not always follow the same trend: a large amount of particles in the water column does not necessarily lead to high downward sedimentation. The significance of difference between the 2 variables lies in the mechanism of damage: the former reduces light available to corals and thus photosynthesis, while the latter leads to smothering, which corals have to remove. The end result is in so far the same that the coral has less energy available and is weakened. Rather surprisingly, there was a significant negative relation between SPM and coral cover, but not between DF-SPM and coral cover. However, this last correlation did become significant when only the summer data were used. Apparently, the physical circumstances (current, waves) determine whether sediment rains down or stays suspended in the water column. The studies mentioned above show reductions of coral growth rates, cover, coral species, diversity, numbers of larvae, settlement, and changes in coral community composition.

Another Caribbean island where extensive studies have been done on anthropogenic effects is Curaçao. Aspects of corals and coral reef ecology have been compared between influenced sites directly in front of Willemstad and non-affected sites upcurrent. Increased sedimentation has been measured by Meesters (Meesters, et al. 1992). This sedimentation caused a reduced capacity to heal artificial lesions (Gast 1992, Meesters, et al. 1992). Comparison of the coral reef community (chain line transect method, the coral species or substrate type under each chain was scored) at the impacted and control sites showed a reduction of coral cover from circa 60% to 35%, a reduction of the number of coral species from 20 to 13 (9 m depth) and reduction of the coral diversity index (Shannon-Weaver) from 2.33 to 1.65 (Gast 1992). The whole coral community had been changed quite drastically. The control reefs showed a high variety in species and growth forms of virtually only stony corals. But reefs in front of Willemstad are dominated by head corals (Siderastrea siderea, Diploria strigosa, Colpophylia natans, Montastreae annularis, Montastreae faveolata, Montastreae cavernose, Dichocoenia stokesii). Soft corals and sponges were rare at the control site, but far more abundant at the impacted site. Branching and leaf-shaped corals (Acropora palmata, Agaricia agaricites, Porites porites, Millepora complanata, Eusmilia fastigiata) were absent or strongly reduced (Gast 1992).

Some interesting quotes from the abstracts of the 10th International Coral Reef Symposium in Bali of which the proceedings will hopefully come out shortly:

- Schelten, C.K. “The number of juveniles was significantly lover on the sediment gradients compared to the control. This was linked to an altered juvenile coral composition, probably towards more sediment tolerant species…. the percentage of damaged juvenile increased while the average size decreased with sedimentation”.
- Ortiz, J.C. Studied coral reef in Venezuela before and after landslides resulting from intense rain. “Coral cover decreased 42% and the number of species from 22 to 17”.
- Young, R.S. Monitoring of reefs in Honduras (Roatan) in the vicinity of broadscale tourism industry development. “Initial results suggest that reef mortality is more closely tied to increases in sedimentation rather than degradation of water quality.”

Dose response relations of the effects of sedimentation are scarce to absent in the literature. A mathematical relationship is presented in Fig. 4 of the review by Pastorok and Biljard (Pastorok and Bilyard 1985). These response curves were calculated with the original data collected at Guam (Randall and Birkeland 1978). Data were collected from upper slope communities and the lines were fitted using least-squares linear regression. The resulting equations are:

- $\ln Y = 4.97 - 0.018X$ for the effect of sedimentation on the number of coral species
- $\ln Y = 3.17 - 0.013X$ for the effect of sedimentation on percent coral cover

X in sedimentation rate (mg.cm-2.day-1). Y is coral species (number) or coral cover (%), respectively. A problem arises with the maximum number applying this equation to other regions. The value of e4.97 is 144 species of corals, which is not realistic for Hawaii or the Caribbean. The number of stony coral species is 55 in the Hawaiian Archipelago (Wilkinson 2000) and in the Caribbean there are roughly 70 species. Hence values of 4.01 and 4.25 should be used when the model is applied to Hawaii and the Caribbean, respectively.

Finally, reef degradation is also partly responsible for a decline of reef fisheries. Sedimentation can kill major reef-building corals, leading to the eventual collapse of the reef framework. The reduction in the percentage of living coral as well as the decrease in the amount of shelter that the reef provides leads to a decline in the number of reef fish and the number of species (Rogers, 1990).

Annex C Physical anthropogenic damage

Several authors have researched and documented snorkeler and diver damage. It is often difficult to distinguish such damage from natural damage or other forms of human-induced damage (for example, see Rogers et al., 1988). This points to the need to design research methods that will target a specific issue and eliminate compounding factors. Hawkins and Roberts (1993b) and Scura and Van't Hof (1993) have applied such methods by comparing heavily dived and little dived areas and by looking at gradients of impact along a line of decreasing recreational activity.

Rogers et al. (1988a, 1988b) monitored coral breakage at two reefs in the Virgin Islands National Park and Biosphere Reserve (VINP), as well as individual Elkhorn coral colonies (*Acropora palmata*). They attributed broken coral branches to careless snorkelers, boat strikes and swells. They noticed divers and snorkelers bumping into corals or standing on them, and overturning corals to reach lobster. Even in the absence of major storms or other stresses, only 10 of 50 tagged Elkhorn coral colonies remained undisturbed over a 7-month period of observation. Rogers et al. (1988a) report that at Trunk Bay in VINP, where a snorkeling trail was established in the early sixties –receiving 170,000 visitors annually by 1986- ".... the trail has deteriorated substantially as a result of people standing on corals, breaking coral branches while snorkeling, and removing organisms as souvenirs."

Tilmant (1987) and Tilmant and Schmahl (1981) studied the impact of recreational activities on buoyed reefs in Biscayne National Park, Florida. Each buoyed reef received three or more times as much use as its control. The most frequent recreational activities were snorkeling and spear fishing. The mean frequency of damaged coral encounters ranged from 35 to 140 per 30-minute count. Although significant differences in damage between buoyed reefs and controls did occur at some sample points, such differences did not follow a consistent pattern that could be readily attributed to human use. Incidence of damage to soft corals was much higher than that to hard corals. They recognized that, since the level of recreational use on the reefs studied was relatively low (no more than 1,500 people per reef per year), the impacts may be more severe at higher levels of use.

Talge (1991) studied the behavior of snorkelers and SCUBA divers in the Looe Key National Marine Sanctuary, Florida, in terms of the number of interactions between divers and coral. The interactions she observed were:

- Hand on the coral to steady or help gain control
- Kicking or brushing with the fins
- Standing on corals
- Grabbing corals (especially soft corals) to pull themselves through the water
- Rubbing against coral with any part of the body
- Sitting coral with the SCUBA tank or other pieces of equipment
- Creating sediment clouds

The most frequent interactions were "finning" and "push-off". The average number of interactions per diver is ten per dive. Snorkelers had significantly less interactions than SCUBA divers, divers without gloves had fewer interactions than divers with gloves, and females had fewer interactions than males. Over two-thirds of the interactions were with hard corals. This contrasts with the findings of Tilmant and Schmahl (1981) and may well be due to the selection of the study sites, one being comparatively richer in soft corals than the other. Coral breakage included only 0.6% of all incidents. This author also expressed concern with increase in nitrogen concentration of the water by divers urinating over the reef. This concern has not been substantiated by further research.

An experimental study by Talge (1992) in the Looe Key National Marine Sanctuary, Florida, consisted of weekly "touching" and "finning" selected corals at two intensities. Weekly touching had no detectable lasting influence on the health of 11 species of corals, either visibly or histologically. Based on an average of 10 interactions per diver, she calculated that 4-6% of the live coral area is touched weekly. However, as a small percentage of divers have much more frequent interactions, she recommended that the touching ban in the Sanctuary be maintained.

Scura and Van't Hof (1993) and Dixon et al. (1993) describe diver impact in Bonaire, Netherlands Antilles. A comparison of sites receiving high levels of use, intermediate levels of use and controls (reserve sites closed to diving) indicated that percent hard coral cover is significantly lower at high-use sites than at control sites, while species diversity is higher at high-use sites than at controls. At intermediate-use sites no such differences were found. The study also suggests that impact is decreasing with linear distance from the center of activity (in the case of Bonaire the dive boat moorings). Percent coral cover and species diversity increase with distance from the mooring. The findings led to the postulation of a "threshold" hypothesis that diver impact becomes quickly apparent when use exceeds a level of 4,000-6,000 divers on a dive site per year.

Hawkins et al. (1999) repeated the study of Scura and Van't Hof (1993) three years later. They found a decrease in coral cover at both control and dived sites, except at one of the sites labeled as high-use in the 1993 study. Their study showed that dive sites suffered no greater loss of coral cover than control sites in the three-year period. However they found a distinct difference in community structure between high-use and control sites. The proportion of massive corals that make up total coral cover decreased at both high-use and control sites, but the decrease was much greater at high-use sites (19.2% vs. 6.7% decrease). The proportion of branching coral increased 8.2% in high-use sites compared to 2.2% in control sites, with coral diversity and species richness showing a similar pattern. They conclude that there has been an increased disturbance of Bonaire's reefs over the three-year period between the studies, with greater disturbance in high-use areas than at control sites. A reduction in cover by massive species is also reported in Connell (1997) for Buck Island, St. Croix, US Virgin Islands. Although no explanation is offered for the decline, it is noteworthy that Buck Island is a location, which is subjected to heavy recreational use.

Tratalos and Austin (2001) conducted a study, similar to that of Scura and Van't Hof (1993) and Hawkins et al. (1999) in Bonaire, in Grand Cayman. They found significantly lower overall hard coral cover and massive hard coral cover at high intensity sites compared with low intensity sites and undived sites. There was also more dead coral and coral rubble at high intensity sites. A "distance effect" was also present, meaning that hard coral cover increased with distance away from the mooring buoy. Most of their results are similar to the findings of the Bonaire studies.

Several studies on diver and snorkeler damage have been conducted in other parts of the world. Although Indo-Pacific reef structure and species composition are different from that of most Caribbean islands (the former possessing extensive reef flats and more branching and foliaceous species), the results of these studies are nevertheless of value.

Allison (1996) researched snorkeler damage to reefs in the Maldive Islands. The study showed a positive correlation between the distribution of broken corals and snorkeling activity on the reef at Vihamanaafushi. The study concludes that: "....the observed breakage is important because of potential reduction of the aesthetic appeal of the reefs to tourists, and degradation of the reefs' ability to sustain the islands they protect and nourish." The author advocates, amongst others, programs to educate and train users to reduce damage, and to develop information packages and simple effective data collection methods suitable for amateurs. Networks of dive and tour operators could be used as the implementation vehicle for such programs.

Hawkins and Roberts (1993b) studied the effect of trampling by SCUBA divers and snorkelers on reefs flats of coral reefs in Egypt. They found significantly more damaged corals and loose fragments of live coral in heavily trampled areas than in little-trampled areas. Percentage of bare rock and rubble was significantly higher, while percentage of live coral cover and number of hard coral colonies were lower. Coral colonies were also smaller in trampled areas compared with control areas. In summary, heavy trampling by divers appears to alter the coral population structure of the reef flat. Although the Western Atlantic reefs do not have extensive reef flats as occur in Indo-Pacific reefs, some Caribbean islands have experienced or continue to experience the effect of divers and snorkelers treading on coral. Buccoo Reef in Tobago is probably the most infamous example of the destruction caused by reef walking (see Rogers et al., 1988b). Where shore diving is practiced, divers and snorkelers will also have some impact on shallow reef areas by trampling.

Hawkins and Roberts (1992a, 1992b, 1993a, 1993b, 1994) compared heavily dived and un-dived areas in Egypt and found significant differences in levels of damage. Numbers of broken hard coral colonies, live loose coral fragments, reattached fragments, abraded colonies, and part-dead colonies were higher in dived areas. They concluded that divers cause significant damage to benthic communities on the fore-reef slope. Their findings suggest that damage accumulates rapidly when a new site is opened up for diving, with impact stabilizing after a certain level of use had been reached. The three study sites received between 5,000 and 13,000 dives per year. Hawkins and Roberts (1994) suggest that dive sites at Sharm-el-Sheik in Egypt can accommodate 10,000 to 15,000 dives per year without serious degradation.

Epstein et al. (1999) compared populations of the hard coral Stylophora pistillata at a site that had been closed to the public for six years with two nearby sites, open to the public, in Eilat, Northern Red Sea. The main results of the study indicate that: (1) live coral cover was three times lower at the open sites than at the closed site; (2) there were significantly more small colonies (recruits) at the open sites and significantly less large-size colonies; (3) the average number of broken colonies was three times higher at the open sites. They interpret the lower breakage level in the closed site as a sign of the effectiveness of the closure, but they also conclude that a no-use policy is not sufficient for protecting small reef areas.

Jameson et al. (1999) developed a Coral Damage Index (CDI) to assess the extent and severity of physical damage to coral. Sites are characterized as "hot spots" if in any transect the percent of broken coral colonies is 4% or more, or if the percent cover by coral rubble is 3% or more. In a study of four diving sites off Hurghada and Safaga, Egypt, in the Red Sea, 40% of the transects surveyed qualified as "hot spots". The relatively large number of hot spots in shallow water suggests that most of the damage was caused by anchors dragging across the reef. They conclude that the diving carrying capacity of the sites is being exceeded by large amounts.

Muthiga and McClanahan (1997) compared the impact of visitor use (diving and snorkeling) in heavily used sites and less frequented sites. They found no significant differences in coral cover or bare rock and rubble between sites, nor differences in coral species composition and diversity. However, there was significantly more damage to coral in the high-use sites, as evidenced by the number of broken, abraded, and broken and reattached coral colonies. Greater damage as observed in shallow than deep areas, which may indicate that snorkelers have more impact than SCUBA divers. Differences between the results of this study and those in the Red Sea may be explained to a large extent by the much higher visitation levels in the Red Sea.

Davis at al. (1995) observed diver interactions in the Julian Rocks Aquatic Reserve in Eastern Australia. Thirty divers were observed for about 30 minutes each. The number of diver contacts ranged from 2 to 121 (average 35 contacts per dive). More than 50% were contacts made with fins. Only 7.2% of contacts resulted in noticeable level of damage. The majority of damaging contacts were with hard corals, with lesser damage inflicted on sponges and turf algae. More experienced divers (those with more than 100 logged dives) made significantly less uncontrolled contacts than less experienced divers.

Harriott et al. (1997) conducted a similar study of diver contacts at four other locations in Eastern Australia (Heron Island and Lady Elliott Island in the Southern Great Barrier Reef, and Gneering Shoals and Solitary Islands in sub-tropical Eastern Australia). There was a large range in the total number of contacts per diver per site, with a few divers having a disproportionate impact. This coincides with the findings of Rouphael (1995). The maximum number of contacts ranged from 192 at Gneering Shoals to 304 at Solitary Islands. There was a significant difference between the mean numbers of contacts between sites, ranging from 31.3 at Heron Island to 121.2 at Solitary Islands. The mean number of coral contacts follows more or less the same pattern, but there is no significant difference between sites in coral breakage. The mean number of corals broken per dive ranged from 0.6 at Heron Island to 1.9 at Solitary Islands. Most contacts were made by fins and 78% of coral breakage was caused by fins. Differences between the number of contacts and coral breakage per site were attributed to:

- Greater awareness among divers at Heron Island and Lady Elliott Island (located within the Great Barrier Reef Marine Park), because of the awareness campaigns by the Great Barrier Reef Marine Park Authority and pre-dive briefings at these sites.
- At Solitary Islands and Gneering Shoals, divers actively explored the small invertebrate fauna and thereby spent more time close to the bottom where they were more likely to make contact with corals.

Apart from the direct physical impact from diver and snorkeler contacts, such as breakage of coral and inflicting lesions, there is also some evidence that damaged corals show reduced growth rates (e.g. Liddle and Kay, 1987, and Meesters et al., 1994).

Table C.1 Mean values of coral cover, at near mid and far distances from the mooring buoys, at sites with high and low diver numbers

	Near	Mid	Far
		High diver numbers	
% hard coral	18.04	19.48	24.69
% massive coral	11.42	11.69	14.07
% dead coral	3.48	3.35	4.12
% coral rubble	1.39	3.15	0.85
		Low diver numbers	
% hard coral	28.9	33.93	30.97
% massive coral	19.44	24.71	23.88
% dead coral	1.64	2.07	1.67
% coral rubble	1.14	0.21	0.76
		Impact of less pressure	
Gain hard coral	60%	74%	25%
Gain massive coral	70%	111%	70%
Less dead coral	-53%	-38%	-59%
Less coral rubble	-18%	-93%	-11%

Source: Tralalos and Austin, 2001, p.72.

Annex D Overfishing

This Appendix describes the economic and ecological consequences of overfishing and looks briefly at marine reserves as a way of coping with fishing pressure. Overfishing is different from the other threats described here, in that it does not have the same type of direct destructive impacts. Modest forms of non-destructive overfishing will, in fact, have very little impact on corals. Extreme forms of non-destructive overfishing could, on the other hand, alter the ecosystem balance, ultimately leading to a reef dominance by sea urchins or macro-algae and resulting in a dramatic drop in fishery yields and reduced coral biomass and productivity (McClanahan, 1995).

An interesting example of overfishing of invertebrates is given by a case of mother-of-pearls (Trochus spp.) in a village in Central Maluku (Indonesia) (Figure D.4). A traditional management scheme, referred to as *sasi*, was in place with three year harvesting cycles. At some stage, annual harvesting was allowed, leading to severe overfishing of the resource. In the time of the 3-year closed season, the average yield was around 3400 kg (over 1100 kg per year). In the time of annual collection since 1987, the average annual yield of just over 400 kg.

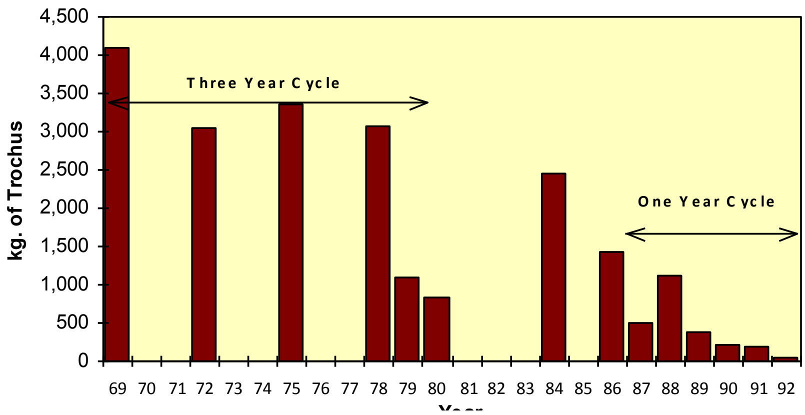

Figure D.4 Yield of Trochus Shells in Noloth (Central Maluku, Indonesia) in 1969-1992 (source: data gathered from village head at Noloth, Maluku, as presented in Cesar et al. 1997).

Costs and benefits of overfishing in Indonesia are presented in Cesar (1996). He compares fisheries benefits in case of open access (OA) versus maximum sustainable yield (MSY) levels. He used data from Southeast Asia and the Pacific partly based on Munro & Williams (1985) and Alcala & Ross (1990). The latter describes an interesting case where protective fishery management was discontinued and where fishery yields dropped very quickly after resource access was re-opened. This is the same experience as in Hawaii where fisheries were re-opened after World War II with trophy catches during the first year. A comparison of fish standing stock in the Main Hawaiian Islands (heavily fished) versus the Northwest Hawaiian Islands (no fishing) also shows a dramatic difference (Friedlander & DeMartini, 2002). Cesar (1996) comes with an estimate in terms of net present benefit (10% discount rate; 25 year time horizon) of US$ 109,900 of the MSY and US$ 38.5 thousand in the case of OA for Indonesia.

Marine reserves: There is a growing body of literature suggesting that the establishment of marine reserves can be a sound management option in the light of overfishing. See for instance, Alcala (1988), White (1989), Alcala & Russ (1990), Polunin & Roberts (1993), Roberts (1995), Russ (1989, 1994). For a recent overview, see Rodwell & Roberts (2000). For instance, Alcala (1988) presents estimates of the island of Sumilon, where fish yields of 14-24 mt/km2/year have been reported before the sanctuary and where these catches increased to 36 mt/km2/year when the marine reserve was in place. Fish yields fell back to about 20 mt/km2/year when island management broke down. Most studies are unambiguous about the positive impact of marine reserves on reef fish abundance and size.

The “spillover” effect (increase in biomass outside the closed area) of closed areas is, however, complex and not yet well understood. It includes movements of both adults, juveniles and larvae, and the transfer mechanisms may be density-dependent (an effect from the closed area) or uni-directional (transport of larvae) (Rodwell & Roberts, 2000). The examples that potentially demonstrate the spillover effect are those of Apo Island in the Philippines and Kisite Marine National park in Kenya. Data from the Soufriere Marine Management Area in St. Lucia and Merritt Island National Wildlife Refuge in Florida confirm the predicted role of marine reserves in supporting fisheries outside the reserve areas (Roberts et al., 2001).

The most effective size, location and design of reserves have been and remain the subject of much discussion and research. Recent studies indicate that marine reserves, in order to be effective fisheries management and biodiversity conservation tools, will need to be established in large-scale networks covering significant portions of marine ecosystems (at least 10-20%) (Ballantine, 1995; Allison et al., 1998; Roberts et al., in press a, b; cited in Rodwell and Roberts, 2000). Roberts and Hawkins (in press; cited in Davis, 2000) detail the scientific arguments for setting aside at least 20% of the ocean as no-take zone. They write: “The main reasons for conservationists and scientists backing a target of 20% closure are: (1) this figure can be justified on the basis of the best biological information currently available; (2) such closures are expected to provide significant economic benefits to fisheries; (3) it is a realistic figure to implement. However, we shouldn’t look upon 20% as a fixed goal, but rather as an average, with some areas and habitats needing less protection and others needing more.” This is the same argument recently made for Hawaii (Birkeland & Friedlander, 2002).