# The Economic Benefits of Ecosystem-Based Marine Recreation: Implications for Management and Policy

by

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## Abstract

Even as global fisheries are in decline, participation in ecosystem-based marine recreational activities (MRAs), defined here as recreational fishing, whale watching and diving, has increased around the world, adding a new dimension to human use of the marine ecosystem and another good reason to strengthen marine ecosystem management measures worldwide. After compiling available data for maritime countries, a meta-analysis was used to estimate the yearly global benefits of the largest MRAs. Results suggest that 121 million people a year participate in MRAs, generating 47 billion USD in expenditures and supporting one million jobs. Aside from offering the first global estimation of socioeconomic benefits from MRAs, this work provides insights on their drivers of participation and possible ecological impacts. In the case of whale watching, potential benefits are estimated for maritime countries that do not currently engage in this industry based on ecological and socio-economic criteria. Results suggest that whale watching could generate an additional 413 million USD in yearly revenue, supporting 5,700 additional jobs; this would bring the total potential benefits from the global whale watching industry to over 2.5 billion USD in yearly revenue, supporting 19,000 jobs. Recreational fishing is the largest MRA in the world, and can be a vital component of regional economies. Using available fisheries and ecosystem data, an Ecopath model was used to explore the ecological and economic effects of specific fisheries management measures in Baja California Sur, Mexico, particularly regarding longlining effort reductions and billfish bycatch. Results suggest that currently mandated policies will have little effect on marlin abundance in the area. The effects of ecosystem dynamics in an already overfished system must not be overlooked, as they can negate or even reverse desired outcomes from management. All results are discussed from an economic and conservation policy perspective, with emphasis on potential benefits and limitations.

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*“Todo lo puedo en Cristo que me fortalece.”* Filipenses 4:13

## Co-Authorship Statement

Several co-authors have contributed to the preparation of the chapters included in this thesis work. However, I am the first author of all the chapters in the thesis, and the main driving force behind the research. Chapter 2 is in press with the *Journal of Bioeconomics* with U. Rashid Sumaila, my supervisor, as a co-author. Chapter 3 is in press with *Marine Policy*, with U. Rashid Sumaila, Kristin Kaschner and Daniel Pauly as co-authors. A preliminary version of Chapter 4 was presented at the 2009 Ecopath 25 Year Conference and published in the conference proceedings with Villy Christensen, Francisco Arreguín-Sánchez and U. Rashid Sumaila as co-authors.

**Chapter 1**

# 1 Introduction

## 1.1 Problem statement

Since the end of the last century, the study of natural resource management has turned its focus, at first gradually and now ever more strongly, from the development of extractive industries to resource conservation. In this thesis I focus on marine resource management, although the previous statement could be applied in other fields. While conservation is now a buzzword in practically any field, the shift in fisheries science perhaps stemmed from the devastating failure of management to prevent overfishing, leading to the significant decline of fish stocks all around the world, many of which may never fully recover (Clark 2006). There are many reasons for this outcome, and many economic and biological explanations have been proposed. However, the underlying themes behind the science include greed on the part of fishermen, unawareness and arrogance from fisheries scientists, and a plain lack of interest and foresight from politicians and the general public (Walters and Martell 2004); needless to say, these adjectives could be assigned in a number of different ways and still remain valid.

In any case, the overwhelming consensus in academia is that, as we are apparently incapable of adequately exploiting fish populations at an optimal level, the best strategy is one that instead stresses their conservation (Pauly *et al.* 2002). The current discourse on the way forward for marine resource management is to find alternative strategies that move away from the basic input and output controls of classic management and toward 'ecosystem-based management' (Pikitch *et al.* 2004), which recognizes a suite of benefits derived from marine resources, including those from 'non-extractive' activities, such as recreational fishing, whale watching and diving (Leslie and McLeod 2007).

The following work focuses on these activities, referred to here as ecosystem-based marine recreation. They are thus distinguished from other forms of recreation, such as surfing, sailing or swimming, which take place in the ocean but do not necessarily make use of the marine ecosystem. Ecosystem-based recreation has become a global industry, generating a significant amount of revenue and jobs around the world (Pitcher and Hollingworth 2002; Gentner and Steinback 2008; O'Connor *et al.* 2009). Usually flying under the flag of conservation and enjoying considerable political backing, these activities continue to grow, in some cases at a very fast pace and without much thought to any possible negative effects. As will be discussed, the purpose of this work is to establish the potential but also the limits for these activities as alternatives to traditional resource use.

### 1.2 Research objectives

Parts of this research fit within a much larger study, the *Global Ocean Economics* project (GOEP; *www.feru.org/goep*), that works closely together with the *Sea Around Us* project (*www.seaaroundus.org*) at the Fisheries Centre, University of British Columbia. The main objective of the GOEP is to provide valuations of the contributions of ocean fish populations to the global economy. In that context, the contribution of this work is to provide an estimate of the global benefits from ecosystem-based marine recreational activities.

A second goal of this work is to provide an analysis of marine recreational activities, which will be useful for management by explicitly recognizing not only their current and potential benefits but also their possible costs from an ecological and economic standpoint. In the following chapters, I provide an estimate of the global benefits from ecosystem-based marine recreation and identify its potential drawbacks; in the context of current debate surrounding whaling and whale watching, I calculate the potential for further development of the whale watching industry based on current industry performance and biogeographical marine mammal data. Finally, I use an ecosystem model to analyze a specific case of management policy that must deal with overlapping commercial and recreational fisheries.

### 1.3 Thesis outline

This thesis is organized into five chapters. Chapter 1 introduces the reader to the main issues surrounding the valuation of marine recreation, both at local and global scales, and provides some context to the current debate on the issue of 'non-extractive' resource use.

Chapter 2 deals with the global socio-economic benefits (participation, employment and income) generated by ecosystem-based recreational activities, focusing on recreational fishing, whale watching and diving. These benefits are estimated through a meta-analysis of data obtained from secondary sources and the application of a benefit transfer valuation approach. Results are analyzed from a policy context, stressing the importance of treating marine recreation as any other human activity that profits from natural resources. Though these activities may indeed have fewer negative impacts than commercial extraction, undertaking them under the assumption that they will inherently lead to conservation is naïve at best.

In chapter 3, I focus on the potential for whale watching around the world. Many countries have already implemented whale watching operations and derive significant benefits from this industry. However, perhaps due to a lack of tourism infrastructure or a perceived lack of opportunity to enter the market, many countries have not yet invested in this industry. Using current industry statistics, together with a global database of marine mammal abundance and distribution, I estimate the proportion of tourist arrivals that might go whale watching in a given country based on its marine mammal population, and use regional socio-economic data to provide an estimate of the benefits this could generate for that country.

Chapter 4 deals with the problems of resource allocation between recreational and commercial uses in Baja California Sur (BCS), Mexico. Even as commercial fisheries decline in this region, ecosystem-based tourism has become the backbone of the regional economy (Ditton *et al.* 1996), with recreational billfishing leading the charge. As with many other sites, this creates tension between industries which directly or indirectly compete for resources. In the case of BCS, this tension is worsened by the fact that billfish are caught as bycatch in commercial longliners. The effect and magnitude of this bycatch on billfish stocks is largely unknown but is generally defined through public perception. Using an ecosystem model (Ecopath with Ecosim; Christensen and Walters 2004), I explore the effects of specific management policies, in ecological and economic terms. In the last chapter, I summarize my results in a management context and discuss the strengths and weaknesses of my work, as well as potential opportunities to improve or expand on the methods and results.

## 1.4 Background and literature review

### 1.4.1 Economic valuation of marine recreation

A considerable amount of attention has been paid to the valuation of 'non-market' natural resources (Costanza *et al.* 2005), which includes deriving proxies that determine how much economic value an individual places on goods and services for which he or she does not have to pay (e.g., clean air, a forest or a whale's existence), usually through a survey of some kind. Although these seem like intuitive ways to estimate the value of non-market goods and services, there is much concern with the shortcomings of these approaches in a policy context, e.g., extreme overvaluation or response bias dependent on the question framing (Kahneman and Knetsch 2005).

This work does not attempt to estimate the total value placed on natural resources, but instead focuses on the actual amount that users pay to enjoy ecosystem-based marine recreational activities; this is the intended meaning of valuation for the rest of the work. In doing so, I use the classification of direct and indirect expenditures as defined in a large part of the marine recreation literature. Direct expenditures are the payments made on the activity itself, such as ticket prices, tackle or diving gear, whereas indirect expenditures include payments made for supporting goods and services such as travel, accommodation and food costs (Hoyt 2001). These expenditures are revealed values of the activities in question, and although they do not include consumer surplus, they provide estimates of actual benefits which can be captured by local or national economies (whether they be formal or informal) and support real jobs.

In the case of marine recreational fishing, there is a great deal of economic valuation literature in the USA, Australia, Canada and Europe (e.g., Steinback *et al.* 2004; Toivonen *et al.* 2004), which is not surprising given the size of the industry in these countries (generating expenditures totaling over 30 billion USD per year in the USA alone (Gentner and Steinback 2008). A growing body of literature is slowly being produced for other countries that depend on recreational fishing tourism benefits, including some in Central and South America and Southern Africa (e.g., Fedler 2008; Pradervand and van der Elst 2008; Aas 2008).

Although it has been around for decades, whale watching as an industry has, until relatively recently, not been a large area of study, hence there are currently a small number of published economic studies. However, the International Fund for Animal Welfare (IFAW) has funded three key global surveys on whale watching, which proved invaluable to this work (Hoyt 2001; Hoyt and Iñiguez 2008; O'Connor *et al.* 2009). While these studies are unequivocally supportive of environmentalism, their methods are overwhelmingly sound and transparent, and make for very useful data.

The case of diving is interesting in that it is perhaps the most competitive global industry among the three considered in this work. A select few privately-owned companies (e.g., Professional Association of Diving Instructors, Scuba Schools International, National Association of Underwater Instructors) actively compete for a share of the diving certification market, so data on actual revenues (or even the number of current divers) is rare and usually consists of gross figures provided in coral reef valuation studies (e.g., Cesar *et al.* 2003). Estimates for this industry were thus based on these available statistics and a review of retail market prices.

It is clear from the previous paragraph that a global-scale valuation is challenging given the overwhelmingly patchy nature of available data. In order to address this fact, I undertook a meta-analytical approach to the study, compiling data from different sources in order to provide a sample size sufficient for subsequent analyses. This method has been used extensively when dealing with data-poor situations, from fisheries to medicine (e.g., Rosenthal and DiMatteo 2001; Sumaila *et al.* 2007). When the goal is to estimate unknown values, meta-data can be used in a regression or a direct value transfer. In either case, a key assumption is that the site which is used to provide an estimate is sufficiently similar to the one lacking data (Rosenberger and Stanley 2006). In this work, the issue of 'transferability' among sites (in this case, countries) is addressed through the use of independently-defined geographic sub-regions (FAO; *www.fao.org*).

#### 1.4.2 Interaction and conflict between recreational and commercial resource use

An important motivation for this component of my thesis is my discontent with the political undertones which permeate 'science-based' policy around recreational and commercial resource use (Cisneros-Montemayor and Ishimura 2009). In some cases, such as the whaling-whale watching issue, these stem from the fundamentally incompatible viewpoints of different stakeholders (Komatsu and Misaki 2001; Miller and Dolsak 2007). Although any group has the right to voice and try to further their particular stance on an issue, scientific data and analysis should not be constructed based on those opinions.

Recreational and commercial fishing conflicts are a very interesting case in point because despite the fact that recreational use is thought to be inherently less detrimental to marine populations than commercial use, it most definitely can impact them (Coleman *et al.* 2004; Lewin *et al.* 2006). This is often underplayed in discussions regarding the potential for conservation through recreational fishing; at any rate, the arguments for recreational fishing most often stress its superior economic benefits (Southwick Associates Inc. *et al.* 2008). Although the total economic benefits of a recreational fishing industry are often overstated through misleading economic analyses (Edwards 1990), it is obvious that they are nonetheless quite significant. However, to ignore their potentially considerable ecological effects can only lead to problems in the long run much like the ones we have seen in commercial fisheries.

The purpose of stressing potential negative impacts is not to detract from the benefits of ecosystem-based marine recreation, but to present a more balanced discussion of potential costs and benefits. Although it might be impossible to completely detach personal biases from science, trying to do so may help policy-makers learn from the mistakes and lessons of a long history of marine resource management. In turn, this can facilitate the adequate implementation of new activities in parallel with traditional natural resource use, relying on stakeholders to state their objectives and science to shed light on the way forward.

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**Chapter 2**

# 2 A Global Estimate of Benefits from Ecosystem-Based Marine Recreation[1]

## 2.1 Introduction

Although traditionally an afterthought in the management of ocean resources (Walsh 1991), marine recreational activities (MRAs) such as kayaking, recreational fishing, whale watching, surfing and diving have recently come to the forefront of discussion and research regarding their impact and importance in ecological, economic and social terms (e.g. Hoyt 2001; Pitcher 2002; Buckley 2002; Aas 2008).

We focus here on three particular activities: whale watching, diving and recreational fishing. These are classified here as forms of ecosystem-based recreation; the magnitude and nature of their effects on ecosystems are arguable, but they undeniably rely on marine populations being in place and will probably benefit from healthier ones (not unlike marine fisheries). So, while their effects on marine organisms can be less pronounced than commercial exploitation, we make the distinction between these activities and other forms of 'ecotourism', which are defined in a number of ways in the literature, but normally involve seeking to avoid human impacts on nature (Holland et al. 1998). We chose these activities because they rely on active use of marine resource populations, for which benefit calculations have up to now been limited to those stemming from capture fisheries. In this way, we hope to provide elements for a more complete picture of the contribution of marine populations to human welfare.

While it is increasingly clear that recreational fishing mortality can be significant (Coleman et al. 2004) and can further impact marine stocks already depleted by commercial fisheries, it has also been shown that recreational fishing[2] can by itself have

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[2] We will use 'recreational fishing' and 'angling' indiscriminately throughout the text to refer to any form of fishing (e.g., line, spear) where the *main* motivation is not consumption, trade or sale of the catch.

the same effects attributed to commercial fishing, such as overfishing leading to the decline of fish populations, and selective fishing mortality effects on population and ecosystem structure (Post *et al.* 2002; Lewin *et al.* 2006). In this sense, the effects of angling should no longer be considered to be any different from those of commercial fishing. Although we might safely say that whale watching and recreational diving are less harmful than whaling, coral mining or blast fishing, we must also acknowledge the possible negative effects of these activities, such as modified whale behavior or degradation of coral reefs (Davis and Tisdell 1995; Orams 2000; Williams *et al.* 2002). Recognizing these effects allows for a more objective assessment of the potential benefits of MRAs, which can be considerable.

Some countries have recognized the significance of MRAs and have subsequently undertaken large-scale data collection efforts in order to estimate their contributions to society. In the US, expenditures on marine angling, whale watching and diving are estimated to be ~30 billion USD (2003) per year (The Leisure Trends Group 2000; Hoyt 2001; US Department of the Interior *et al.* 2006; Gentner and Steinback 2008). Canada, Australia, and most European countries have also begun to collect data on MRAs (e.g., Henry and Lyle 2003; DFO 2005; Pawson *et al.* 2006). Besides providing a gauge of the magnitude and benefits of these activities, these types of studies can allow managers to integrate them into economic and ecosystem management plans.

Along with other fields, marine science has gradually shifted some of its focus from small scale studies towards global-scale analyses of the dynamics and effects of capture fisheries on marine populations and ecosystems (e.g., Myers and Worm 2003; Pauly *et al.* 2003; World Bank 2008). Although some precision may be lost in this shift, the results of a broader scope can allow for useful generalizations and for the emergence of patterns which often cannot be readily distinguished at smaller scales but which nonetheless affect them (Costanza *et al.* 1997; Rosenthal and DiMatteo 2001). Therefore, the main purpose of this study is to serve as a first attempt at estimating the socioeconomic benefits of ecosystem-based marine recreational activities at a global scale, while recognizing their potential costs to provide a basis for undertaking a full cost-benefit analysis in future research.

## 2.2 Methods

### 2.2.1 Defining the activities

Although there are many forms of marine recreation, such as swimming, fishing, diving, surfing and sailing, to name only a few, the first step in this work is to identify and define the activities to be included in our study, as well as the indicators to be examined for each one. Here, the main indicators for each activity will be the number of users, total expenditure and employment in each sector. The MRAs considered are those which fit our definition of ecosystem-based tourism, i.e. those that directly rely on marine populations. These are:

- Recreational fishing: Defined here as fishing where the *main* motivation is not consumption, trade or sale of the catch;
- Whale watching: Including watching whales as well as other marine organisms (such as sea lions and dolphins) from above water;
- Diving: Including snorkeling and SCUBA; although some diving takes place around underwater wrecks, the assumption is made that observing marine life is one of the main attractions for most divers. Although this may overlap with the definition of whale watching in some occasions, an effort was made to separate between the two to avoid double-counting.

### 2.2.2 Data collection

Data was collected on recreational fishing, whale watching and diving for the 144 maritime countries of the world. Information was obtained through a review of secondary sources, including peer reviewed publications, government and NGO reports and all other available documents. The first step in this was to determine whether MRAs take place in a given country or not; if the answer was yes, this was followed by a search for (preferably country-level) participation, expenditure and employment data. It was assumed that finding no data for a particular country means that MRAs do not take place there or are insignificant. One inherent drawback in meta-analyses is the varying quality of the data used (Rosenberger and Stanley 2006); this is fully acknowledged here and a full list of references is provided, taking care to note the source and quality of the information (Appendix I). This approach to data collection has been used in previous attempts to estimate global values of fish prices and subsidies (Khan *et al.* 2006; Sumaila *et al.* 2007). When no data for a country could be found, an effort was made to contact the appropriate government agency to request any available data. In order to make comparisons and evaluations of data across countries, 2003 was set as the base year, and efforts were made to obtain data from as close to this year as possible. Expenditure data found for different years were transformed into 2003 USD prices using country-specific consumer price index (CPI) and 2003 currency exchange rates (World Bank 2009).

#### 2.2.3 Filling the gaps

Data was not available for all countries considered in the study, so data gaps had to be filled with estimates. We describe how this was done for each of the three MRA sectors below.

#### 2.2.4 Recreational fishing

A benefit transfer approach (direct value transfer) was used to fill gaps in countries with partial data. For example, when only total angling expenditure was found for a country and the ratio of total to marine anglers was available, marine expenditure was obtained using this proportion, assuming that expenditure per capita was equal for marine and freshwater anglers. When these types of proportions were needed but not known for a country, those of similar (neighboring) countries, sub-regions or regions (in that order), as classified by the FAO (UN 2008), were used. Using independently defined geo-political subunits helps address the problems in generalization error that may occur when transferring values from one location to another under the assumption that these share similar traits (Brouwer 2000; Rosenberger and Stanley 2006). Such an approach is indeed imperfect, and ideally one would like to develop an econometric model allowing for statistical inferences. It is, however, a vital tool for large scale analyses which would otherwise prove impossible to perform (Rosenthal and DiMatteo 2001).

Once these partial data gaps were filled, expenditure and participation data were converted into participation rate (as a percentage of the country population) and expenditure per capita (2003 USD per angler per year). After classifying countries according to their FAO sub-region (UN 2008), data gaps were filled through direct value transfer. This involved assigning values of participation rate and expenditure per capita to data-less countries using the estimates for countries in the corresponding country group (Sumaila *et al.* 2008; Sumaila *et al.* 2007), using the continent average when no sub-regional estimate was available (Table 2.1).

Table 2.1. Initial data for benefit transfer method. 'Participation rate' refers to the number of anglers in a country as a percentage of that country's population. 'Expenditure per capita' is in 2003 USD. 'Data/region' refers to the number of countries with country-level data on participation and expenditure and the number of countries (where recreational fishing takes place) in each FAO region or sub-region.

| **Region** | **Participation rate** | **Expenditure per capita (USD)** | **Data/Region** |
|---|---|---|---|
| ***Africa*** | **0.28** | **239** | **4/23** |
| North Africa | 0.0042 | 239 | 1/3 |
| South Africa | 0.38 | 324 | 2/2 |
| Middle Africa | - | - | 0/3 |
| West Africa | - | - | 0/8 |
| Eastern Africa | 0.38 | 68 | 1/7 |
| ***Asia*** | **0.18** | **394** | **7/27** |
| Western Asia | 0.19 | 325 | 3/11 |
| Eastern Asia | 0.31 | 514 | 3/5 |
| Southern Asia | 0.0006 | 239 | 1/5 |
| South-Eastern Asia | - | - | 0/6 |
| ***Europe*** | **3.73** | **413** | **17/25** |
| Northern Europe | 6.21 | 278 | 8/10 |
| Southern Europe | 1.43 | 557 | 4/6 |
| Western Europe | 1.88 | 613 | 4/4 |
| Eastern Europe | 0.54 | 110 | 1/5 |
| ***Americas*** | **1.83** | **907** | **8/31** |
| North America | 4.72 | 1,016 | 2/2 |
| Caribbean | 0.23 | 540 | 1/11 |
| Central America | 1.13 | 1,527 | 3/8 |
| South America | 0.79 | 50 | 2/10 |
| ***Oceania*** | **17.69** | **522** | **2/12** |
| Australia and New Zealand | 17.69 | 522 | 2/2 |
| Melanesia | - | - | 0/5 |
| Micronesia | - | - | 0/4 |
| Polynesia | - | - | 0/1 |
| **Total** | **2.97** | **469** | **38/118** |

Under the assumption that participation rates and expenditure per capita have remained constant over the recent past, total expenditure and participation was calculated using the 2003 population of each country (CIA, 2003) where data suggests that recreational fishing takes place. Countries with known participation rate and expenditure per capita were assigned their specific values. According to case studies in Mexico and the US, between 23 and 45 thousand USD (2003), respectively, of direct expenditure on recreational fishing generates one full time job (Steinback *et al.* 2004; Gentner and Steinback 2008; Southwick Associates, Inc. *et al.* 2008), so employment was calculated using the US estimate for developed countries and the Mexico estimate for developing countries. Countries were classified under these two groups using their Human Development Index (HDI) as reported by the UNDP[3] (2007). This is clearly not an ideal estimation, but it is useful here because of the limited availability of complete data (reported employment and expenditure) in the same year for a given country.

### 2.2.5 Whale watching

In the case of whale watching, data gaps were minimal thanks in large part to very complete and comprehensive studies carried out by Hoyt (2001) and Hoyt and Iñiguez (2008), and the country estimates from these studies were included in the current analysis. Detailed information on expenditure and participation in the whale watching industry was extracted from these reports for 1998 and 2006. For each country, expenditures were first converted to 2003 USD using country specific consumer price indexes (CPI) and the 2003 exchange rate. Expenditure per capita was then calculated for every country. Employment in the whale watching industry is not well documented, but Hoyt (2001) and Hoyt and Iñiguez (2008) reported the number of communities in each country which were engaged in the whale watching industry. We used other community-level case studies on whale watching-generated employment to estimate the average

[3] The Human Development Index calculated by the United Nations Development Programme provides an index of average well-being in each country. The figure is a weighted average of GDP per capita, health (measured by average longevity) and average level of education.

number of operators per community (~4) and the number of direct jobs per whale watching operator (~7) (Parsons 2000; Hoyt 2001; Rossing 2006; Economists At Large 2007; Hoyt and Iñiguez 2008). This method was used to fill the gaps in employment data; employment was then calculated as a function of expenditure in each country.

In 2001, Hoyt reported that participation in whale watching around the world increased at an average of 12.1% during the 1990s. The latest findings at a large scale, a study for Latin America (Hoyt and Iñiguez 2008), show a continued increase in whale watching participation at an average of 11.3% per year from 1998 to 2006. So, we assumed an average worldwide rate of 10% increase per year to calculate participation in 2003. Expenditure was calculated assuming expenditure per capita remained the same as in 1998 or 2006, depending on the year of original country data. Using these estimations, we then calculated the number of jobs that would likely be generated as a result of total expenditure for whale watching in each country.

#### 2.2.6 Diving

For diving, the extremely limited amount of country-level information restricted the analysis to gross estimations of the global value and participation in the industry based on data from the US (PADI 2009) and Caribbean markets, together with world participation rates for diving and snorkeling[4]. Employment was calculated using the average yearly expenditure per one full time job in the US (The Leisure Trends Group 2000), under the assumption that expenditure per capita on diving and snorkeling equipment is comparable around the world. While seemingly unrealistic, this assumption is reasonable given the relatively standard equipment, training and certification systems that are used for these activities worldwide.

[4] The number of snorkelers was calculated using the US ratio of divers to snorkelers, 4:1 (NPD Group, Inc. 2000). Based on retail equipment prices, we assumed that snorkeling requires only 15% of the per capita expenditure of SCUBA diving.

## 2.3 Results and discussion

### 2.3.1 Recreational fishing

We found that recreational fishing takes place in 118 maritime countries around the world; country-level data on expenditure, participation and employment was found for 38 of them (32%) (Fig. 1).

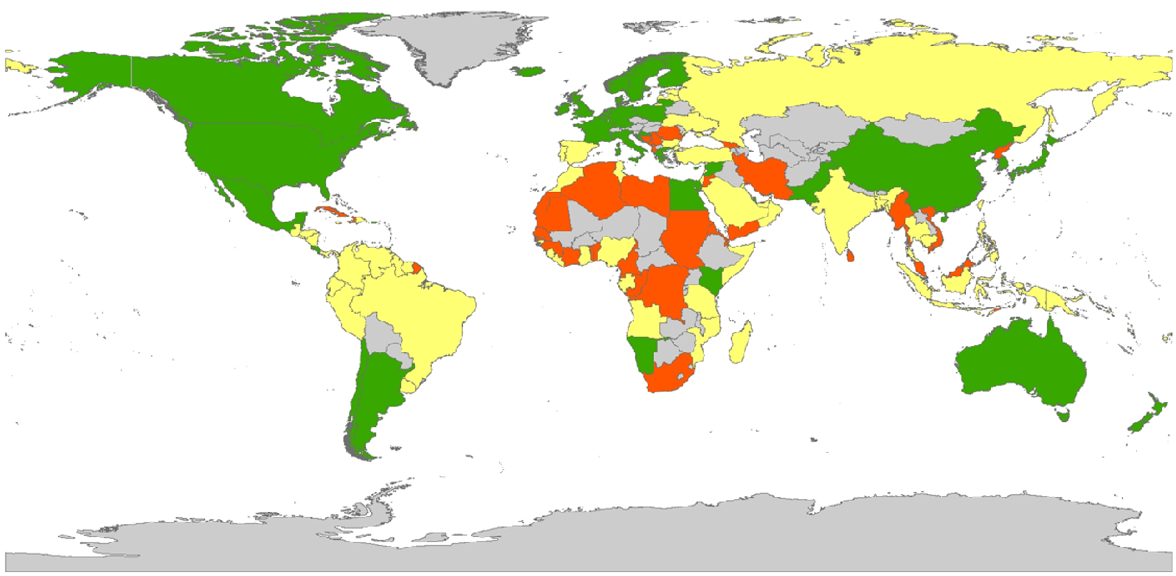

Figure 2.1. Results of initial data review of country participation in marine recreational fishing. Green= Yes (data available); Yellow= Yes (no data available); Red= No. Grey= Not a maritime country.

We assumed that finding no data on a country implies that recreational fishing either does not take place at all or is insignificant; this assumption likely makes our estimates conservative. In general, such countries have substantial social and political problems, or are in regions of the world with little marine commercial fishing, which could indicate that there is little opportunity for this activity to take place. It is important to reiterate that our working definition of recreational fishing excludes all forms of fishing whose *main* motivation is not pure recreation. Thus, countries with long traditions of fishing but where a large portion of the catch is either sold or relied upon for consumption would be classified as not taking part in recreational fishing.

Benefit transfer methods have been used widely to provide global level estimates in light of limited data (Rosenberger and Loomis 2001; Sumaila *et al.* 2007). We split the world according to FAO sub-regions to more accurately account for socioeconomic and ecological similarities between countries, which was supported by the data within regions and sub-regions (UNDP 2007). Perhaps unsurprisingly, Oceania has the highest participation rate in marine recreational fishing, an estimated 17% of the population. No data was found for Melanesia, Micronesia and Polynesia, which are somewhat more similar to Caribbean island-states and perhaps should have been assigned those values. Nonetheless, the values from Australia and New Zealand were used for these countries to maintain methodological consistency. The next highest participation rate was found in Europe at 3.7%. Although a vast number of Europeans fish for recreation (over 40% in some northern European countries; Toivonen *et al.* 2004), the high availability of freshwater bodies results in a much lower proportion of marine recreational fishing. A similar result occurs in the North America sub-region (Canada and the US under the FAO classification), where around 80% of recreational fishing takes place in freshwater; in all of the Americas, the average marine recreational fishing participation rate is almost 2%. At 0.3 and 0.2 %, respectively, Africa and Asia have the lowest participation rate among FAO regions.

Recreational fishing participation rates depend on many socio-demographic and cultural factors and to some degree on the amount of leisure time available to people, which is related to income level. Using socio-economic, geographic and demographic indicators, Arlinghaus (2006) was able to explain 40% of total variance on the likelihood of angling in a given year in Germany. While exploring these types of indicators was not one of the main objectives of this study, the observed trend of the relation between sub-regional participation rate and average HDI lends support to the idea that well-being and income are positively correlated to fishing as a leisure activity (Fig. 2).

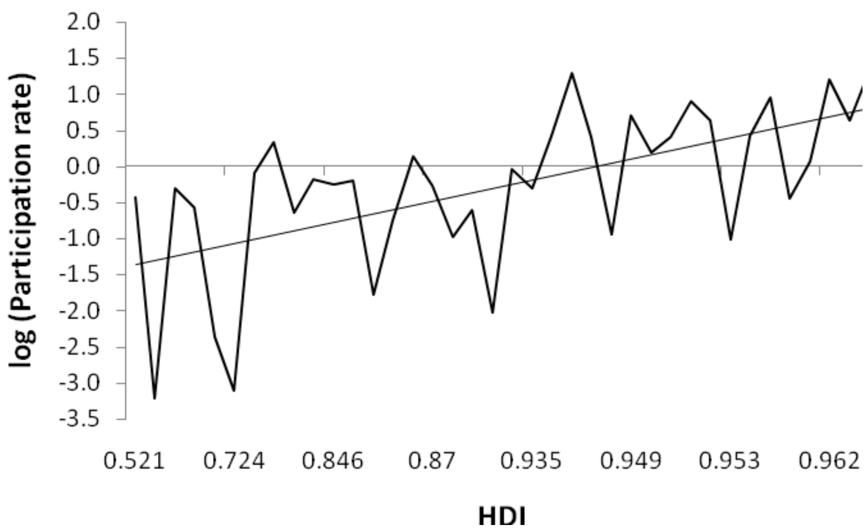


Figure 2.2. HDI and participation rate of countries in which data on marine recreational fishing was found. $R^2$= 0.3.

Generally, country-level expenditure data is obtained through surveys, which ask respondents to provide an estimate of all recreational fishing-related expenditures (including bait, tackle, boats, license fees, accessories and trip-related costs) during a given year. As such, we would expect some correlation between expenditure per capita on recreational fishing and the angler's income level, so it is perhaps not surprising that yearly expenditure per capita was highest in the Americas (\$907), Oceania (\$522) and Europe (\$413), and lowest in Asia (\$394) and Africa (\$239). However, plotting GDP per capita by itself against yearly expenditure per capita of individual countries does not reveal any clear trends (Fig. 3).

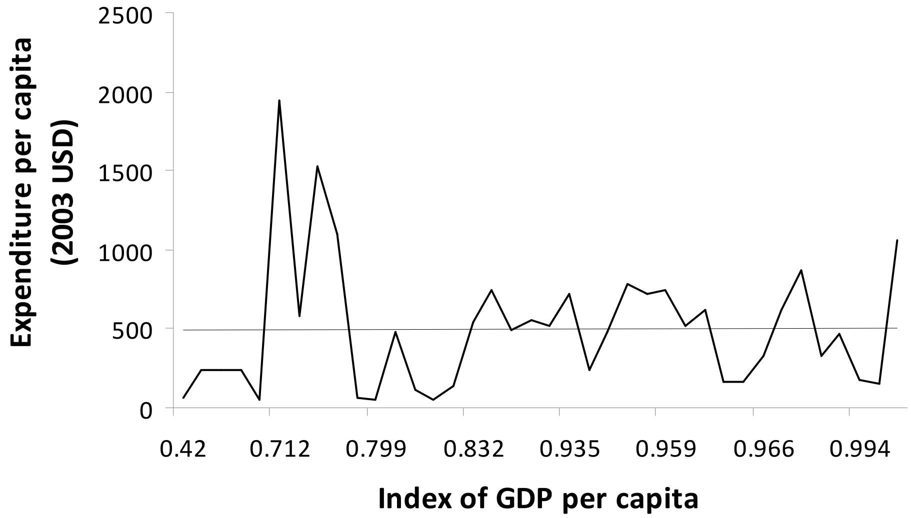


Figure 2.3. UN (2007) index of country GDP per capita and yearly expenditure per capita in 2003 USD. Mean= 497 USD.

As with participation rate, expenditure on recreational fishing depends to a great extent on the socio-economic characteristics of each country. Nevertheless, recreational fishing in many countries does not rely solely on the local population, but (sometimes to a great extent) on foreign visitors. This is perhaps most evident in the Central America sub-region which, despite a relatively low average HDI (0.762), has by far the highest expenditure per capita in the world, over 1,500 USD per year (Table I). The great majority of expenditures in these countries stem from foreign anglers, generally from the US and Canada (IBERINSA 2007; Fedler 2008; Southwick Associates, Inc. *et al.* 2008). When surveyed, at least some part of their transport and accommodation costs would be included under reported expenditure, driving the average expenditure higher than in their own region of origin. Although this may make it difficult to understand local anglers' expenditure level, much of the expenditure occurs in the country of destination, and thus can be used by these countries in their own economic analyses.

We estimate that in 2003, 58 million recreational anglers around the world generated a total of 39.7 billion USD in expenditures, supporting over 954 thousand jobs

(Table 2.2). Although further data collection is needed to better quantify the magnitude of recreational fishing worldwide and resolve potential issues of double-counting, countries with data in the current analysis accounted for almost 95% of estimated total expenditures and 87% of participation, so this estimate likely provides a close approximation to actual recreational fishing effort and expenditure levels.

One of the main reasons why many countries have embraced MRAs is their high economic benefits, particularly their per-biomass value as compared to commercial extraction (Hoyt and Hvenegaard 2002; Brander *et al.* 2007). The main reason for this is that MRAs, if undertaken in a manner which reduces their potential negative environmental impacts, can continually provide value from the same ecosystems or individual organisms through time. In the case of recreational fishing, the exceptionally high expenditure for catching fish has an implied but highly relevant implication for fish conservation. While it has been suggested that the economics of commercial fishing serves as a barrier to severe overfishing or extinction of marine fish stocks (Gordon 1954), recreational fishers do not fish for economic profit and therefore do not have this constraint. Besides the biological and ecological impacts inherent to any kind of fishing, this point must also be taken into account when considering the potential effects of recreational fisheries. In a study of the impacts of recreational fishing on US marine stocks, Coleman *et al.* (2004) found that once small pelagic fisheries are removed, recreational catches are equal to 10% of total US commercial catch. Assuming a similar trend in catch per angler, world landings in 2003 (*www.seaaroundus.org)*, and the estimate of total marine anglers in this study, recreational fisheries landings would total about 1 million metric tonnes per year, or 1.7% of world commercial catch minus small pelagics (sardines, herring and anchovies); if catch-release mortality is taken into account, impacts from recreational fishing could be significantly greater.

### 2.3.2 Whale watching and diving

Much less material was found for whale watching and diving compared to recreational fishing. This is likely a product of the relative size and recent development and growth of these two industries compared to recreational fishing.

Data on whale watching was found in a total of 70 countries (Fig. 4), mostly from 1994-2006 (Hoyt 2001; Hoyt and Iñiguez 2008). One of the most interesting aspects of this activity is that it has been growing steadily and substantially for the past decade. In Latin America, the whale watching industry growth rate (in terms of participation) is three times higher than tourism as a whole (Hoyt and Iñiguez 2008).

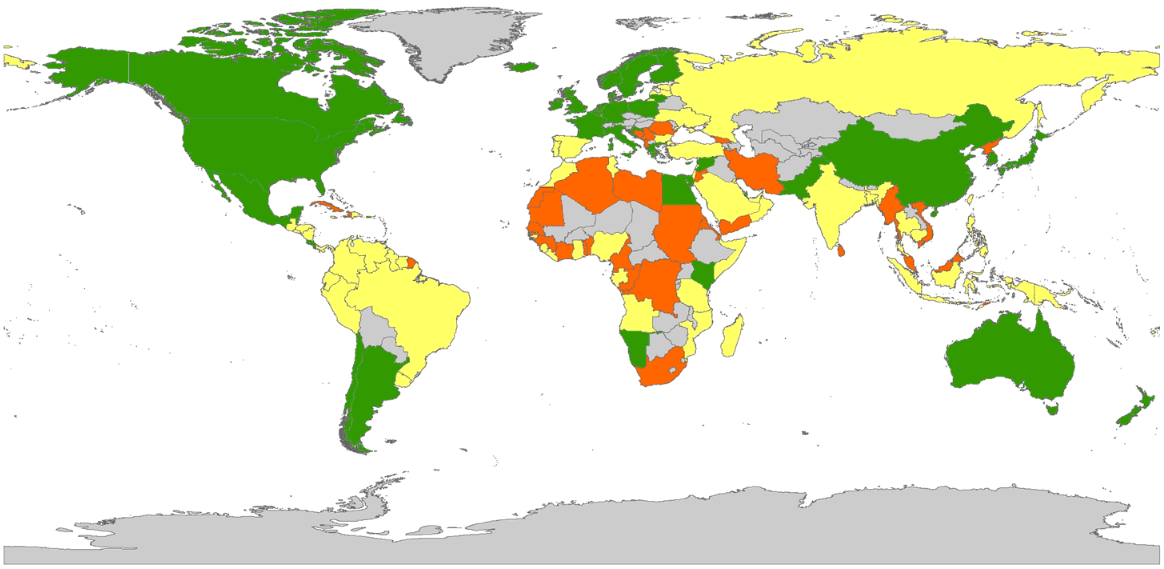

Figure 2.4. Results of initial data review for participation in whale watching by country. Green= Yes (data available); Yellow= Yes (minimal); Red= No. Grey= Not considered.

After adjusting the data to 2003, it is estimated that over 13 million people worldwide participated in whale watching in that year. Their expenditure (including ticket price, accommodation and travel expenses wholly attributable to whale watching) per year is estimated to be over 1.6 billion USD (2003) worldwide. Based on available data, it is estimated that 18 thousand jobs worldwide are supported by this industry each year

(Table 2.2). Given that marine mammals can occur in practically all of the world's oceans (Kaschner *et al.* 2006), any maritime country could theoretically engage in whale watching as an industry. The fact that this is currently not the case implies that other conditions, which may be cultural or social in nature (e.g., tourism infrastructure), must be addressed in order to effectively utilize whales in this manner.

There is very little country-level data on recreational diving outside of the US, Australia and, to some extent, Canada and the Caribbean region. In the US, the Diving Equipment and Marketing Association (*www.dema.org*) has undertaken market research studies which provide total expenditure and participation in recreational diving. There are also studies of Canadian recreational diving from which we estimated that an average of 48 thousand USD of direct expenditure on recreational diving creates one full-time job (Ivanova 2004). Using these surveys and available data on total active divers (Cesar *et al.* 2003), it is estimated that every year, 10 million active recreational divers and 40 million snorkelers around the world generate over 5.5 billion USD (2003) in direct expenditures, supporting 113 thousand jobs (Table 2.2). Based on the yearly per capita expenditure calculated from surveys (341 USD), this is most likely an underestimation.

Table 2.2. Summary of estimated benefits from ecosystem-based marine recreation. All estimates are for 2003.

Table 2.2.

| **Activity** | **Expenditure (USD billions)** | **Participation (millions)** | **Employment (thousands)** |
|---|---|---|---|
| **Recreational fishing** | 39.7 | 58 | 954 |
| **Whale watching** | 1.6 | 13 | 18 |
| **Diving and snorkeling** | 5.5 | 50 | 113 |
| **Total** | **46.8** | **121** | **1,085** |

Whale watching and recreational diving are similar in that they theoretically should have minimal to no negative impacts on the marine ecosystem. In fact, if they are well managed and made to follow proper guidelines, they can have positive impacts through education or even financing of marine parks and protected areas (Wilson and Tisdell 2003). However, these activities are often not conducted under these guidelines and merely use the 'ecotourism' label to attract customers (Carrier and Macleod 2005; Hoyt 2005). As has been observed in many diving locations, natural sites have a carrying capacity above which users perceive the site as overcrowded. This will in turn affect their travel to and expenditure in those sites (Brander *et al.* 2007).

### 2.4 Summary

From our current analysis, we estimate that in 2003, 121 million people around the world, almost 2% of the world population, engaged in ecosystem-based MRAs. Their total expenditure alone, 47 billion USD (0.1% of global GDP in 2003), supported over one million jobs around the world. In contrast, marine capture fisheries employ between 21 and 50 million people worldwide (Berkes *et al.* 2001; Garcia and de Leiva-Moreno 2003), with a global ex-vessel catch value of 80 – 85 billion USD a year (Sumaila *et al.* 2007; World Bank 2008). Marine resource populations clearly have a very high market value; uses other than marine capture fisheries are an increasingly important source of this value, and will continue to gain strength if managed appropriately. We must stress here that all of these activities are not mutually exclusive and that commercial fisheries can also improve through better management.

Although it is unlikely that the current increasing trends in ecosystem-based tourism can be maintained forever, there is much value to be added by undertaking MRAs in a manner that avoids disturbing the marine ecosystem and seeks to educate the public on the importance of its protection. Healthy ecosystems with abundant life are not only good in ecological terms, but are directly related to human activities that create economic benefits, a win-win situation.

### 2.5 Concluding remarks

Ecosystem-based MRAs are a major industry that directly benefits from the marine ecosystem. These activities provide considerable socio-economic benefits to people around the world. Meanwhile, although their ecological effects are very likely much less than those of fishing or other commercial extraction, the magnitude at which these activities take place can only augment their impacts. As such, it is vital that their further development be undertaken with precaution and in full recognition of the potential costs associated with their considerable economic benefits.

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**Chapter 3**

# 3 The Global Potential for Whale Watching[5]

## 3.1 Introduction

As with practically all other marine organisms, whales have historically been subject to human use. Whales have long been, and still are, hunted by some aboriginal groups around the world, including communities in Canada, USA, Greenland, Russia, South-Eastern Asia and the Caribbean (Reeves 2002). Although their meat, blubber and bones are used, a great value is placed on the hunt itself, which often is a ceremony unto itself and pays homage to community identity and historical traditions (Beck 1996). However, much of the current controversy around whales and whaling concerns the much larger industry that operated from the late 1600s until the late 1900s. This was an unequivocally commercial enterprise, and led to the decline of many whale populations around the world (Holt 1985; Christensen 2006).

In 1946, what is now known as the International Whaling Commission (IWC; *www.iwcoffice.org*) was created with the mandate of regulating whaling nations and ensuring that their hunts are sustainable. However, due to shifts in country membership and the persistent inefficacy of management measures, the IWC evolved into a preservationist body, culminating with the 1982 proclamation of a global moratorium on the commercial whale hunt. While this was intended to be a temporary measure pending new scientific data, the moratorium has until now not been lifted. Nevertheless, in addition to aboriginal communities, a few countries have continued to hunt whales, either through legal maneuvering or outright objection to the resolution (Clapham *et al.* 2007). The IWC voting system has a remarkably political undertone, with pro-whaling (led by Japan, with support from Iceland and Norway) and anti-whaling (led by a coalition of 'like-minded' countries including Australia, New Zealand and the USA) voting blocs (Mandel

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1980), each supported by NGOs (representing aboriginal groups in the former, and conservation groups in the latter camp). While these blocs have traditionally included a set of core countries, each side has accused the other of using aid and distorted scientific information to influence the vote of a host of small developing countries which often have not historically had a particular stance on the whaling issue (Ishii and Okubo 2007; Miller and Dolsak 2007).

Pro-whaling arguments contend that the whale hunt is part of a national identity (Komatsu and Misaki 2001), that some whale populations can be hunted sustainably under strict and precautionary scientific guidelines (Morishita 2006), and that such whaling would, as a side benefit, increase the fish available to fisheries, because, after all, "whales eat fish" (Komatsu and Misaki 2001). Opposition to whaling hinges on several issues, the main ones being a fundamental aversion to the killing of charismatic and intelligent animals (Einarsson 1993), and the past failure of the IWC to ensure sustainable whaling (Whitehead *et al.* 1997). As for the whales-eat-fish argument, it has been repeatedly shown to be without substance (Kaschner and Pauly 2005; Gerber *et al.* 2009; Morissette *et al.* 2010). Moreover, a strong case can be made for sparing whales, which can be a source of significant benefits sustainable over time through the whale watching industry, which requires, and in fact profits from, their continued existence and protection (O'Connor *et al* 2009).

Whale watching[6] is a rapidly growing industry around the world, currently generating an estimated ~2.1 billion USD (2009) in expenditures and supporting about 13,000 jobs worldwide (O'Connor *et al.* 2009). The increasing preference of affluent individuals for 'environmentally friendly' leisure activities suggests that there is further potential for this sector, which in some regions has shown a much higher growth rate than tourism in general. Indeed, many countries have invested in this industry for more than a decade, with generally positive results (Hoyt 2001; Hoyt and Iñiguez 2008). But can all coastal countries access this market? Using available data on global marine mammal

[6] Whale watching is defined here as watching any marine mammal from a boat or shore; depending on the context, 'whales' may include marine mammals other than large cetaceans.

distribution and the current whale watching industry, an estimate of the potential benefits from whale watching is provided for maritime countries that have yet to undertake whale watching operations, or have done so only marginally.

## 3.2 Methods

### 3.2.1 Input data

A number of large-scale studies of the global whale watching industry were used for this analysis. They included values (measured or estimated) for participation, expenditure (direct and indirect) and employment in whale watching (Hoyt 2001; Hoyt and Iñiguez 2008; O'Connor *et al.* 2009; Cisneros-Montemayor and Sumaila *in press*). With the exception of Cisneros-Montemayor and Sumaila (*in press*), these studies were all prepared for the International Fund for Animal Welfare (IFAW; *www.ifaw.org*) and represent the most comprehensive global surveys of whale watching available; unless otherwise stated, data used in this study are from these reports.

Indicators of whale watching industry performance were calculated for each of the countries in which whale watching occurs. These indicators are: average expenditure per capita, ratio of indirect to direct expenditure, and the number of yearly whale watchers needed to support one job. These indicators were averaged by FAO sub-region, or 'areas' (FAO; *www.fao.org*) and applied to countries which have yet to establish whale watching operations, or where these are marginal (<50 whale watchers/year, in four cases), here called 'non-whale watching' (NWW).

In light of the spotty nature of quantitative information regarding marine mammal occurrence, predictions about large-scale species distribution have been generated based on a Relative Environmental Suitability (RES) model (Kaschner *et al.* 2006). This model computed probable habitat from a range of oceanographic factors, and outputs were then tested against available observation data to validate predictions for over 100 species of marine mammals. The number of marine mammal species (MS) and their relative abundance (MA) within each country's EEZ, jointly with the yearly total tourist arrivals to that country (World Tourism Organization (WTO); *www.world-tourism.org*), was used to predict the number of whale watchers in a country in one year.

Estimates were calculated using a binomial generalized linear model (GLM) which defined success as the number of whale watchers in a country ($w_1$) and failure as the number of total tourists to a country who did not go whale watching ($w_0$); values of MS and MA were used to explain the probability of success. The resulting regression coefficients β were then used together with yearly tourist arrivals $T$ in a country $i$ to estimate the potential number of whale watchers $W$ as:

$$W_i = T_i * \left[\frac{(e^{\alpha+\beta_1 MS_i+\beta_2 MA_i})}{(1+e^{\alpha+\beta_1 MS_i+\beta_2 MA_i})}\right] \quad (1)$$

### 3.2.2 Regional analyses

To address differing socio-economic characteristics across countries, sub-regional values were used as the baseline for calculating expenditure and employment in NWW countries through a benefit transfer approach. Direct benefit transfer is a form of valuation technique in which data gaps for specific points are filled with values from others that are assumed to be similar, i.e., in independently-defined sub-regions, or strata (Brouwer 2000; Rosenberger and Stanley 2006). While benefit transfer has its shortcomings, it is nonetheless necessary in dealing with data-poor situations and has been used widely in global-scale studies (Rosenberger and Loomis 2001). In this case, potential direct (*DE*) and indirect expenditure (*IE*), and employment (*J*) from whale watching for each country *i* in sub-region *j*, were computed based on the previous estimate of yearly whale watchers (*W*) as:

$$DE_{ij} = W_i * (DE_j/W_j) \quad (2)$$

$$J_{ij} = W_i * (J_j/W_j) \quad (3)$$

$$TE_{ij} = DE_{ij} * (IE_j/DE_j) \quad (4)$$

In the case of employment, sub-regional averages were estimated based on values that were themselves estimated based on a review of several large-scale studies and site-specific case studies (Cisneros-Montemayor and Sumaila *in press*). To maintain consistency with other global whale-watching studies, 'direct expenditures' were defined as money spent on the whale watching activity itself (e.g. ticket price), while 'indirect expenditures' are the total amount spent on trip-related goods and services, such as accommodation, food, and travel costs (Hoyt 2001). The mean for Melanesia and Polynesia was used for Micronesia, as no sub-regional values were available in this case.

The assumption that investing in whale watching will necessarily boost total tourism has been avoided; instead, the potential for the industry is considered as an additional source of revenue from current tourist arrivals, bounded by marine mammal distribution and abundance.

### 3.3 Results

In total, 144 maritime countries were included in the analysis, spanning 21 sub-regions around the world. Of these countries, 68 have already invested in the whale watching industry. A summary of whale watching indicators by sub-region is presented in Table I. All results are presented by regional values, which (with the exception of Micronesia; see Section 2) represent the average of specific country estimates in that region.

Table 3.1. Regional indicators for whale watching industry performance: countries with whale watching compared to total countries in the region (CWW(TC)); percentage of whale watchers relative to total tourist arrivals (WW/T%); yearly whale watchers per job created (WW/Job); number of marine mammal species (MS) and the sum of their relative abundance (MA) within a country's EEZ. Expenditure is in USD (2009).

| FAO Area | CWW (TC) | WW/T % | Expenditure/ Capita (USD) | WW/ Job | MS | MA |
|---|---|---|---|---|---|---|
| ***Africa*** | **11 (38)** | **2.62** | **115** | **623** | **33** | **0.26** |
| Eastern Africa | 5 (10) | 2.17 | 119 | 623 | 33 | 0.26 |
| Middle Africa | 1 (7) | 0.14 | 78 | 150 | 28 | 0.09 |
| Northern Africa | 1 (6) | 0.29 | 98 | 334 | 25 | 0.10 |
| Southern Africa | 2 (2) | 5.31 | 90 | 2,405 | 49 | 0.97 |
| Western Africa | 2 (13) | 3.46 | 160 | 180 | 27 | 0.03 |
| ***Americas*** | **24 (33)** | **3.24** | **282** | **1,283** | **36** | **1.11** |
| Caribbean | 7 (13) | 4.00 | 271 | 1,438 | 26 | 0.04 |
| Central America | 6 (8) | 1.40 | 305 | 348 | 32 | 0.56 |
| North America | 2 (2) | 8.53 | 162 | 3,657 | 61 | 5.33 |
| South America | 9 (10) | 2.69 | 303 | 1,259 | 41 | 1.36 |
| ***Asia*** | **16 (34)** | **1.44** | **63** | **1,549** | **31** | **0.62** |
| Eastern Asia | 4 (5) | 3.06 | 72 | 4,215 | 44 | 1.14 |
| South-Eastern Asia | 5 (9) | 1.13 | 54 | 650 | 29 | 0.78 |
| Southern Asia | 4 (6) | 1.10 | 45 | 217 | 31 | 0.31 |
| Western Asia | 3 (14) | 0.25 | 94 | 1,267 | 17 | 0.08 |
| ***Europe*** | **11 (26)** | **1.46** | **225** | **1,016** | **37** | **1.45** |
| Eastern Europe | 1 (5) | 0.01 | 892 | 86 | 39 | 5.26 |
| Northern Europe | 5 (10) | 2.71 | 129 | 937 | 40 | 1.38 |
| Southern Europe | 4 (7) | 0.63 | 161 | 1,557 | 23 | 0.37 |
| Western Europe | 1 (4) | 0.01 | 295 | 173 | 76 | 2.30 |
| ***Oceania*** | **6 (13)** | **12.75** | **101** | **1,544** | **39** | **1.57** |
| Australia and New Zealand | 2 (2) | 25.75 | 127 | 4,334 | 57 | 4.32 |
| Melanesia | 2 (4) | 2.19 | 55 | 81 | 30 | 0.33 |
| Micronesia | 0 (5) | 6.25 | 88 | 149 | 30 | 0.20 |
| Polynesia | 2 (2) | 10.32 | 120 | 218 | 30 | 0.07 |
| **Total** | **68 (144)** | **3.26** | **179** | **1,219** | **35** | **0.95** |

Current data suggests that the number of whale watchers is significantly related to the magnitude of a given country's overall tourism industry (Fig. 3.1), which is comforting given the large spatial scale of the analysis. Nevertheless, the occurrence and abundance of whales are logical constraints to the potential for whale watching growth. These parameters are therefore included as explanatory variables for whale watchers in a binomial GLM; the resulting coefficients were used to estimate the potential yearly whale watchers in each country (Fig. 3.2). The number of total tourist arrivals, as well as the number of marine mammal species and their relative abundance within a country's EEZ were significantly ($p<0.001$) and positively correlated to the number of whale watchers.

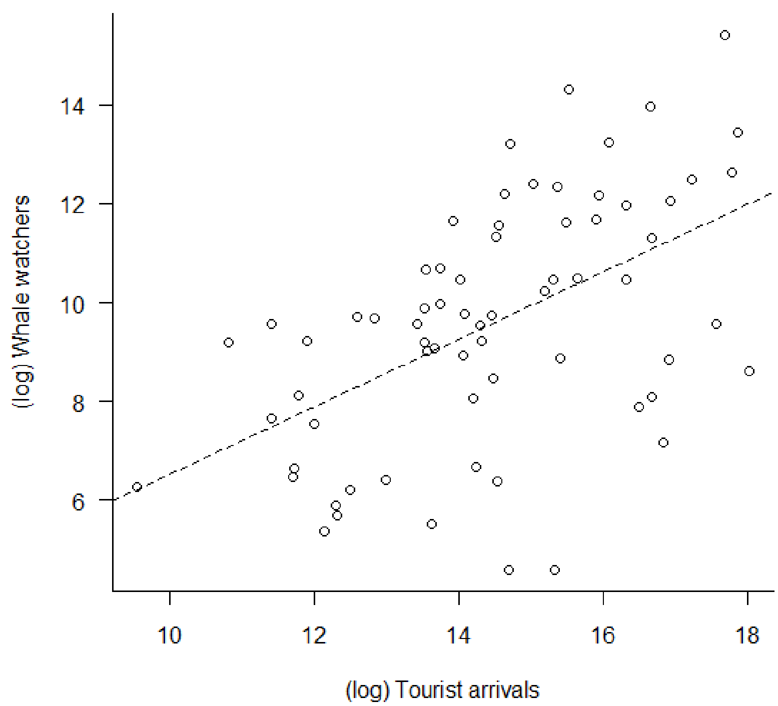


Figure 3.1. Relationship between total tourist arrivals and whale watchers (observed) by country (Hoyt and Iñiguez 2008; O'Connor *et al.* 2009; *www.world-tourism.org*). $R^2$= 0.29.

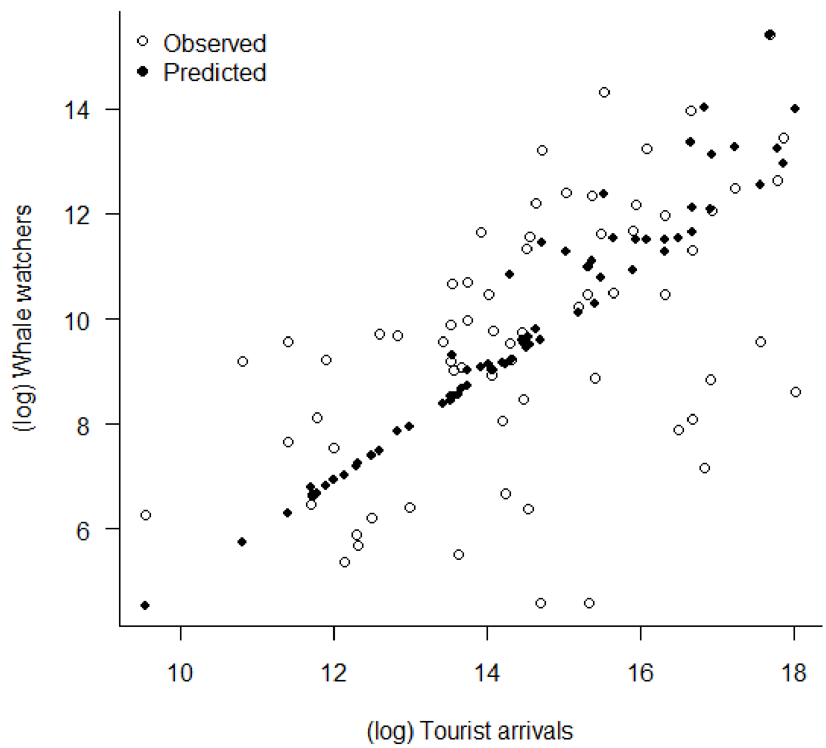


Figure 3.2. Observed and predicted yearly whale watchers for countries which currently engage in the activity.

Based on the resulting estimated potential whale watchers, sub-regional data were used to calculate the potential expenditure and employment for NWW countries. These results are presented in Table 3.2.

Table 3.2. Estimated potential yearly whale watchers (WW), expenditures and employment generated by whale watching.

| FAO Area | WW (1000) | Direct expenditure (1000 USD) | Total expenditure (1000 USD) | Employment |
|---|---|---|---|---|
| ***Africa*** | **145** | **8,132** | **15,588** | **516** |
| Eastern Africa | 2 | 76 | 218 | 18 |
| Middle Africa | 3 | 186 | 273 | 23 |
| Northern Africa | 118 | 6,839 | 11,556 | 353 |
| Western Africa | 22 | 1,030 | 3,540 | 121 |
| ***Americas*** | **46** | **5,715** | **27,523** | **63** |
| Caribbean | 32 | 3,841 | 15,480 | 18 |
| Central America | 14 | 1,722 | 7,457 | 44 |
| South America | 1 | 151 | 4,587 | 1 |
| ***Asia*** | **506** | **17,162** | **46,670** | **537** |
| Eastern Asia | 49 | 1,124 | 3,334 | 12 |
| South-Eastern Asia | 81 | 3,184 | 10,423 | 158 |
| Southern Asia | 16 | 411 | 1,002 | 83 |
| Western Asia | 360 | 12,444 | 31,911 | 284 |
| ***Europe*** | **657** | **150,294** | **323,299** | **4,585** |
| Eastern Europe | 260 | 108,683 | 231,796 | 3,032 |
| Northern Europe | 83 | 3,910 | 11,327 | 88 |
| Southern Europe | 63 | 2,272 | 5,868 | 8 |
| Western Europe | 252 | 35,430 | 74,308 | 1,456 |
| ***Oceania*** | **5** | **285** | **296** | **61** |
| Melanesia | 4 | 244 | 244 | 55 |
| Micronesia | 1 | 41 | 53 | 6 |
| **Total** | **1,360** | **181,588** | **413,377** | **5,762** |

* Total expenditure is equal to the sum of direct and indirect expenditure, where direct expenditure is the amount spent on a whale watching trip (e.g., ticket price) and indirect expenditure is the amount spent on trip-related costs (e.g., travel and accommodation) attributable to whale watching (Hoyt 2001).

To allow for comparison across regions, the number of marine mammal species and their abundance within the EEZ, as well as yearly tourist arrivals, were calculated relative to all countries in the analysis. In the case of potential whale watchers, values are relative to other countries for which estimates are provided (Fig. 3.3).

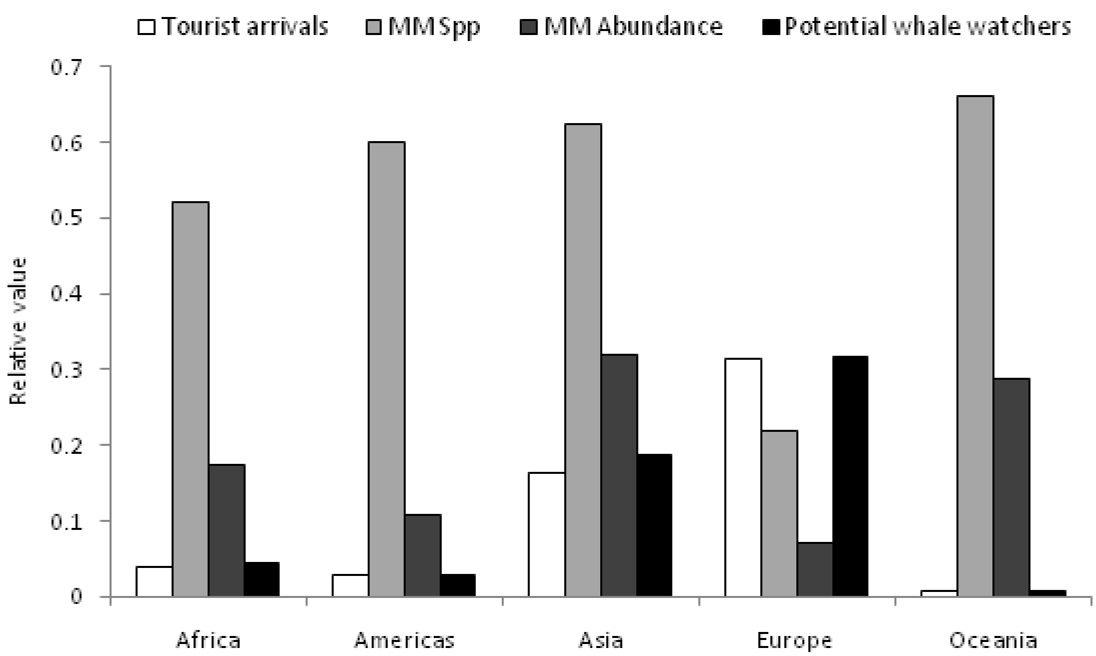


Figure 3.3. Average yearly tourist arrivals (white), species of marine mammals (light grey), abundance of marine mammals (dark grey) relative to all countries included in the analysis, and potential whale watchers (black), relative to the number in countries that do not currently engage in whale watching.

### 3.4 Discussion

Many countries around the world have invested in whale watching for several decades, and currently host about 13 million whale watchers a year, generating a total of over 2.1 billion USD (2009) and supporting about 13,000 jobs (O'Connor *et al.* 2009); around 20% of these totals are accrued by developing countries (as defined by the UN; UNDP 2007). The results of this study suggest that an additional 413 million USD and 5,700 jobs (Table

3.2) could potentially be generated by starting whale watching operations in countries that do not currently do so. Together with current reported figures, these estimates would bring the total benefits from whale watching to over 2.5 billion USD a year, supporting around 19,000 jobs around the world. Because the assumption that more whale watchers would necessarily result in more total tourists has been avoided, these estimates reflect the potential for whale watching as added value given current tourist arrivals to a country. A key finding is that about half of these estimated potential benefits would be captured by developing countries.

Based on worldwide marine mammal distributions, almost any coastal country could theoretically engage in whale watching (Fig. 3). However, the potential for this industry is also bounded by the total number of tourists that currently visit a country (Fig. 1), which is dependent on factors such as ease of access and security. In the case of small and remote island states in Melanesia, Micronesia and the Caribbean, the whale watching experience, e.g., of tourists from Europe and North America, would be strongly enhanced by the surrounding coastal and marine environments, which are in themselves appealing. However, this would be subject to the constraints and potentially strong fluctuations in overall tourism, which are driven externally by relatively high travel costs and the availability of alternative destinations (Hoyt 1999; Orams 2002). Conversely, marine mammals in a given country may not be sufficiently abundant to support a whale watching industry despite overall tourism. Regardless of the fact that some species may draw more whale watchers than others, the extensive global distribution of marine mammals (Kaschner *et al.* 2006) suggests that few regions should have this problem (Fig. 3).

Obviously, the highest potential for whale watching exists in regions that have a relatively high abundance of whales and/or tourists (Fig. 3; Table II). However, even seemingly ‘marginal’ benefits can be very important to the countries considered here, and may be relatively easy to achieve (Wilson and Tisdell 2003). Insufficient guidance on adequate implementation, as well as a lack of foresight as to the potential benefits from it, seem to be the main reasons for these countries not entering the whale watching market (Hoyt 1999). An important point to consider when establishing whale watching as an industry in any area is that it should be done in a precautionary manner to avoid potential negative impacts on marine mammals (Williams *et al.* 2002; Curtin 2003) or the marine ecosystem in general (Honey and Krantz 2007) and ensure that the industry be sustainable at a benefit-maximizing level. Particular attention should be paid to user preferences in order to avoid placing unnecessary stress on whales or other marine mammals. It has been shown that aggressive whale watching, potentially harmful for whales, does not result in increased consumer satisfaction (Orams 2000; Semeniuk *et al.* 2009).

Although whale watching can evolve into a very large commercial enterprise, in many developing countries it can be launched with little initial investment and can be carried out by local fishers who are already familiar with the area (Rossing 2006; Hoyt and Iñiguez 2008). This can therefore be an excellent alternative to expand income sources in the face of declining fisheries (Pauly *et al.* 2002), which are the pillar of livelihoods for communities in many coastal developing countries (Pauly 2006). Indeed, the involvement of local communities is vital for the implementation of conservation measures in an ecotourism context, as it is often these people who have the most to lose from restrictive management policies (Ransom and Mangi 2010).

A fact that must be stressed is that investing in the development of whale watching does not guarantee that it will bring in additional tourism. Rather, available data suggests that the number of tourists that already visit a particular country explains a large part of the number of whale watchers there (Fig. 1). Judging by the number of whale watchers as a percentage of total tourists to a given country (about 3% worldwide; Table II), the converse is probably not true, though whale watching may indeed become an additional attraction to a given site. An important point to consider is that tourism activities in general require both material and social infrastructures, which must be in place before any significant benefits can be realized from whale watching (Khadaroo and Seetanah 2008).

It has been reported that the stance of a particular country towards whaling significantly influences that country's appeal to whale watchers (Parsons and Draheim 2009), although countries supporting whaling at meetings of the IWC (Miller and Dolsak 2007; Swartz and Pauly 2008) have nonetheless become involved in the whale watching industry, generating yearly totals of about half a million visitors, 40 million USD in total revenue and 400 jobs (O'Connor *et al.* 2009). While this suggests that maintaining a pro-whaling stance (or at least voting as such in the IWC) and investing in whale watching by a country are apparently not mutually exclusive (and may in fact be a rational choice; Herrera and Hoagland 2006), commercial whaling of the sort that occurred in the past will lead to further declines in whale populations (Baker and Clapham 2004), directly and negatively impacting whale watching around the country in question, and the world.

Although it is becoming increasingly obvious that animals such as whales have intrinsic value, the fact remains that, in many regions of the world, a low potential for profits from non-extractive resource use translates into little incentives for conservation (Wells 1992). Given these conditions, it would be interesting to explore the potential role of international side payments, in the form of payments for ecosystem services (Bulte *et al.* 2008), from countries that accrue significant benefits from whale watching to those which do not, as an incentive for conservation (Munro 2008). While this study offers an estimate of potential revenue from whale watching, there is much less information regarding the possible costs (e.g., foregone fishing opportunities or foreign aid contingent on expressions of support for whaling) of marine mammal conservation, necessary for a full cost-benefit analysis to be undertaken. This is clearly an interesting future research project, particularly as the widespread development of whale watching industries may contribute to a shift of votes at the IWC, and a dissolution of the blocs that have made it largely dysfunctional.

### 3.5 Concluding remarks

Whale watching is undoubtedly an industry capable of generating socio-economic and ecological benefits to a country over time. However, a lack of proper infrastructure and/or a perceived lack of opportunity for entering the market have prevented some countries (particularly in the developing world) from realizing potential benefits from whale watching. With proper guidelines in place, and even without assuming any subsequent increases in total tourist arrivals, the continued protection of marine mammals can translate into benefits that are significant, sustainable, and relatively easy to attain.

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**Chapter 4**

# 4 Ecosystem Models for Management Scenario Analysis: A Case Study of Recreational and Commercial Fisheries Policy[7]

## 4.1 Introduction

It is widely accepted that commercial fisheries have, in great part, caused the large-scale decline of global fish stocks (Pauly *et al.* 2002). Although it is clear that the commercial fishing industry as a whole is not operating at its biological or economic optimum (Sumaila *et al.* 2008; World Bank 2008), it nonetheless continues to provide a significant amount of animal protein (FAO 2009) while generating millions of jobs (Garcia and de Leiva-Moreno 2003) and billions of USD in revenue (Sumaila *et al.* 2007). Meanwhile, recreational fishing has continuously developed around the world and now supports a very large industry which in some areas has overtaken commercial fishing as a source of revenue derived from the marine ecosystem (Cisneros-Montemayor and Sumaila *in press*). Ecological impacts from recreational fishing can be substantial (Lewin *et al.* 2006; Coleman *et al.* 2004), but commercial fisheries are still held to be chiefly responsible for the overall decline of fish stocks. This has caused widespread conflict over fish resources, which must now be allocated between two industries in direct or indirect competition. We focus here on one such case in Baja California Sur (BCS), Mexico.

Although large-scale recreational fishing is a relatively recent development in this region, BCS has a fairly long history of exploitation of pelagic fish stocks. Although most large-scale fisheries in Mexico were established around 1970, the Japanese long-liner fleet

[7] Parts of this chapter have been published as: Cisneros-Montemayor, A.M., Christensen, V., Arreguín-Sánchez, F. and U.R. Sumaila. (2009). Fisheries in Baja California Sur: a trophic based analysis of management scenarios. *In:* Palomares, M.L.D., Morissette, L., Cisneros-Montemayor, A.M., Varkey, D., Coll, M. and C. Piroddi (eds.). Ecopath 25 Years Conference Proceedings: Extended Abstracts. Fisheries Centre Research Reports 17 (3): 29-30 pp.

was operating in the region since the late 1950s, actively targeting tunas, marlin, sailfish and other billfish. This fleet had annual catches of several million fish before Mexico established and enforced its 200 nm Exclusive Economic Zone in 1976 (Squire and Au 1990). Since then, both the artisanal and industrial Mexican fleets have grown very rapidly due to aggressive government promotion and subsidies to fishermen and fishing infrastructure (Hernandez and Kempton 2003). Currently, around 80% of Mexican fisheries are over-exploited or at their maximum sustainable yield (Federal Gazette (DOF) 2004); in many cases, operating profits depend largely on government subsidies.

During the last 20 years, the recreational fishing industry in BCS (mainly targeting billfish) has grown and far surpassed commercial fishing as an industry in the state, most noticeably in the Los Cabos area, including the towns of Cabo San Lucas, San Jose del Cabo and Bellavista. In this area alone, recreational fishing is estimated to generate ~1.1 billion USD (2007) a year in total economic activity, supporting almost 25 thousand jobs (Southwick Associates Inc. *et al.* 2008). Billfish catches have remained relatively stable, but for some years have shown slight but steady declines (DOF 2004).

Although many artisanal and industrial fisheries continue to operate in the same region, none has had more conflict with the recreational sector than the shark long-lining fleet. In addition to shark population declines in the area, the main point of contention is its bycatch rate, primarily of billfish such as marlin, which by law are reserved for sport fishing up to 50 nm from shore (DOF 2004).

Conflicts between the sectors have recently been worsened by the proposal and approval of a shark fishery management law (NOM-029-PESC-2006; DOF 2007) that, although containing useful management measures for the shark fishery, does not specifically prohibit bycatch of billfish, but only states that such bycatch must be reported. Presumably, this was done in light of the fact that no explicit bycatch assessments had been conducted, although some estimates were of 40% billfish bycatch (DOF 2004). The tourism sector’s reaction was strong, mobilizing social and political groups against what they view as a direct threat to their livelihoods and economic benefits (El Sudcaliforniano 2007a,b; The Billfish Foundation 2006). Under intense sociopolitical pressure, a scientific study was carried out which found a 15% bycatch rate (as a proportion of shark catch) of marlin (INP 2007). This was followed by a mandate setting a 4% limit on bycatch for marlin and dorado and a 30% total bycatch limit for all non-target species (DOF 2008). Using current momentum, some have advocated for the complete shutdown of commercial long-lining in the region (El Sudcaliforniano 2010a,b). Whether this measure would stop billfish declines in the region is unknown.

Using Ecopath with Ecosim version 6 (*www.ecopath.org*), we constructed a model representing the pelagic ecosystem and fishing dynamics of BCS, as well as the outcomes, particularly pertaining to billfish and shark populations, of various fisheries management scenarios. Our objectives are to represent the current ecological effects and economic performance of fishing in the BCS region, allowing for an evaluation of the results of proposed policies meant to improve on these performance measures.

## 4.2 Methods

### 4.2.1 Study area

Baja California Sur is located on the southern part of the Baja California Peninsula, a mostly arid region with a very long coastline and a large EEZ (Fig. 1). The California Current runs along the western coast, producing cold-water upwelling events and a year-round mixing zone (Sanchez-Montante 2004). The eastern coast borders on the Gulf of California, a warmer sea with many rocky and coral reefs and with large upwelling events caused by seasonal wind patterns and several geostrophic gyres (Lavin and Marinone 2003). Both coasts support productive commercial and recreational pelagic fisheries year round, the most important of which target small pelagic fish, squids, sharks and billfish.

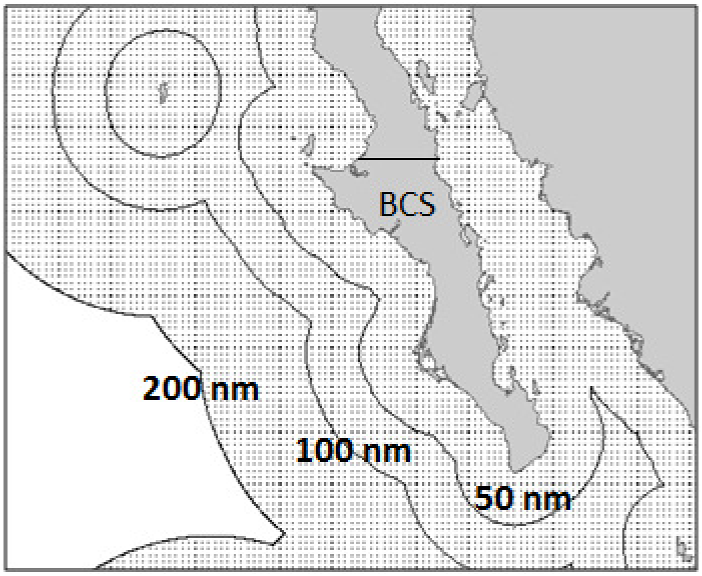


Figure 4.1. Baja California Sur (BCS), including Mexico's Exclusive Economic Zone (EEZ). Lines depict 50, 100 and 200 nautical mile (nm) EEZ limits.

### 4.2.2 Ecopath with Ecosim

Ecopath with Ecosim (EwE) is a software package which allows for ecosystem mass balance analysis as well as dynamic and spatially explicit modeling of human or environmental effects on the ecosystem (Christensen and Walters 2004). It is essentially a graphical user interface, which simplifies the application of underlying mathematical equations representing the biological characteristics of functional ecosystem groups (a species or a group of species) and their trophic relationship with one another. This allows for subsequent exploration of outcomes given ecosystem disturbances. The production ($P$) of each functional group ($i$) is calculated as:

$$P_i = Y_i + M_i \cdot B_i + E_i + BA_i + MO_i \cdot B_i \quad (1)$$

where $Y$ is the total fishery catch rate, $M$ is the instantaneous predation rate, $B$ is biomass, $E$ is the net emigration rate (emigration – immigration), $BA$ is the biomass accumulation rate and *MO* is other mortality. Predators and prey are linked through the predation mortality term *M* as:

$$M_i = \sum_{j=1}^{n} Q_j \cdot DC_{ij}/B_i \quad (2)$$

where *Q* is the total consumption rate for predator group *j* and DC is the fraction of its diet contributed by prey *i*. For each group *j*, *Q* is calculated from the leading parameters *B* and *Q/B*, the consumption/biomass ratio. Biomass dynamics are expressed through a series of differential equations calculating biomass growth *B* in time interval *t* as:

$$\frac{\partial B_i}{\partial t} = g_i \cdot \sum Q_{ij} - \sum Q_{ji} + I_i - (MO_i + F_i + e_i) \cdot B_i \quad (3)$$

where *g* is the net growth efficiency, *F* is the fishing mortality rate, and *e* and *I* are the emigration and immigration rates. The consumption rates *Q* are calculated based on the foraging arena concept (Walters *et al.* 1997), where only a portion of prey biomass (determined by its specific vulnerability parameter) is susceptible to predation at a given time. For an extensive review of EwE methods, capabilities and limitations, see Christensen and Walters (2004). Because of the nature of available data, we were unable to use the spatial analysis capabilities of EwE.

### 4.2.3 Ecopath input parameters

In order to provide a representative yet manageable model of the BCS pelagic ecosystem, 18 functional groups were identified (Appendix B, Table B1). Input parameters for the model were initially based on published large-scale pelagic EwE models for the north-central and Eastern Tropical Pacific Ocean (ETPO) (Kitchell *et al.* 2002; Watters *et al.* 2003) and on an ETPO EwE model developed by the Inter-American Tropical Tuna Commission through an expert-panel review process (Olson and Watters 2003).

These large-scale models were adapted to the study area and objectives based on EwE models for the Gulf of California (Arreguín-Sánchez *et al.* 2002; Arreguín-Sánchez *et* *al.* 2004; Morales-Zárate *et al.* 2004; Rosas-Luis *et al.* 2008) and reported diet composition and ecological dynamics studies for key functional groups (Alverson 1963; Abitia-Cardenas *et al.* 1997; Abitia-Cardenas *et al.* 1999; Lasso and Zapata 1999; Rosas-Alayola *et al.* 2002; Markaida and Hochberg 2005). The initial parameters used for the subsequent analyses are presented in Appendix B.

Using available data, the initial balanced Ecopath model was fit to time series of fishery data (discussed below); this estimates Ecosim vulnerability parameters based on the best fit to catch data, which are used for all subsequent scenarios.

#### 4.2.4 Fishery data

Commercial catch and effort data were obtained from government fishery statistics (National Fisheries and Aquaculture Commission (CONAPESCA) 2003-2007; DOF 2004, 2006) and peer-reviewed literature (Squire and Au 1990; Ditton 1996; Ortega-Garcia *et al.* 2003; Southwick Associates Inc. *et al.* 2008). We included recreational, artisanal, longlining, small seine, large seine, and Japanese longlining fleets, as well as two dummy fleets used to model changes in bycatch. Available data was sufficient to determine the historic increase in vessels engaged in the fishery since 1972 (Fig. 2), but does not address issues of capacity increase ("technology creep"), which increases the overall fishing mortality exerted on target species even when fleet size remain the same (Daniel Pauly, *pers. comm.*). We therefore adopted a conservative (1% for recreational and artisanal fleets, 2% for industrial fleets) yearly increase in effective fishing effort in order to include this effect into our data series of relative effort, which in fact improved our model fit.

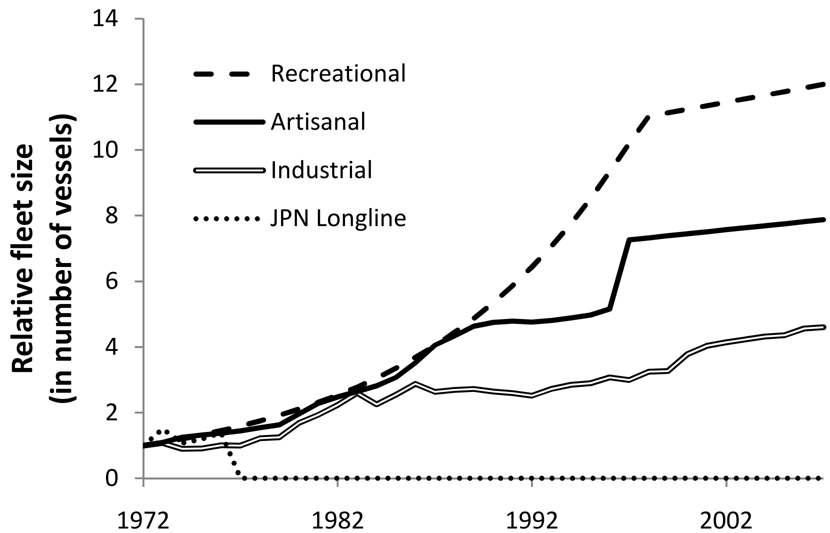


Figure 4.2. Fishing effort of recreational, artisanal, industrial (longliner and seiner), and Japanese longliner fleets operating in BCS, relative to 1972.

To our knowledge, there are no readily available datasets on recreational landings in BCS. Recreational billfish landings were calculated using the reported estimate of fish caught annually (DOF 2004), the average species length (Melo-Barrera and Felix-Uraga 2004; Madrid y Beltran-Pimienta 2001) and their length-weight relationship ($W = aL^b$) parameters (*www.fishbase.org*) to calculate tonnes caught per year. We divided this into landings and discards assuming a 70% release rate (Ditton 1996), and applied a discard (release) mortality rate of 26% for all billfish based on empirical findings for striped and white marlin (Cramer 2004). Because EwE requires catch data to be entered in terms of tonnes/km$^2$, total landings were divided by the model area, calculated from shore to the 100 nm limit using ArcGIS (*www.esri.com*), under the assumption that most fishing occurs closer to productive coastal areas where fish are more abundant. Density-dependent catchability has also been shown to have significant influence on catches for some fish species (particularly small pelagics), so values for this parameter were used in the fitting process following Piroddi *et al.* (*in press*).

### 4.2.5 Management scenarios

Different scenarios were chosen to represent the outcome of specific management measures; most have been discussed by stakeholders or already mandated by the federal government. Our model scenarios were run from 1972 to 2040, with all 'policies' implemented from 2010 onwards. The scenarios included in the analysis are:

- *Status quo* (SQ): Fishing effort dynamics were represented according to published sources (see Section 2.2) and 2007 fishing effort levels were maintained for the duration of the model run. For all purposes, this is a 'business as usual' scenario.
- *Government mandate* (GM): Current allowed bycatch levels for marlin and dorado in shark longliners were set to 4% (from 15% and 4%, respectively; INP 2007) in accordance with the government mandate (DOF 2008), with all other parameters held constant. Because this policy is already in place (adequate enforcement is another matter, which will not be dealt with here) this 4% bycatch limit is assumed for all subsequent scenarios.
- *Effort restrictions* (ER): A general sentiment among many recreational fishing operators in BCS is that shark long-liners are mostly responsible for marlin catch declines (see Section 1). Also, many environmental groups have opposed this fishery because of its negative effects on shark populations in the region. In the case of recreational fishing, effort restrictions are very unlikely to take place given the importance of this industry to BCS. However, we have included effort reductions for analytical reasons. In the case of both fisheries, we explore reductions of 20, 35 and 50% of current effort, as well as complete closure (setting effort to zero). We ran all cases separately and kept all other inputs constant.

## 4.3 Results and discussion

### 4.3.1 *Status quo*

Through a process of fitting the initial Ecopath model to available catch and effort data, Ecosim vulnerability parameters were adopted to optimize the fit of estimated and observed landings (Fig. 3). The fitting process is particularly difficult for highly migratory species such as yellowfin tuna, or whose regional abundance is related to environmental drivers in an unclear manner, such as squids, scombrids and small pelagic fish (Lluch *et al.* 2007). However, relative trends for key species, such as sharks, are well represented in the model and allow for further analysis.

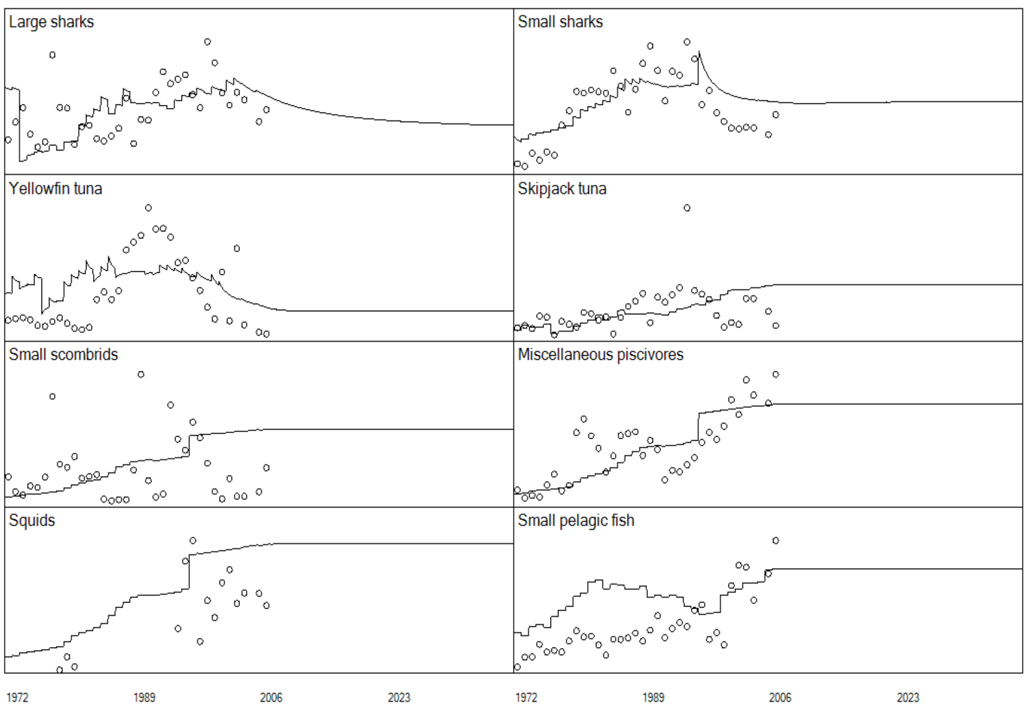


Figure 4.3. Model fit to catch data. Circles represent observed data, lines depict model predictions based on Ecopath parameters and fleet effort;  SSQ= 111.6.

Our model results capture the stock reductions caused by Japanese distant-water fleets, which have been reported in formal stock assessments and are not unique to this region (Ahrens 2010; Myers and Worm 2003). After the large-scale removals of large pelagic predators caused by these fleets operating in the study area since the 1950s, results suggest an initial ‘rebound’ stage after Mexico enforced its EEZ in the late 1970s (Fig. 4.4). By the late 1980s however, this trend was checked and reversed by the growth of Mexico’s own fishing fleets (Fig. 4.2). These effects are reflected in our SQ scenario results (Fig. 4.4), which also provides a tentative forecast of future abundance based on EwE parameterization and current fishing effort held constant to 2040.

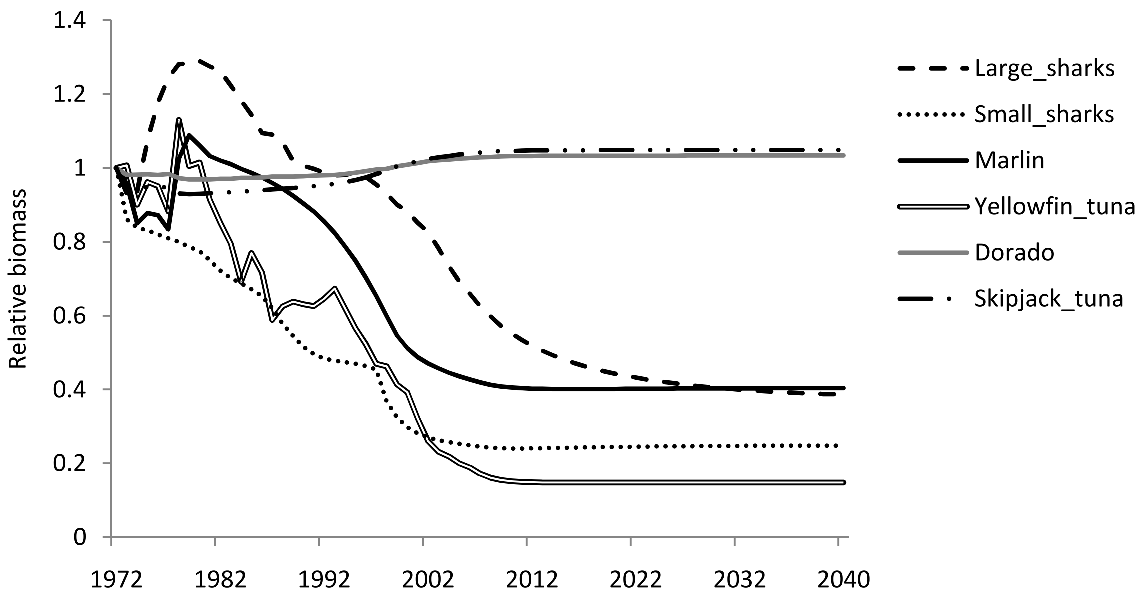


Figure 4.4. Relative biomass of select functional groups from 1972-2010, *Status quo* scenario (see Section 2.3). Groups are large sharks (LSh), small sharks (SSh), marlin (M), dolphins (Dn), yellowfin tuna (YT), dorado (Do) and skipjack tuna (ST).

Both recreational and commercial pelagic fisheries in BCS have drastically increased fishing mortality over time and primarily target large predators (Fig. 4.5). Their removal has acted on all species by reducing overall predation mortality (Fig. 4.6), helping maintain or even slightly increase the relative abundance of prey species which are not fished as intensively (e.g. skipjack tuna, dorado; Fig. 4.4).

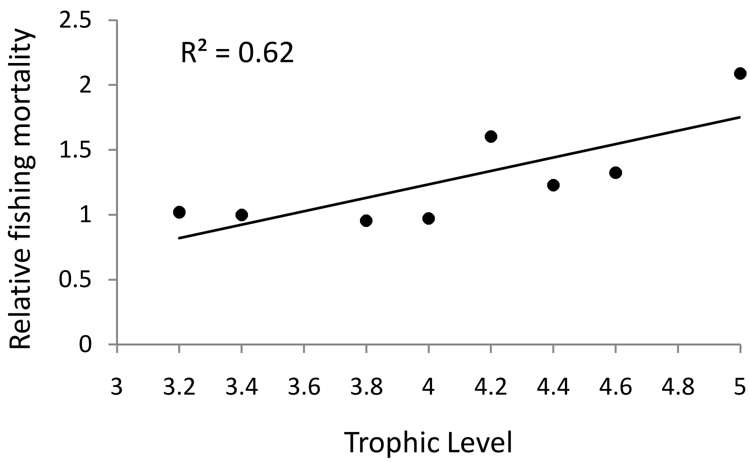


Figure 4.5. Relative change in fishing mortality by trophic level (in 0.2-length bins), 1972-2010.

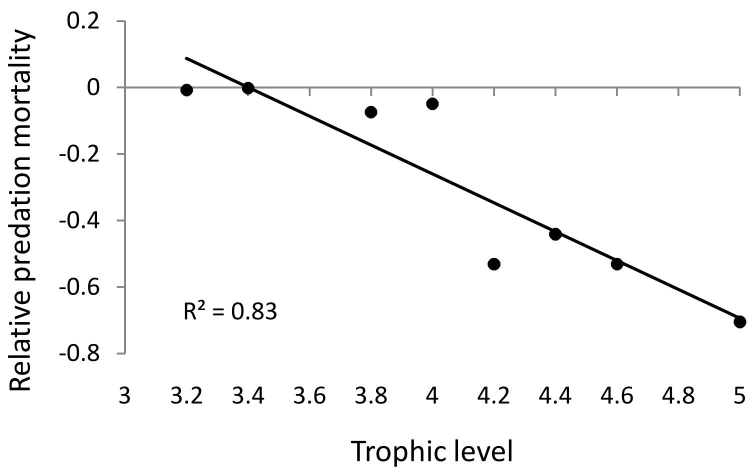


Figure 4.6. Change in predation mortality by trophic level in 2010 relative to 1972 (presented in 0.2-length bins).

Our results match up well with current stock assessments and reports in that, overall, pelagic fisheries are particularly impacting large predatory fishes (Myers and Worm 2003), but current catches in the BCS region are reported to be relatively stable, with some concerns for sharks, in particular (DOF 2004). After intense current and historic fishing pressure, the yellowfin tuna population has been severely depleted, whereas skipjack tuna have remained stable and are an increasingly important component of the overall tuna fishery (Ahrens 2010; IATTC 2009). Because of their relatively low rate of population growth, shark populations are very sensitive to intense fishing pressure. In a study regarding the rebound potential of different shark species, Smith *et al.* (1998) found that coastal sharks are particularly susceptible to fishing effects. In conjunction with the exponential increase in artisanal fishing effort, this supports their relatively higher decline relative to larger oceanic shark species in our model. In the case of striped marlin, which make up over 90% of recreational catches, the BCS population is less abundant but apparently stable, and continues to support a growing recreational fishery (Ortega-Garcia *et al.* 2003).

It has been proposed that the seemingly stable catches in BCS are in fact due to a serial depletion of species which is masked by total landings statistics (DOF 2004; Sala *et al.* 2004). This effect has indeed been observed in many marine fisheries (Pauly *et al.* 1998; Essington *et al.* 2006) and its occurrence in BCS is a possibility which requires pertinent attention. This ecosystem currently supports a commercial capture fishery with an ex-vessel landed value of ~72 million USD (2007) and a recreational fishery which generates ~630 million USD (2007) in retail sales (Southwick Associates Inc. *et al.* 2008). These figures represent different metrics and should not be compared directly, but together illustrate the vital importance of the marine environment to BCS. Results do not suggest that maintaining current fishing pressure would lead to further significant population declines, but historical trends strongly support an increase in fishing capacity, which could have that effect.

### 4.3.2 Government mandate

It is no secret that environmental policy decisions in Mexico (and most of the world, for that matter) are often politically charged (Young 2001), but that does not necessarily preclude their environmental usefulness. We tested the ecosystem effects of currently mandated bycatch limits for the shark longliner fleet (70% reduction in marlin bycatch, see section 4.2.3) over 30 years, assuming that these limits are effectively enforced from 2010 onwards. Under our model assumptions, this policy would result in a marginal (~4%) improvement in marlin abundance compared to the baseline scenario (Fig. 4.7), with almost no other differences save for a slightly smaller increase in skipjack tuna abundance due to increased predation from marlin.

Although our results do not suggest a significant increase of marlin abundance as a result of reducing their bycatch in shark fisheries, it is important to underline other possible benefits of such a measure. Mainly driven by recreational fishing and other forms of ecosystem-based recreation, tourism is the fastest growing economic sector in BCS (López-Espinosa 2002), generating ~15% of BCS GDP (2007), compared with ~2% from commercial fisheries (INEGI 2009). As with other recreational activities, it has been found that a particular site's stance towards and support of recreational fishing is of particular importance for individuals deciding on a destination (Ditton *et al.* 2002). This is of particular relevance for BCS given that the great majority of recreational fishermen in Latin America are foreigners traveling from the US or Canada (Cisneros-Montemayor and Sumaila *in press*), creating much competition between alternative sites. With commercial fisheries in the area already at or above their maximum sustainable yield, it seems logical to provide further support for the recreational sector through investments in infrastructure and public perception. It is of vital importance to recognize, however, that recreational fishing has the potential for all of the same negative impacts as commercial fishing and should be managed with this fact in mind (Coleman *et al.* 2004; Lewin *et al.* 2006).

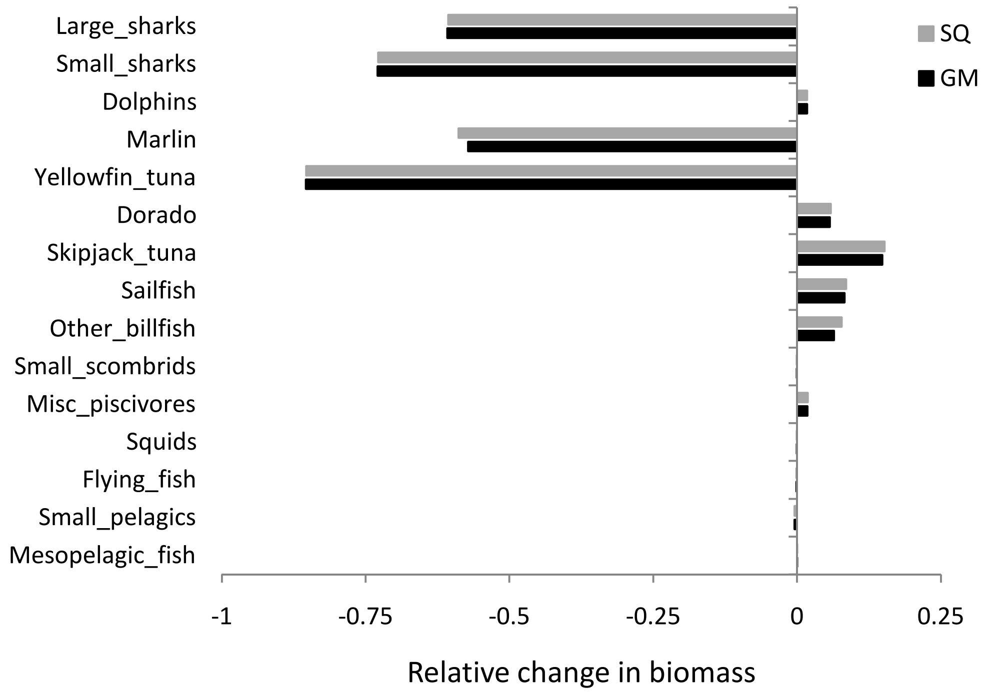


Figure 4.7. Change in biomass of functional groups in 2040, relative to 1972 value, under the *Status quo* (SQ) and *Government mandate* (GM) scenarios (see Section 2.3).

### 4.3.3 Effort restrictions

Implementation of fishery resource policies is normally approached either through input (effort) or output (catch) controls (Walters and Martell 2004). In the case of Mexico and particularly in the Gulf of California (GC) region, an immense coastline with almost limitless landing sites has traditionally limited management to restrictions in effort. We test several scenarios for longlining and recreational fishing, leading to some interesting outcomes. As shown above, pelagic fisheries in BCS target mainly larger predatory fish, so the effects of fleet effort reductions act most strongly upon these top predators, which often engage in both competition and predation with each other.

Results from longlining reductions suggest that large shark populations would indeed benefit, with increasing marginal returns from relative increases in effort reductions (Fig. 4.8, LLR). In the most extreme case (setting effort to zero), large shark biomass stabilized at an increase of 1.2 times the original (2010) biomass. In our model, this result is due to intra-species predation, an effect supported by known opportunistic feeding behavior of large sharks (Motta and Wilga 2001). Increasing shark abundance has negative but relatively small impacts on some of its prey (particularly smaller sharks and billfish), but are somewhat dampened by the fact that some of their main diet items such as small pelagic fish and squids, are very abundant and productive in this region (Lluch-Cota *et al.* 2007). Squid abundance in the GC has an erratic history (Ehrhardt 1991), but seems to be related to ocean climate patterns and has been increasing substantially for the past decade (Nevárez-Martínez *et al.* 2000). Continued increase would boost the ecological importance of squid in this ecosystem, specifically with positive effects in predators such as sharks (Rosas-Luis *et al.* 2008).

One of the main arguments for reducing shark longlining in BCS is its perceived negative effects on marlin populations. Our results show that there are indeed some very slight increases in marlin biomass, although only up until a 20% reduction in longlining effort. Past this point, increased shark abundance leads to slight but *negative* effects on marlin abundance (Fig. 4.8 LLR). It is important to take into account the fact that although sharks and marlin are competing top predators in the pelagic environment, sharks are also a predator of juvenile marlin (Olson and Watters 2003) and thus an increase in shark biomass leads to an increase in marlin predation mortality.

Reducing recreational fishing effort is perhaps the least likely scenario in BCS, although analyzing its effects provides some interesting insights. As with longlining reductions, our data suggests that reductions in recreational fishing effort result in increasing marginal returns, although marlin biomass under no fishing stabilized at 90% increase of the 2010 value (Fig. 4.8, RFR), again due to intra-species predation and a reduction in important prey items such as tuna (Fig. 4.4). Due to their relatively high production and food consumption rates, increases in marlin biomass had a greater (and mostly negative) impact on other species, particularly sharks (a competitor) and tunas and other billfish (prey items) (Fig. 4.8, RFR).

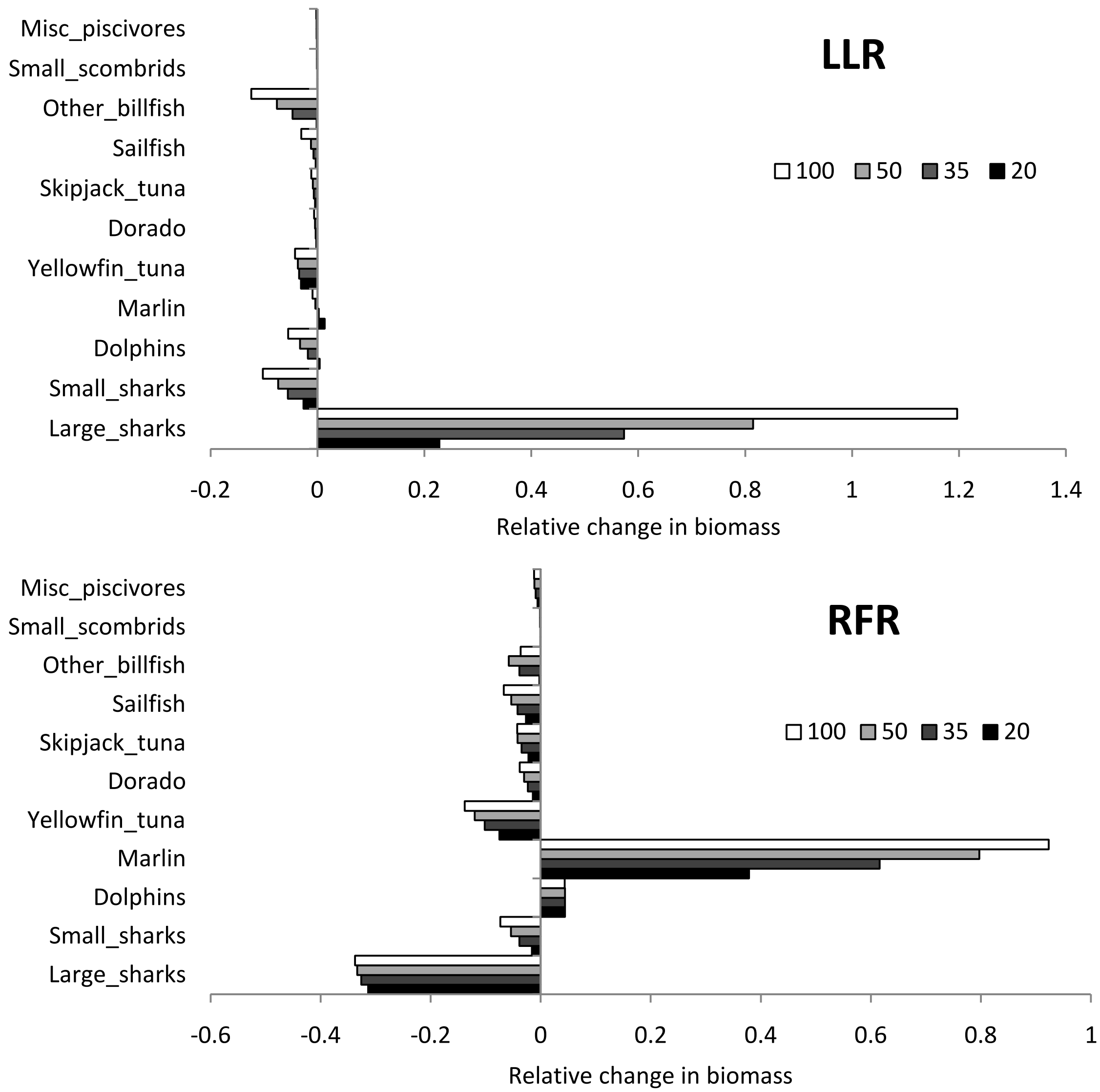


Figure 4.8. Biomass of predatory functional groups in 2040, relative to 2010 SQ value, for effort restrictions to longlining (LLR) and recreational fishing (RFR). Legend denotes percentage of effort reduction (relative to 2010; see section 4.2.3).

These responses in biomass abundance are reflected in catches for BCS fleets (Fig. 4.9). In both cases (longlining and recreational fishing), decreased effort allows for biomass to recover from overfishing and results in an increase in catch. The time it takes for a given population to rebuild depends on the magnitude of effort reduction and the natural population growth rate of the species in question (Fig. 4.10). In our model, given a 35% reduction (which results in the greatest marginal improvement in catch) in longlining and recreational fishing effort (respectively), large sharks would take about 13 years to rebuild to the point that catches exceed those in 2010, compared to about 5 years for marlin. Due to their life history characteristics, these differences in rebuilding time are not unexpected and correspond to those reported by Kitchell *et al.* (2006). However, due to the more region-specific diet composition parameters in our model, our results show a higher impact on other species as a consequence of increased marlin biomass.

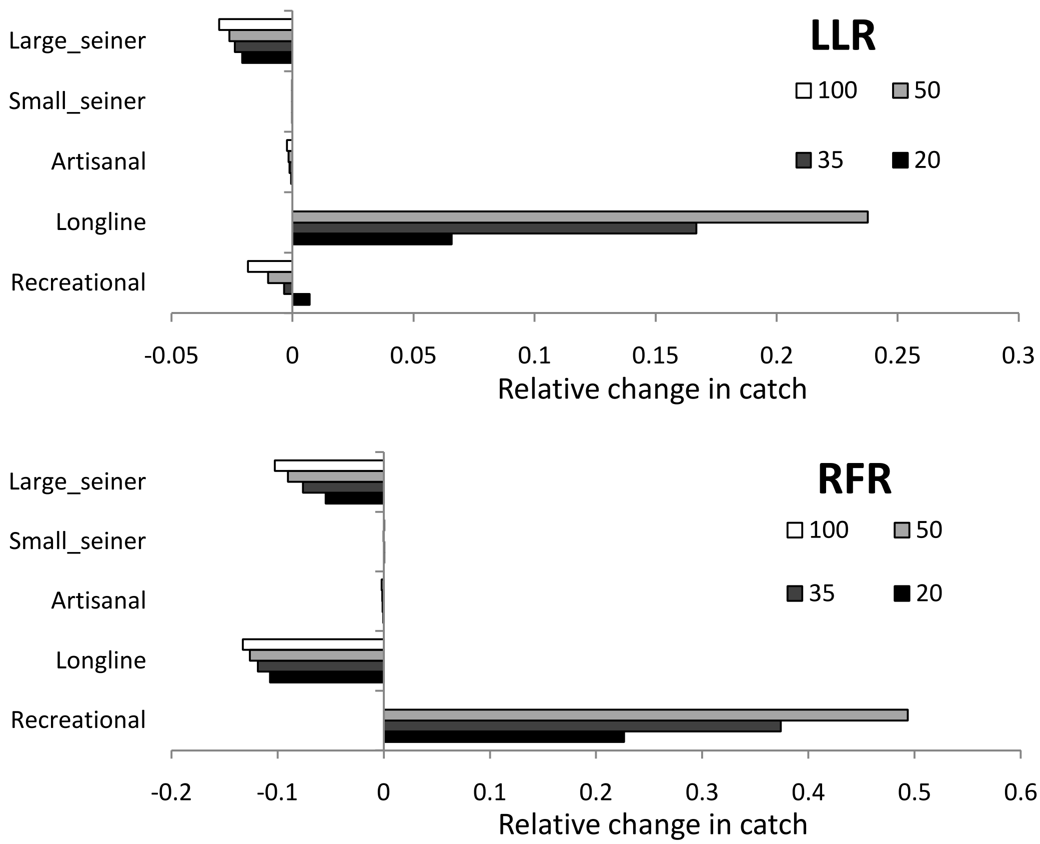


Figure 4.9. Change in catch in 2040 relative to SQ (2040) scenario after effort restrictions to longlining (LLR) and recreational fishing (RFR). Legend denotes percentage of effort reduction (relative to 2010; see section 4.2.3).

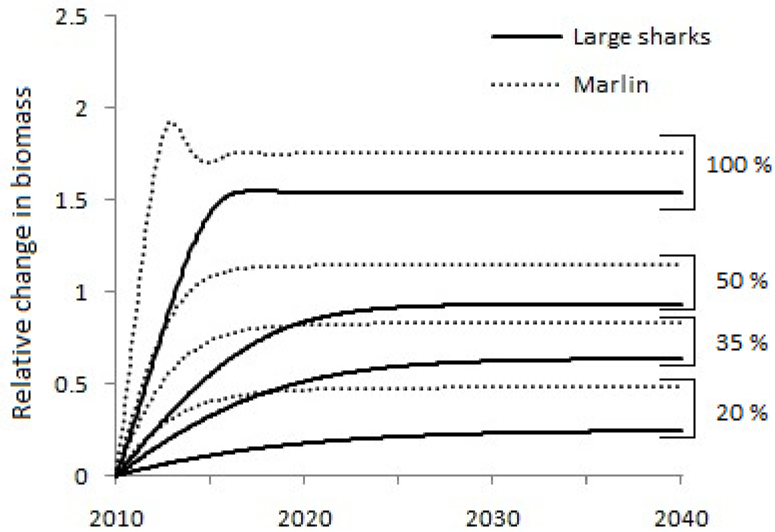


Figure 4.10. Change in biomass of large sharks and marlin relative to 2010 given effort reductions to longlining and recreational fishing fleets, respectively. Percentages denote effort reductions relative to 2010 values.

In the case of the shark fishery, based on government fishery data (CONAPESCA 2007), these reductions would amount to a loss of ~100 jobs and, after stock rebuilding, a net increase in revenue of ~330 thousand USD (2010) per year. Revenue from recreational fishing depends mainly on effort; although catch is indeed a driver of effort, the relationship between the two is highly unclear (Arlinghaus 2006). Assuming constant per capita fishing expenditure by recreational anglers, a 35% reduction in recreational fishing effort would represent a yearly loss of ~200 million USD to the BCS economy, without taking into account the further losses from downstream and upstream economic impacts. Economically, this makes it a very unlikely policy (more like political suicide), although the model results do lend insight into the ecological role of marlin in the ecosystem, which is still unclear (Kitchell *et al.* 2006).

The management analysis capabilities of ecosystem models have slowly begun to be explored, although there is clearly an increasing trend in this regard as ecosystem-based management gains footing in the environmental management discourse (Plaganyi and Butterworth 2004). In our current study we have focused on a specific and grounded problem and provided results which are based on our best understanding of the underlying ecological and fishery parameters and dynamics and on the small amount of varyingly adequate data which all models must deal with (Christensen and Walters 2005). Although we have made our best attempt at using the tools and data at our disposal in an informative analysis, we must stress that results should not be accepted uncritically. In the case of BCS or any other region facing a problem of resource allocation, the best outcomes will always come from scientific understanding geared towards meeting local demands and objectives.

### 4.4 Concluding remarks

While conservation may be a desired goal for any marine ecosystem, fisheries management primarily deals with deriving sustainable human benefits from fish, whatever their use may be. In the case of BCS, maximizing benefits from the commercial and recreational sectors is perceived to be mutually exclusive. Although ecosystem models have yet to be used explicitly for management, we offer a case where they could provide an arena for stakeholders to explore different policy options and evaluate tradeoffs. Our results show that commercial and recreational fisheries are not necessarily at odds, but could probably reach mutually beneficial arrangements. Recognizing potential value in joint cooperation is key for continued development of this region, which has all the potential for the implementation of several parallel sustainable and profitable industries based on the marine environment.

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**Chapter 5**

# 5 Conclusion

## 5.1 Discussion

Ecosystem-based marine recreational activities are a significant worldwide industry, which continues to grow. Their study usually parts from one of two overarching angles: the socio-economic benefits generated by the activity in question (e.g., Toivonen *et al.* 2004), or the drivers of utility derived by individuals (e.g., Arlinghaus 2006). Although results provide insights into the latter discussion, this work is concerned with the former field of study. Based on current available data, it was estimated that 121 million people a year fish for recreation, dive, snorkel, or go whale watching around the world. Their expenditures total approximately 47 billion USD per year, supporting around one million jobs. In contrast, marine capture fisheries employ between 21 and 50 million people worldwide (Berkes *et al.* 2001; Garcia and de Leiva-Moreno 2003), with a global ex-vessel catch value of 80 – 85 billion USD a year (Sumaila *et al.* 2007; World Bank 2008).

While a large part of total benefits from ecosystem-based marine recreation are generated in highly developed countries, developing countries capture a significant portion of these benefits, which may prove to have a larger impact to their regional economies. This is an important finding given that most coastal communities in these developing countries have until now relied mainly on the benefits from commercial fisheries (Pauly 2006), which are in decline and will probably not recover in the near future (Clark 2006).

Recreational fishing is by far the largest industry considered in this thesis, generating over 80% of total expenditures. It is perhaps also the most controversial one because, despite some arguments to the contrary, it has direct and potentially significant negative impacts on the abundance of marine populations (Lewin *et al.* 2006). The two main supporting arguments for its superiority over commercial fishing, namely, that its catches are insignificant relative to commercial fisheries, and that it inherently generates greater total economic benefits, have both been widely contested (e.g., Edwards 1990;

Post *et al.* 2002; Coleman *et al.* 2004). This does not negate the fact that it can provide significant benefits in a sustainable manner, but only reinforces the need for adequate science and management to ensure that this occurs.

In the case of Baja California Sur, the regional significance of benefits from recreational fishing is not in question (Ditton *et al.* 1996; Southwick Associates, Inc. *et al.* 2008). However, the current political climate has been conducive to a very biased approach to resource management in the region (e.g., The Billfish Foundation 2006), which still includes important commercial fisheries. Using an Ecopath with Ecosim model (Christensen and Walters 2004), I explored the results of various mandated and proposed fisheries policies. Results show that, while recreational fishing indeed generates higher direct expenditures than commercial fishing (estimating the total economic contribution of each industry was beyond the scope of this study), currently proposed policies (DOF 2008), might not have the desired effects.

While some reductions in billfish bycatch by shark longliners are beneficial for the marlin population, reducing overall shark fishing effort results in higher abundance of sharks. This may be good for shark conservation, but in an overfished ecosystem, these 'extra' predators might soon start to negatively impact their prey, such as marlin (Olson and Watters 2003). The long-term results of misguided policies can be devastating for commercial fisheries and have marginal benefits or even *negative* impacts on the recreational fishery, a lose-lose scenario. From a bioeconomic perspective, it has been proposed that reducing effort in an overfished system leads to improvements both in the biomass of the target species and on the economic profits of the fleet (Clark 2005). This theoretical principle was simulated in an ecosystem context and generally supported by results. This type of analysis is becoming more feasible thanks to developments in modeling techniques (Christensen and Walters 2005) and hindsight gained from previous management failures (Walters and Martell 2004). Thus, in addition to exploring a particular management issue, the results of this work lend support for the use of ecosystem models for 'virtual' adaptive ecosystem-based management, where measures designed to affect a particular species take ecosystem dynamics into account (Pikitch *et al.* 2004; Christensen and Walters 2005).

Whale watching and diving are two somewhat similar cases in that, in theory, they could be undertaken with minimal negative impacts due to the fact that they profit solely from watching marine life (Holland *et al.* 1998). Probably due to the competitive nature of the diving market, there is little publicly accessible information for this industry. Estimates from available data suggest that around 10 million recreational divers and four times as many snorkelers generate over 5.5 billion USD per year, supporting over 110,000 jobs around the world. There are some ecological concerns regarding the high concentration of divers at certain sites, mainly because of physical damage to coral and rocky reefs (Barker and Roberts 2004), as well as indications of problems related to site carrying capacity of divers (Brander *et al.* 2007). The evident lack of public knowledge on this subject suggests a need for management to pay greater attention to the diving industry, at least to ensure that it is performed within adequate ecological impact and personal safety guidelines.

Whale watching as an industry started in the USA several decades ago, and currently generates over 2.1 billion USD in total expenditures and supports about 13,000 jobs (O'Connor *et al.* 2009). Assuming no increases in the total number of tourist arrivals to a country, the results of this work suggest that an additional 413 million USD and 5,700 jobs could potentially be generated by establishing whale watching operations in countries that do not currently have them. This would bring the total benefits from whale watching to over 2.5 billion USD a year, supporting around 19,000 jobs around the world. About half of these estimated potential benefits would be captured by developing countries.

Results of this work suggest that maintaining a pro-whaling stance and investing in whale watching by a country are apparently not mutually exclusive, although commercial whaling of the sort that occurred in the past will lead to further declines in whale populations (Baker and Clapham 2004), directly and negatively impacting whale watching around the country in question, and the world. Given these conditions, it would be interesting to explore the potential role of international side payments for ecosystem services (Bulte *et al.* 2008), from countries that accrue significant benefits from whale watching to those which do not, as an incentive for conservation (Munro 2008).

Quite logically, almost any coastal country could theoretically engage in ecosystem-based marine recreational activities. For developing countries however, the current niche of these activities as tourism-driven industries implies that they are subject to the influence and constraints of overall tourism (Hoyt 1999; Orams 2002). Together with the fact that social and material infrastructure are a prior necessity (Khadaroo and Seetanah 2008), this is vital to consider when planning for a regional economy based on these activities, whose benefits can be quite substantial but are not guaranteed.

In any case, results of this work show that ecosystem-based marine recreational activities, if developed under strict guidelines and with full consideration of their possible drawbacks, have the potential to provide substantial and sustainable economic benefits. For many coastal communities around the world, these alternative uses of the marine ecosystem are key for maintaining livelihoods in the face of declining fisheries.

**5.2 Strengths, weaknesses, and future work**

In Chapters 2 and 3, I have provided global-scale estimations of marine recreation, which must be broad by definition. Perhaps the single most important potential drawback to any study requiring a model is the nature of the data, a fact which is augmented in large-scale meta-analytical studies. The best way to deal with this uncertainty is to approach the study in a systematic and structured manner which is simple and reproducible (Nelson and Kennedy 2009). I have attempted to do so here by detailing the methods used and providing a list of all primary sources (Appendix A). Even so, meta-analysis is by definition a method to deal with data-poor situations, which means that any estimates must be considered cautiously. It is, however, a well-established and widely used approach to deal with data-poor situations that nonetheless require management advice (Rosenberger and Loomis 2001).

In the case of whale watching, benefits for the year 2003 were estimated based on data and industry trends in prior years (mostly 1991-1998), and resulted in estimates of 13 million whale watchers and 1.6 billion USD (2003) in yearly total expenditure. By comparison, the most recent global survey of whale watching (O'Connor *et al.* 2009) reports yearly totals of 13 million whale watchers and 1.8 billion USD (adjusted to 2003 USD) in total expenditures in 2006-2008. While this is an encouraging result, it points out an overlooked but inherent problem in forecast models in that, when making future projections, any parameters that are not included in the model are assumed to remain constant. In the case of whale watching, tourism growth as a whole slowed due to an economic downturn and worldwide security concerns (World Tourism Organization 2009). Because these factors are often unforeseeable, perhaps the best way to deal with them is to forecast outcomes based on particular scenarios, as is done with climate change or other large-scale effects (Jones 2000).

This type of scenario analysis was used in Chapter 4, which was concerned with possible effects of specific management policies on recreational and commercial fisheries. In this case, the implications or nuances of enforcing particular policies were not tested; rather, these policies were simply assumed to occur. Again, this model is faced with the same problems discussed before in that all unaccounted-for parameters are assumed to remain constant, including the physical environment itself. For future work, it would be interesting to gather enough quantitative data to explore the influence of large-scale ecosystem drivers, such as climate, that occur in parallel to management efforts (Watters *et al.* 2003). An additional parameter to include in future model is the varying appeal of specific animal species to potential users of recreational activities. Not all marine life is created equal when it comes to attracting whale watchers, anglers or divers, and these relationships would be very interesting to explore.

Some concerns have been raised regarding the management-applicability of Ecopath with Ecosim (Planganyi and Butterworth 2004), though these are for the most part due to potential misuse of the program and misinterpretation of results, rather than issues with the program itself. The main drawback to using relatively complex models is the inevitable lack of adequate data (Hilborn and Mangel 1997). In this work, for example, the limited number of local ecological modeling studies made it necessary to use large-scale studies to set up the regional model, under the assumption that the same general relationships are applicable to this ecosystem. The strength of this assumption varies among different parameters and species (Froese and Pauly 2010), but in any case, the regional ecosystem would probably be better represented using in-site biological data. This information is not always readily available, but in some cases is generated as an overlooked byproduct of biological census studies in the area and might be gleaned from those results.

Fisheries data are notoriously suspect (Zeller *et al.* 2007), but were the best available information on the magnitude of resource extraction in that region. In particular, effort data in Mexico are highly dubious; for example, the fisheries management body stopped counting artisanal fishing boats over 10 years ago and assumes the number has remained constant since then (CONAPESCA 2003-2008). In lieu of a better estimate, I included a very conservative 1-2% yearly increase in vessel fishing capacity as a proxy for 'technology creep' (Daniel Pauly, *pers. comm.*). This is a step forward from assuming zero, but is most likely an underestimation. Further work is needed to a establish a better method for evaluating capacity increase, perhaps based on metrics such as boat size, horsepower, winch capacity, or on-board technology such GPS or fish-finders. Also, the assumption that the magnitude of recreational catch does not warrant quantification should be absolutely dispelled (Coleman *et al.* 2004).

### 5.3 Concluding remarks

It is evident from the results of this work that ecosystem-based marine recreation is a globally significant industry. It has proven to be able to generate considerable economic benefits for particular regions or countries. Furthermore, there is still potential for improvement, either through existing operations or by establishing new ones. To ensure that the current and potential benefits from this industry are sustainable, it is vital to approach them as one more form of natural resource use, which has benefits and costs like any other. If conservation is a management objective, further work must be done regarding the negative impacts of these activities, as well as the potential role of side payments, as an incentive for conservation, from countries that benefit significantly from these ecosystem-based activities to those which do not.

There is clearly much left to be done to improve on the different aspects of this work. However, I have made every effort to ensure that, based on available data and through a structured and detailed method, the results provided here are relevant for the current debate on the future of natural resource management in general, and ecosystem-based marine recreation in particular.

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The Billfish Foundation. (2006). Pressure from within the Calderon administration may force changes to shark norma (NOM-029). http://www.billfish.org/new/newsarticle.asp?ArticleID=66 [*Last accessed March 5, 2010*]

Toivonen, A., E. Roth, S. Navrud, G. Gudbergsson, H. Applelblad, B. Bengtsson and P. Tuunainen. 2004. The economic value of recreational fisheries in Nordic Countries. *Fisheries Management and Ecology* 11: 1-14

Walters, C.J. and S.J.D. Martell. (2004). Fisheries Ecology and Management. Princeton University Press, 448 p.

Watters, G.M., Olson, R.J., Francis, R.C., Fiedler, P.C., Polovina, J.J., Reilly, S.B., Aydin, K.Y., Boggs, C.H., Essington, T.E., Walters, C.J. and J.F. Kitchell. (2003). Physical forcing and the dynamics of the pelagic ecosystem in the eastern tropical Pacific: simulations with ENSO-scale and global-warming climate drivers. *Canadian Journal of Fisheries and Aquatic Sciences* 60: 1161-1175

Wells, M. (1992). Biodiversity conservation, affluence and poverty: Mismatched costs and benefits and efforts to remedy them. *Ambio* 21 (3): 237-243

World Bank. (2008). The sunken billions. The economic justification for fisheries reform. Agriculture and Rural Development Department. The World Bank. Washington, D.C.

World Tourism Organization. (2009). UNWTO World Tourism Barometer. Interim update, September 2009. [Available at http://unwto.org/facts/eng/barometer.htm]

Zeller, D., Booth, S., Davis, G. and D. Pauly. (2007). Re-estimation of small-scale fishery catches for U.S. flag-associated island areas in the western Pacific: the last 50 years. *Fisheries Bulletin* 10: 266-277

## Appendix A. Sources for marine recreation database

**Table A1.** Sources for marine recreation database. Source types are 1) peer-reviewed publication; 2) government agency report; 3) government agency website; 4) FAO/UN report; 5) NGO report; 6) newspaper source; 7) commercial or public website.

| Country | Source | Source Type |
|---|---|---|
| Albania | Food and Agriculture Organisation of the United Nations. Fisheries and Aquaculture Department. 2009. Fishery and Aquaculture Country Profile, Albania. http://www.fao.org/fishery/countrysector/FI-CP_AL/en [*Last accessed May 3,2009*] | 4 |
| Algeria | No data found. | |
| Angola | http://www.anglingclassics.co.uk/Angola.html [*Last accessed May 11, 2009*] | 7 |
| Antigua and Barbuda | Food and Agriculture Organisation of the United Nations. Fisheries and Aquaculture Department. 2007. Fishery and Aquaculture Country Profile, Antigua and Barbuda. ftp://ftp.fao.org/FI/DOCUMENT/fcp/en/FI_CP_AG.pdf [*Last accessed May 11, 2009*] | 4 |
| Argentina | Fundación Proteger. 2007. La pesca deportiva es un bloom: la practican mas de 3 millones de Argentinos. http://www.proteger.org.ar/doc654.html [*Last accessed May 3, 2009*] | 5 |
| Australia | Henry, G.W. and J.M. Lyle. 2003. The National Recreational and Indigenous Fishing Survey. Australian Government Department of Agriculture, Fisheries and Forestry, Australia. pp. 188 | 2 |
| Bahamas | http://fishinthebahamas.com/index.php?gclid=CL3z2sTttJoCFRk_awodziKXcg [*Last accessed May 11, 2009*] | 7 |
| Bahrain | http://www.bahrainguide.org/BG3/sportsandleisure.html [*Last accessed May 3, 2009*] | 3 |
| Bangladesh | Food and Agriculture Organisation of the United Nations. Fisheries and Aquaculture Department. 2007. Fishery and Aquaculture Country Profile, People's Republic of Bangladesh. ftp://ftp.fao.org/FI/DOCUMENT/fcp/en/FI_CP_BD.pdf [*Last accessed May 11, 2009*] | 4 |
| Barbados | http://www.barbados.org/fishing.htm [*Last accessed May 11, 2009*] | 7 |
| Belgium | Pawson, M.G., D. Tingley, G. Padda and H. Glenn. 2006. Final Report, EU Contract FISH/2004/011 on "Sport Fisheries" (or Marine Recreational Fisheries) in the EU. European Commission Directorate-General for Fisheries. pp. 213 | 2 |
| Belize | Fedler, A.J. 2008. Economic impact of recreational fishing for bonefish, permit and tarpon in Belize for 2007. Friends of Turneffe Atoll. pp. 26 | 5 |
| | Ibérica de Estudios e Ingeniería S.A. 2007. Proyecto regional: Manejo sostenible de la pesca marina, con énfasis en las especies objetivo de la pesca deportiva. Segundo Informe Intermedio Parte I. Versión final del Diagnóstico. | 2 |
| Benin | No data found. | |

Table A1 (continued).

| Country | Source | Source Type |
|---|---|---|
| Brazil | http://www.sportfishing-brazil.com/ [*Last accessed May 11, 2009*] | 7 |
| Brunei Darussalam | http://www.bruneifishing.com/ [*Last accessed May 11, 2009*] | 7 |
| Bulgaria | http://www.informationbulgaria.com/fishing_in_bulgaria.html [*Last accessed May 11, 2009*] | 3 |
| Cambodia | http://home.earthlink.net/~lasweet1/ [*Last accessed May 11, 2009*] | 7 |
| Cameroon | No data found. | |
| Canada | Department of Fisheries and Oceans Canada. 2007. Survey of Recreational Fishing in Canada 2005. Economic Analysis and Statistics Policy Sector. http://www.dfo-mpo.gc.ca/communic/statistics/recreational/canada/2005/REC2005_EN_20070727.pdf [*Last accessed May 3, 2009*] | 2 |
| Cape Verde | http://www.capeverdeinfo.org.uk/sport_game_fishing.htm [*Last accessed May 11, 2009*] | 7 |
| Chile | Servicio Nacional de Turismo. 2006. Pesca Recreativa Chile. http://www.sernatur.cl /institucional/PDF/estadisticas/tur-interes-especial/pesca-recreativa.pdf [*Last accessed May 3, 2009*] | 2 |
| China Main | Shen, J. 2008. Current status and challenges facing recreational fishing in the People's Republic of China. *In:* Aas, O. (Ed.) Global Challenges in Recreational Fisheries. Blackwell Publishing, Singapore. p. 18-21 | 1 |
| Colombia | http://www.colombia.travel/es/ [*Last accessed May 11, 2009*] | 3 |
| Comoros | http://businessafrica.net/africabiz/countries/comoros.php [*Last accessed May 11, 2009*] | 7 |
| Congo | No data found. | |
| Congo Dem. Rep | http://www.fao.org/fi/oldsite/FCP/en/COD/BODY.HTM [*Last accessed May 11, 2009*] | 4 |
| Costa Rica | Ibérica de Estudios e Ingeniería S.A. 2007. Proyecto regional: Manejo sostenible de la pesca marina, con énfasis en las especies objetivo de la pesca deportiva. Segundo Informe Intermedio Parte I. Versión final del Diagnóstico. | 2 |
| Cote d'Ivoire | No data found. | |
| Croatia | Food and Agriculture Organisation of the United Nations. Fisheries and Aquaculture Department. 2007. Fishery and Aquaculture Country Profile, Croatia. ftp://ftp.fao.org/FI/DOCUMENT/fcp/en/FI_CP_HR.pdf [*Last accessed May 11, 2009*] | 4 |

Table A1 (continued).

| Country | Source | Source Type |
|---|---|---|
| Cuba | Baños-González, R., Pacheco-Roberto, J. and Casas-Corrales, W. 2003. Caracterización integral de los sitios de buceo en Cayo Levisa para la mejora y sostenibilidad de su producto. Simposio Internacional de Calidad. La Habana, October 2003. | 1 |
| Cyprus | Hadjistephanou, N and L. Vassiliades. 2004. The present status of fishery and information system in Cyprus. GCP/INT/918/EC - TCP/INT/2904/TD-4.2. MedFisis Technical Document No. 4.2: 55pp. | 2 |
| Denmark | Pawson, M.G., D. Tingley, G. Padda and H. Glenn. 2006. Final Report, EU Contract FISH/2004/011 on "Sport Fisheries" (or Marine Recreational Fisheries) in the EU. European Commission Directorate-General for Fisheries. pp. 213 | 2 |
| Djibouti | No data found. | |
| Dominican Republic | http://www.godominicanrepublic.com/ [*Last accessed May 11, 2009*] | 3 |
| Dominica | http://www.dominica-weekly.com/information/island-style-fishing-at-its-best/ [*Last accessed May 11, 2009*] | 7 |
| Ecuador | http://www.ecuadorboutiquetravel.com/fishingtours [*Last accessed May 11, 2009*] | 7 |
| Egypt | Food and Agriculture Organisation of the United Nations. Fisheries and Aquaculture Department. 2003. Fishery and Aquaculture Country Profile, Egypt. http://www.fao.org/fishery/countrysector/FI-CP_EG [*Last accessed May 11, 2009*] | 4 |
| El Salvador | Food and Agriculture Organisation of the United Nations. Fisheries and Aquaculture Department. 2003. Fishery and Aquaculture Country Profile, El Salvador.<br>http://www.fao.org/fishery/countrysector/FI-CP_SV/es [*Last accessed May 11, 2009*] | 4 |
| Equatorial Guinea | No data found. | |
| Eritrea | No data found. | |
| Estonia | Food and Agriculture Organisation of the United Nations. Fisheries and Aquaculture Department. 2003. Fishery and Aquaculture Country Profile, Estonia.<br>http://www.fao.org/fishery/countrysector/FI-CP_EE/en [*Last accessed May 11, 2009*] | 4 |

Table A1 (continued).

| Country | Source | Source type |
|---|---|---|
| Fiji | http://www.sportfishingfiji.com/ [*Last accessed May 11, 2009*] | 7 |
| Finland | Toivonen, A. 2008. Recreational fishing in Finland. *In:* Global challenges in recreational fisheries. Aas, O. (Ed.) Blackwell Publishing, Singapore. p. 21-25 | 1 |
| | Toivonen, A., E. Roth, S. Navrud, G. Gudbergsson, H. Applelblad, B. Bengtsson and P. Tuunainen. 2004. The economic value of recreational fisheries in Nordic Countries. *Fisheries Management and Ecology* 11, 1-14 | 1 |
| | Pawson, M.G., D. Tingley, G. Padda and H. Glenn. 2006. Final Report, EU Contract FISH/2004/011 on "Sport Fisheries" (or Marine Recreational Fisheries) in the EU. European Commission Directorate-General for Fisheries. pp. 213 | 2 |
| France | Institut Français de Recherche pour l'Exploitation de la Mer. 2007 Enquete relative a la peche de loisir (recreative et sportive) en mer en Metropole et dans les DOM. http://agriculture.gouv.fr/sections/magazine/dossiers/littoral-peche-loisir [*Last accessed May 3, 2009*] | 2 |
| Gabon | Food and Agriculture Organisation of the United Nations. Fisheries and Aquaculture Department. 2007. Fishery and Aquaculture Country Profile, Gabon. ftp://ftp.fao.org/FI/DOCUMENT/fcp/fr/FI_CP_GA.pdf [*Last accessed May 11, 2009*] | 4 |
| Gambia | Food and Agriculture Organisation of the United Nations. Fisheries and Aquaculture Department. 2007. Fishery and Aquaculture Country Profile, Gambia. ftp://ftp.fao.org/FI/DOCUMENT/fcp/en/FI_CP_GM.pdf [*Last accessed May 11, 2009*] | 4 |
| Georgia | No data found. | |
| Germany | Arlinghaus, R. 2008. The social and economic significance of recreational fishing in Germany. *In:* Aas, O. (Ed.) Global Challenges in Recreational Fisheries. Blackwell Publishing, Singapore. p. 25-30 | 1 |
| Ghana | http://www.ghanablues.com/faq.html [*Last accessed May 11, 2009*] | 7 |
| Greece | Anagnopoulos, N., Papaconstantinou, K., Oikonomou, A., Fragoudes, K., Papaharisis, L., Papachristou, E., Pappa, D., Lousi, M., Cingolani, N., Belardinelli, A., Santojanni, A., Colella, S., Donato, F., Kavathas, S., Penna, R. and C. Sdogati. 1996. Sport fisheries in Eastern Mediterranean. Project 96/018. European Union. | 2 |
| Grenada | Food and Agriculture Organisation of the United Nations. Fisheries and Aquaculture Department. 2007. Fishery and Aquaculture Country Profile, Grenada. ftp://ftp.fao.org/FI/DOCUMENT/fcp/en/FI_CP_GD.pdf [*Last accessed May 11, 2009*] | 4 |

Table A1 (continued).

| Country | Source | Source Type |
|---|---|---|
| Guatemala | Ibérica de Estudios e Ingeniería S.A. 2007. Proyecto regional: Manejo sostenible de la pesca marina, con énfasis en las especies objetivo de la pesca deportiva. Segundo Informe Intermedio Parte I. Versión final del Diagnóstico. | 2 |
| Guinea | No data found. | |
| Guinea-Bissau | http://www.worldsportfishing.com/guinea_bissau_flylure.htm [*Last accessed May 11, 2009*] | 7 |
| Guyana | http://www.kaieteurnews.com/2008/10/15/sport-fishing-industry-to-be-developed-in-guyana/ [*Last accessed May 11, 2009*] | 6 |
| Haiti | No data found. | |
| Honduras | Ibérica de Estudios e Ingeniería S.A. 2007. Proyecto regional: Manejo sostenible de la pesca marina, con énfasis en las especies objetivo de la pesca deportiva. Segundo Informe Intermedio Parte I. Versión final del Diagnóstico. | 2 |
| Hong Kong | http://www.tackletour.com/reviewHK.html [*Last accessed May 11, 2009*] | 7 |
| Iceland | Agnarsson, S., Radford, A. and G. Riddington. 2008. Economic impact of angling in Scotland and Iceland. *In:* Aas, O. (Ed.) Global Challenges in Recreational Fisheries. Blackwell Publishing, Singapore. p. 188-201 | 1 |
| | Toivonen, A., E. Roth, S. Navrud, G. Gudbergsson, H. Applelblad, B. Bengtsson and P. Tuunainen. 2004. The economic value of recreational fisheries in Nordic Countries. *Fisheries Management and Ecology* 11, 1-14 | 1 |
| India | http://www.gamefishingindia.com/ [*Last accessed May 11, 2009*] | 7 |
| Indonesia | http://www.taka-adventure.com/ [*Last accessed May 11, 2009*] | 7 |
| Iran | Food and Agriculture Organisation of the United Nations. Fisheries and Aquaculture Department. 2007. Fishery and Aquaculture Country Profile, Iran (Islamic Republic of). http://www.fao.org/fishery/countrysector/FI-CP_IR/en [*Last accessed May 11, 2009*] | 4 |
| Ireland | Marine Institute. 2003. A national survey of water-based leisure activities in Ireland, 2003. Marine Institute, Galway Technology Park, Ireland. http://www.marine.ie/NR/rdonlyres/2A571A28-486D-4CA5-B697-7D796AD31AAA/0/SurveyofWaterBasedLeisure.pdf [*Last accessed May 7, 2009*] | 2 |
| Israel | Food and Agriculture Organisation of the United Nations. Fisheries and Aquaculture Department. 2007. Fishery and Aquaculture Country Profile, Israel. ftp://ftp.fao.org/FI/DOCUMENT/fcp/en/FI_CP_IL.pdf [*Last accessed May 11, 2009*] | 4 |

Table A1 (continued).

| Country | Source | Source Type |
|---|---|---|
| Italy | Pawson, M.G., D. Tingley, G. Padda and H. Glenn. 2006. Final Report, EU Contract FISH/2004/011 on "Sport Fisheries" (or Marine Recreational Fisheries) in the EU. European Commission Directorate-General for Fisheries. pp. 213 | 2 |
| | Anagnopoulos, N., Papaconstantinou, K., Oikonomou, A., Fragoudes, K., Papaharisis, L., Papachristou, E., Pappa, D., Lousi, M., Cingolani, N., Belardinelli, A., Santojanni, A., Colella, S., Donato, F., Kavathas, S., Penna, R. and C. Sdogati. 1996. Sport fisheries in Eastern Mediterranean. Project 96/018. European Union. | 2 |
| Jamaica | Food and Agriculture Organisation of the United Nations. Fisheries and Aquaculture Department. 2009. Fishery and Aquaculture Country Profile, Jamaica. http://www.fao.org/fishery/countrysector/FI-CP_JM/en [*Last accessed May 11, 2009*] | 4 |
| Japan | Statistics department. Ministry of Agriculture, Forestry and Fisheries. 2007. The 81th [*sic*] Statistical Yearbook of Ministry of Agriculture, Forestry and Fisheries. Japan. Tokyo, 828 p. | 2 |
| Jordan | No data found. | |
| Kenya | Abuodha, P. 1999. Status and trends in Kenyan Recreational Marine Fisheries. *In:* Pitcher, T. (Ed.) Evaluating the Benefits of Recreational Fisheries. Fisheries Center Research Reports. 7 (2): 46-50 | 1 |
| Kiribati | http://www.visit-kiribati.com/kiribati/export/sites/KTO/attractions/sport_and_recreation.html [*Last accessed May 11, 2009*] | 3 |
| Korea, Rep. of | Cheong, S. 2005. Korean fishing communities in transition: limitations of community-based resource management. *Environment and Planning A*. 37: 1277-1290 | 1 |
| Kuwait | Food and Agriculture Organisation of the United Nations. Fisheries and Aquaculture Department. 2009. Fishery and Aquaculture Country Profile, Kuwait. http://www.fao.org/fishery/countrysector/FI-CP_KW/en [*Last accessed May 11, 2009*] | 4 |
| Latvia | http://www.celotajs.lv/cont/cntr/acti/fishing_en.html [*Last accessed May 11, 2009*] | 7 |
| Lebanon | Food and Agriculture Organisation of the United Nations. Fisheries and Aquaculture Department. 2009. Fishery and Aquaculture Country Profile, Lebanon. http://www.fao.org/fishery/countrysector/naso_lebanon/en [*Last accessed May 11, 2009*] | 4 |

Table A1 (continued).

| Country | Source | Source Type |
|---|---|---|
| Liberia | Food and Agriculture Organisation of the United Nations. Fisheries and Aquaculture Department. 2009. Fishery and Aquaculture Country Profile, Liberia. http://www.fao.org/fishery/countrysector/FI-CP_LR/en [*Last accessed May 11, 2009*] | 4 |
| Libya | No data found. | |
| Lithuania | Domarkas, A. and E. Radaityte. 2008. Recreational fisheries in Lithuania: putting Lithuania on the recreational fishing map in Europe. *In:* Aas, O. (Ed.) Global Challenges in Recreational Fisheries. Blackwell Publishing, Singapore. p. 30-34 | 1 |
| Madagascar | Food and Agriculture Organisation of the United Nations. Fisheries and Aquaculture Department. 2008. Fishery and Aquaculture Country Profile, Madagascar. ftp://ftp.fao.org/FI/DOCUMENT/fcp/en/FI_CP_MG.pdf [*Last accessed May 11, 2009*] | 4 |
| Malaysia | Food and Agriculture Organisation of the United Nations. Fisheries and Aquaculture Department. 2009. Fishery and Aquaculture Country Profile, Malaysia. http://www.fao.org/fishery/countrysector/FI-CP_MY/en [*Last accessed May 11, 2009*] | 4 |
| Maldives | http://maldanglers.tripod.com/ [*Last accessed May 11, 2009*] | 7 |
| Malta | Food and Agriculture Organisation of the United Nations. Fisheries and Aquaculture Department. 2009. Fishery and Aquaculture Country Profile, Malta. http://www.fao.org/fishery/countrysector/FI-CP_MT/en [*Last accessed May 11, 2009*] | 4 |
| Marshall Islands | http://www.visitmarshallislands.com/activities/sportfishing.htm [*Last accessed May 11, 2009*] | 7 |
| Mauritania | No data found. | |
| Mauritius | Food and Agriculture Organisation of the United Nations. Fisheries and Aquaculture Department. 2009. Fishery and Aquaculture Country Profile, Malta. http://www.fao.org/fishery/countrysector/FI-CP_MU/en [*Last accessed May 11, 2009*] | 4 |
| Mexico | Secretaria de Turismo. 2000. Comportamiento y tendencias de la pesca deportivo recreativa en México. Resumen Ejecutivo. http://www.sectur.gob.mx/wb/sectur/sect_9287_comportamiento_y_ten | 2 |
| | Southwick Associates, Inc., Nelson Consulting, Inc. and Firmus Consulting. 2008. The economic contributions of anglers to the Los Cabos economy. The Billfish Foundation. | 5 |

Table A1 (continued).

| Country | Source | Source Type |
|---|---|---|
| Micronesia | http://micronesia.hawaii.com/fsm/chuuk/activities/index.php [*Last accessed May 11, 2009*] | 3 |
| Morocco | Food and Agriculture Organisation of the United Nations. Fisheries and Aquaculture Department. 2009. Fishery and Aquaculture Country Profile, Morocco. http://www.fao.org/fishery/countrysector/FI-CP_MA/fr [*Last accessed May 11, 2009*] | 4 |
| Mozambique | http://www.macuacuane.com/ [*Last accessed May 11, 2009*] | 7 |
| Myanmar | No data found. | |
| Namibia | Barnes, J.I., F. Zeybrandt, C.H. Kirchner and A.L. Sakko. 2002. The economic value of Namibia's recreational shore fishery: A review. DEA Research Discussion Paper. Number 50. pp .21 | 1 |
| Nauru | No data found. | |
| Netherlands | Smit, M., de Vos, B. and J.W. de Wilde. 2004. De economische betekenis van de sportvisserij in Nederland. Den Haag, LEI Rapport 2.04.05 | 2 |
| New Zealand | Sport and Recreation New Zealand. 2008. Sport, recreation and physical activity participation among New Zealand adults: Key results of the 2007/2008 Active NZ Survey. Wellington: SPARC | 2 |
| | Wheeler, S. and R. Damania. 2001. Valuing New Zealand recreational fishing and an assessment of the validity of the contingent valuation estimates. *The Australian Journal of Agricultural and Resource Economics*, 45:4, pp. 599-621 | 1 |
| Nicaragua | Ibérica de Estudios e Ingeniería S.A. 2007. Proyecto regional: Manejo sostenible de la pesca marina, con énfasis en las especies objetivo de la pesca deportiva. Segundo Informe Intermedio Parte I. Versión final del Diagnóstico. | 2 |
| Nigeria | http://www.onlinenigeria.com/travel/index.asp [*Last accessed May 11, 2009*] | 3 |
| Norway | Toivonen, A., E. Roth, S. Navrud, G. Gudbergsson, H. Applelblad, B. Bengtsson and P. Tuunainen. 2004. The economic value of recreational fisheries in Nordic Countries. *Fisheries Management and Ecology* 11, 1-14 | 1 |
| Oman | No data found. | |
| Pakistan | Food and Agriculture Organisation of the United Nations. Fisheries and Aquaculture Department. 2009. Fishery and Aquaculture Country Profile, Pakistan. ftp://ftp.fao.org/FI/DOCUMENT/fcp/en/FI_CP_PK.pdf [*Last accessed May 11, 2009*] | 4 |

Table A1 (continued).

| Country | Source | Source Type |
|---|---|---|
| Palau | Food and Agriculture Organisation of the United Nations. Fisheries and Aquaculture Department. 2009. Fishery and Aquaculture Country Profile, Palau. http://www.fao.org/fishery/countrysector/FI-CP_PW/en [*Last accessed May 11, 2009*] | 4 |
| Panama | Ibérica de Estudios e Ingeniería S.A. 2007. Proyecto regional: Manejo sostenible de la pesca marina, con énfasis en las especies objetivo de la pesca deportiva. Segundo Informe Intermedio Parte I. Versión final del Diagnóstico. | 2 |
| Papua New Guinea | Food and Agriculture Organisation of the United Nations. Fisheries and Aquaculture Department. 2009. Fishery and Aquaculture Country Profile, Papua New Guinea. http://www.fao.org/fishery/countrysector/FI-CP_PG/en [*Last accessed May 11, 2009*] | 4 |
| Peru | Food and Agriculture Organisation of the United Nations. Fisheries and Aquaculture Department. 2009. Fishery and Aquaculture Country Profile, Peru. http://www.fao.org/fishery/countrysector/FI-CP_PE/en [*Last accessed May 11, 2009*] | 4 |
| Philippines | http://pgff.org/ [*Last accessed May 11, 2009*] | 7 |
| Poland | Wolos, A., Mioduszewska, H., and H.L. Shramm Jr. 2008. Socio-economic analysis of competitive fishing in Poland. *In:* Aas, O. (Ed.) Global Challenges in Recreational Fisheries. Blackwell Publishing, Singapore. p. 249-254 | 1 |
| Portugal | http://www.reefcatfishing.com/ [*Last accessed May 11, 2009*] | 7 |
| Qatar | Food and Agriculture Organisation of the United Nations. Fisheries and Aquaculture Department. 2007. Fishery and Aquaculture Country Profile, Qatar. | 4 |
| | http://www.fao.org/fishery/countrysector/FI-CP_QA/en [*Last accessed May 11, 2009*] | |
| Romania | No data found. | |
| Russian Feds | Food and Agriculture Organisation of the United Nations. Fisheries and Aquaculture Department. 2007. Fishery and Aquaculture Country Profile, Russian Federation. | 4 |
| | ftp://ftp.fao.org/FI/DOCUMENT/fcp/en/FI_CP_RU.pdf [*Last accessed May 11, 2009*] | |
| Saint Kitts and Nevis | Food and Agriculture Organisation of the United Nations. Fisheries and Aquaculture Department. 2009. Fishery and Aquaculture Country Profile, Saint Kitts and Nevis. | 4 |
| | http://www.fao.org/fishery/countrysector/FI-CP_KN/en [*Last accessed May 11, 2009*] | |

Table A1 (continued).

| Country | Source | Source Type |
|---|---|---|
| Saint Lucia | Mohammed, and Joseph. 2003. St. Lucia, Eastern Caribbean: Reconstructed fisheries catches and fisheries effort, 1942-2001. Fisheries Center Research Reports. Vol. 11 (6), 21-43 | 1 |
| Saint Vincent and the Grenadines | http://www.svgtourism.com/articles/detail/detail1.asp?id=135&archive=1 [*Last accessed May 11, 2009*] | 3 |
| Samoa (Western) | No data found. | |
| Sao Tome and Principe | Food and Agriculture Organisation of the United Nations. Fisheries and Aquaculture Department. 2009. Fishery and Aquaculture Country Profile, Sao Tome and Principe. | 4 |
| | http://www.fao.org/fishery/countrysector/FI-CP_KN/en ftp://ftp.fao.org/FI/DOCUMENT/fcp/es/FI_CP_ST.pdf [*Last accessed May 11, 2009*] | |
| Saudi Arabia | Food and Agriculture Organisation of the United Nations. Fisheries and Aquaculture Department. 2009. Fishery and Aquaculture Country Profile, Saudi Arabia. | 4 |
| | http://www.fao.org/fishery/countrysector/FI-CP_SA/en [*Last accessed May 11, 2009*] | |
| Senegal | http://www.senegal.co.uk/Docs/Senegal-Holidays/Fishing/Default.aspx [*Last accessed May 11, 2009*] | 7 |
| Seychelles | Food and Agriculture Organisation of the United Nations. Fisheries and Aquaculture Department. 2009. Fishery and Aquaculture Country Profile, Seychelles. | 4 |
| | http://www.fao.org/fishery/countrysector/FI-CP_SC/en [*Last accessed May 11, 2009*] | |
| Sierra Leone | Food and Agriculture Organisation of the United Nations. Fisheries and Aquaculture Department. 2008. Fishery and Aquaculture Country Profile, Sierra Leone. | 4 |
| | ftp://ftp.fao.org/FI/DOCUMENT/fcp/en/FI_CP_SL.pdf [*Last accessed May 11, 2009*] | |
| Singapore | No data found. | |
| Solomon Islands | http://www.visitsolomons.com.sb/index.php?option=com_content&view=category &layout=blog&id=65&Itemid=137 [*Last accessed May 11, 2009*] | 3 |
| Somalia | No data found. | |

Table A1 (continued).

| Country | Source | Source Type |
|---|---|---|
| South Africa | Pradervand, P. and R. van der Elst. 2008. Assesment of the charter-boat fishery in KwaZulu-Natal, South Africa. *African Journal of Marine Science*, 30 (1): 101-112 | 1 |
| Spain | Ministerio de Agricultura, Pesca y Alimentacion. Secretaria General de Pesca Maritima. 2003. Estudio socioeconomico de la pesca recreativa del Mediterraneo Español. http://www.pescaresponsable.es/docs/socio_economicos_1.pdf [*Last accessed May 3, 2009*] | 2 |
| Sudan | Food and Agriculture Organisation of the United Nations. Fisheries and Aquaculture Department. 2008. Fishery and Aquaculture Country Profile, Republic of the Sudan. ftp://ftp.fao.org/FI/DOCUMENT/fcp/en/FI_CP_SD.pdf [*Last accessed May 11, 2009*] | 4 |
| Suriname | Food and Agriculture Organisation of the United Nations. Fisheries and Aquaculture Department. 2008. Fishery and Aquaculture Country Profile, Suriname. ftp://ftp.fao.org/FI/DOCUMENT/fcp/en/FI_CP_SR.pdf [*Last accessed May 11, 2009*] | 4 |
| Sweden | Toivonen, A., E. Roth, S. Navrud, G. Gudbergsson, H. Applelblad, B. Bengtsson and P. Tuunainen. 2004. The economic value of recreational fisheries in Nordic Countries. *Fisheries Management and Ecology* 11, 1-14 | 1 |
| Syria | Food and Agriculture Organization of the United Nations. 2007. Fishery Country Profile. The Syrian Arab Republic. ftp://ftp.fao.org/FI/DOCUMENT/fcp/en/FI_CP_SY.pdf [*Last accessed May 2,2009*] | 4 |
| Taiwan | http://www.tackletour.com/reviewfishingtw.html [*Last accessed May 11,2009*] | 7 |
| Tanzania | http://lathamisland.com/record/recordaug05.html [*Last accessed May 11,2009*] | 7 |
| Thailand | http://www.tourismthailand.org/festival-event/content-5902.html [*Last accessed May 2,2009*] | 3 |
| Togo | No data found. | |
| Tonga | Food and Agriculture Organization of the United Nations. 2009. Fishery Country Profile, Tonga. http://www.fao.org/fishery/countrysector/FI-CP_TO/en [*Last accessed May 11,2009*] | 4 |

Table A1 (continued).

| **Country** | **Source** | **Source Type** |
|---|---|---|
| Trinidad and Tobago | Mike, A. and I.G. Cowx. 1996. A preliminary appraisal of the contribution of recreational fishing to the fisheries sector in north-west Trinidad. *Fisheries Management and Ecology* 3: 219-228 | 1 |
| Tunisia | | |
| Turkey | Food and Agriculture Organization of the United Nations. 2007. Fishery Country Profile, Republic of Turkey. ftp://ftp.fao.org/FI/DOCUMENT/fcp/en/FI_CP_TR.pdf [*Last accessed May 11,2009*] | 4 |
| United Arab Emirates | No data found. | |
| United Kingdom | Agnarsson, S., Radford, A. and G. Riddington. 2008. Economic impact of angling in Scotland and Iceland. *In:* Aas, O. (Ed.) Global Challenges in Recreational Fisheries. Blackwell Publishing, Singapore. p. 188-201 | 1 |
| | Crabtree, B., Willis, K, Powe, N., Carman, P., Rowe, D., MacDonald, D. and Usher-Benwell, Y. 2004. *Research into the economic contribution of sea angling.* Final report to UK Department for Environment Food and Rural Affairs, March 2004, 71 pp. plus 7 annexes. | 2 |
| Ukraine | Food and Agriculture Organization of the United Nations. 2009. Fishery Country Profile, Ukraine. http://www.fao.org/fishery/countrysector/FI-CP_UA/en [*Last accessed May 11,2009*] | 4 |
| United States | Gentner, B. and S. Steinback. 2008. The economic contribution of marine angler expenditures in the United States, 2006. U.S. Dep. Commerce, NOAA Tech. Memo, NMFS-FSPO-94, 301 p.<br>U.S. Department of the Interior, Fish and Wildlife Service and U.S. Department of Commerce, Bureau of the Census. 2006. National Survey of Fishing, Hunting and Wildlife-Associated Recreation. | 2 |
| Uruguay | http://uruguay.pordescubrir.com/2009/01/19-la-pesca-deportiva-en-uruguay.html [*Last accessed May 11,2009*] | 7 |
| Vanuatu | http://www.govanuatu.com/fishing.html [*Last accessed May 11,2009*] | 7 |
| Venezuela | Food and Agriculture Organization of the United Nations. 2009. Fishery Country Profile, Venezuela. http://www.fao.org/fishery/countrysector/FI-CP_VE/es [*Last accessed May 11,2009*] | 4 |
| Viet Nam | Food and Agriculture Organization of the United Nations. 2009. Fishery Country Profile, Viet Nam. http://www.fao.org/fishery/countrysector/FI-CP_VN/en [*Last accessed May 11,2009*] | 4 |
| Yemen | No data found. | |

## Appendix B. Ecopath input parameters

Table B1. Initial parameters for groups in the pelagic ecosystem model of Baja California Sur.

| Group name | Taxon | TL | B ($t/km^2$) | P/B (/year) | Q/B (/year) | EE |
|---|---|---|---|---|---|---|
| Large sharks | Alopiidae, Carcharhinidae, Sphyrnidae | 4.5 | 0.06 | 0.32 | 7.81 | 0.71 |
| Small sharks | Carcharhinidae, Mustelidae | 4.4 | 0.05 | 0.58 | 9.16 | 0.43 |
| Dolphins | Delphinidae | 4.4 | 0.12 | 0.04 | 16.50 | 0.61 |
| Marlin | *T. audax*, *Makaira spp.* | 4.9 | 0.14 | 0.60 | 8.00 | 0.58 |
| Yellowfin tuna | *T. albacares* | 4.2 | 0.07 | 1.54 | 17.63 | 0.80 |
| Dorado | *C. hippurus* | 4.3 | 0.18 | 3.00 | 20.39 | 0.42 |
| Skipjack tuna | *K. pelamis* | 4.2 | 0.18 | 1.90 | 20.52 | 0.96 |
| Sailfish | *I. platypterus* | 4.4 | 0.08 | 0.68 | 7.00 | 0.76 |
| Other billfish | Istiophoridae, Nematistiidae | 4.4 | 0.06 | 0.49 | 5.00 | 0.42 |
| Small scombrids | Scombridae | 3.8 | 2.68 | 2.00 | 10.00 | 0.30 |
| Misc. piscivores | Carangidae | 4.0 | 0.23 | 2.40 | 7.70 | 0.93 |
| Squids | *Loligo spp.*, *D. gigas* | 3.7 | 7.39 | 3.16 | 19.27 | 0.86 |
| Flying fish | Exocoetidae | 3.3 | 1.40 | 2.61 | 19.68 | 0.82 |
| Small pelagic fish | Clupeidae, Engraulidae | 3.1 | 15.99 | 3.31 | 10.56 | 0.84 |
| Mesopelagic fish | Myctophidae | 3.3 | 13.43 | 1.99 | 9.27 | 0.95 |
| Zooplankton | | 2.3 | 26.37 | 34.00 | 97.00 | 0.99 |
| Phytoplankton | | 1 | 31.62 | 132 | | 0.50 |
| Detritus | | 1 | 1 | | | |

Table B2. Diet matrix for groups in the pelagic ecosystem model of Baja California Sur.

| | Prey/Predator | 1 | 2 | 3 | 4 | 5 | 6 | 7 | 8 | 9 | 10 | 11 | 12 | 13 | 14 | 15 | 16 |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 1 | Large sharks | 0.005 | | | | | | | | | | | | | | | |
| 2 | Small sharks | 0.021 | 0.006 | | 0.004 | | | | | | | | | | | | |
| 3 | Dolphins | 0.011 | 0.006 | | | | | | | | | | | | | | |
| 4 | Marlin | 0.005 | 0.002 | | 0.005 | | | | 0.001 | | | | | | | | |
| 5 | Yellowfin tuna | 0.001 | 0.02 | | 0.020 | 0.002 | 0.002 | | | 0.02 | | 0.002 | | | | | |
| 6 | Dorado | 0.026 | 0.012 | 0.008 | 0.093 | 0.021 | 0.017 | | | 0.041 | | | | | | | |
| 7 | Skipjack tuna | 0.074 | 0.075 | 0.001 | 0.225 | 0.037 | | 0.001 | 0.010 | 0.122 | | | | | | | |
| 8 | Sailfish | 0.002 | 0.002 | | 0.011 | | | | 0.005 | | | | | | | | |
| 9 | Other billfish | 0.011 | | | 0.007 | | | | | | | | | | | | |
| 10 | Small scombrids | 0.079 | 0.083 | | 0.097 | 0.104 | 0.126 | 0.214 | | 0.162 | | | | | | | |
| 11 | Misc. piscivores | 0.021 | 0.021 | 0.007 | 0.112 | 0.013 | 0.047 | | 0.241 | | | 0.014 | | | | | |
| 12 | Squids | 0.291 | 0.413 | 0.423 | 0.300 | 0.134 | 0.103 | 0.216 | 0.098 | 0.284 | 0.200 | 0.082 | 0.084 | | | | |
| 13 | Flying fish | 0.006 | 0.006 | 0.050 | 0.004 | 0.075 | 0.310 | 0.009 | | | | 0.068 | 0.010 | | | | |
| 14 | Small pelagic fish | 0.208 | 0.108 | 0.125 | 0.120 | 0.277 | 0.176 | 0.222 | 0.193 | 0.227 | 0.200 | 0.588 | 0.202 | | 0.032 | | |
| 15 | Mesopelagic fish | 0.132 | 0.088 | 0.383 | | 0.188 | 0.092 | 0.125 | 0.404 | | 0.100 | 0.096 | 0.142 | | | | |
| 16 | Zooplankton | 0.106 | 0.177 | 0.005 | | 0.149 | 0.126 | 0.214 | 0.048 | 0.162 | 0.500 | 0.150 | 0.562 | 1 | 0.806 | 1 | 0.2 |
| 17 | Phytoplankton | | | | | | | | | | | | | | 0.161 | | 0.8 |