THE VALUE OF INFORMATION FOR FISHERIES POLICY

by

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Abstract

For policy-makers and managers, knowing what information to collect is just as important as collecting information. I apply economics-based methods, including the value of information approach, to natural resource management in order to identify new optimal policies and priority areas for investment. Explicitly incorporating uncertainty is key to these methods, both in formally acknowledging alternative hypothesis and strategies, and for selecting policies that are most robust to uncertainty about natural and social systems.

Given their differences in objectives and current challenges, I develop and apply methods to both developing and developed marine fisheries. In Mexico, for example, I estimate that total fish catch over the last fifty years could be almost twice that reported in official data. This 'informal' catch reduces economic benefits from fisheries output, including informal processing and sales that add less value to production. Based on current monitoring investment and informal catch rates, I estimate that this represents an almost US$1 billion annual loss in foregone economic impacts, that could be partially gained by an annual investment of US$100 million to increase formalization of current catch.

The benefits of assessing information value are not limited to developing fisheries or "data-poor" contexts. Linking ecosystem models with economic data and frameworks, I estimate that the supporting service value of forage fishes as food for other fished species vastly outweighs their yearly landed value (in the Southern Baja California Peninsula, US$180 million compared to US$62 million). For the California Current, which includes Mexico, the US and Canada, I couple game-theoretic and ecosystem models and find that moving beyond single-species valuation

supports arguments for sustainable fishing of forage fishes, and creates incentives for cooperative fishing strategies across a range of climate scenarios.

Aside from developing new and broadly applicable methods and frameworks, the overarching finding of this work is that it is always beneficial to formally and openly acknowledge uncertainty and alternative management strategies in natural resource assessments. This allows us to provide robust advice to policy-makers given, and not stymied by, uncertainty.

Preface

All chapters aside from Chapters 1 and 6 were prepared as individual manuscripts. Chapters 2 and 3 were submitted to peer-reviewed journals, with Chapter 2 published and Chapter 3 under review. Chapter 4 has been published as a book chapter, and Chapter 5 is being prepared for submission to a peer-reviewed journal. I am the senior author on all chapters, and led the design and implementation of analyses.

A version of Chapter 2 has been published as: Cisneros-Montemayor, A.M., Cisneros-Mata, M.A., Harper, S. and Pauly, D. (2013). Extent and implications of IUU catch in Mexico's marine fisheries. Marine Policy 39: 283-288. I gathered most reference materials, wrote the estimation model and analyzed results, and wrote the core of the manuscript. M.A. Cisneros-Mata, S. Harper and D. Pauly provided key references and in-country information and commented on and improved the model and manuscript.

A version of Chapter 3 has been submitted to a peer-reviewed journal and accepted for review with co-authors C.J. Walters, W. Swartz and U.R. Sumaila. I gathered necessary data and developed the estimation model with the help of C.J. Walters. W. Swartz provided key data used for the analysis. I wrote the core of the manuscript, and U.R. Sumaila helped improve the initial versions. All co-authors provided comments for improvement of the final manuscript.

A version of Chapter 4 has been published as a book chapter (Spanish-language). Cisneros-Montemayor, A.M. (2013). Ecosystem models to explore the economic role of forage fishes. [Modelos ecológicos para analizar el papel económico de los peces de forrage]. In: Blanco, J.A. (Ed.) Application of ecosystem models to natural resource management. [Aplicaciones de modelos ecológicos a la gestión de recursos naturales] (pp. 67-76). Barcelona: OmniaScience.]. I gathered necessary data, developed the estimation model and wrote the manuscript. My thesis supervisor, U.R. Sumaila, and committee member, V. Christensen, provided comments for improvement.

A version of Chapter 5 is being prepared for submission with co-authors G. Ishimura, G. Munro and U.R. Sumaila. I led the method design and analysis of results, and wrote the core of the manuscript. G. Ishimura helped develop the core model and select results for analysis and G. Munro and U.R. Sumaila helped analyze results, and contributed to the framework for discussion. All co-authors provided comments for improvement of the manuscript.

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I could not have been successful, personally or professionally, without the unwavering love and support from my wonderful family, Jessica and our daughter, Eva, my parents, Miguel and Gabriela, my brother, Tomás, and the other 92 (actual approximation) Cisneros-Montemayor (and now, plus some Mackenzie-Feder!).

To my second family of friends, you are too many to count yet I thank you all individually. I won't try to name you, you know who you are: ¡Muchísimas gracias a todos y todas! and Thank you all so much!

I gratefully acknowledge funding and support from the Mexican Council of Science and Technology (CONACYT). Throughout the course of my studies, I had a chance to participate in various projects and research, and acknowledge funding from the Pew Charitable Trusts, the Fisheries Economics Research Unit, the *Sea Around Us* Project, Project Seahorse, the Department of Fisheries and Oceans, Oceans Initiative, Hokkaido University, the Gulf of California Marine Program, and the Global Environment Facility, among others.

Fp. 4:13.

A mi familia

1. Introduction

Any policy-maker is faced with limited resources to allocate to research. One of the overarching needs of managers is therefore to design strategies that recognize limitations in available information, and seek ways to acquire or avoid relying on this information, or gauge the robustness of management strategies to potential assumption errors (Walters 1986). In any of these cases, any honest manager must accept that uncertainty (whether structural, numerical, or from random error) is an unavoidable factor in any decision. Appropriately, there are many methods and applied cases that deal with various types of uncertainty, either by resolving it directly or by providing sensitivity analyses to a range of possible assumptions (Hilborn & Mangel 1997; Walters & Martell 2004). In all cases, the purpose of improvements to information and decision frameworks is to help design and implement strategies with the best chance of meeting societal objectives, however varied they may be. With this in mind, economics-based approaches have shown to be useful for contextualizing management issues, incorporating key variables, anticipating outcomes and conveying options to an array of stakeholders (Clark 2006; Munro 2009).

The value of information concept is a straightforward, yet logical and powerful framework to assess the economic implications of making a decision when there is a risk of error around our qualitative or quantitative assumptions (Walters 1986; Hilborn & Walters 1992). In this context, it provides optimal policies that are most robust to uncertainty, as well as estimates of the expected economic value of resolving matters of parameter or structural uncertainty in management models, and therefore a baseline for investment decisions (Brennan et al. 2007). In the context of fisheries, these approaches have been proposed to deal with matters of uncertainty around, for example, stock-recruitment dynamics (Walters 1986), the estimated abundance or spatial distribution of a stock of fish (Frederick & Peterman 1995; Mäntyniemi et al. 2009), and the optimal placement of marine protected areas (Costello et al. 2010). The computation of these values is straightforward, but the exercise of formally recognizing uncertainty in a management choice with different expected payoffs given competing assumptions is already important in a policy context.

While it may not always be possible to apply a formal value of information method to resource management issues, its qualitative implications nonetheless help to contextualize one of the most important issues for fisheries policy. In a realm of great uncertainties, managers must decide on how to allocate their own scarce resources for the maximum benefit of policy goals. Informed decisions thus rely on many resource management theories and methods developed to optimize performance metrics under given structures and policy goals. For example, analytical frameworks of optimal economic action in fisheries have been developed for use in multi-species settings, cooperative management frameworks, ecosystem-valuation, marine protected area management, fleet investment, bycatch reduction, high-seas management, recreational fisheries, effort allocation, fisher discount rates, fines for illegal fishing, and fishery contribution to national GDP, to name only a very few (e.g., Sumaila 1997a, 2002; Zeller et al. 2006; Sumaila et al. 2006; Munro et al. 2009; Teh et al. 2011). The VI framework does not in any way supplant these methods, but rather evaluates their results in the context of providing managers with a benchmark for investment decisions to resolve uncertainty by improving monitoring or enforcement.

Improved data availability offers a quite different set of opportunities and limitations depending on the initial state of available information, which is highly correlated with the level of management strength. On the one hand, improvements in data-limited situations can have very basic yet powerful effects on the status of the fishery and surrounding ecosystem, the two overarching policy goals for any management setting. In terms of catch, the most basic of fishery statistics, unreported catch (often widely referred to as illegal, unreported and unregulated; IUU) has been rightfully identified as a main concern for management, with subsequent attempts to estimate it even at the global scale (Sumaila et al. 2006; Zeller & Pauly 2007; Agnew et al. 2009; World Bank 2012). However, answering why a public-resource owner (a nation) would be preoccupied with unreported catch has received less explicit attention. In developing countries, this can mean addressing societal losses from overexploitation leading to trade sanctions and foregone potential production; lost tax revenues; decreased downstream economic impact when illegal catch is quickly exported without entering into national processing chains; and a distorted view of returns when fishing subsidies are seen as an investment for the nation. This assumes that, as representatives of society, managers are in fact attempting to maximize societal goals. Unfortunately, this may not be true in many cases, and particularly in developing countries (Robbins 2000; Kolstad & Soreide 2009). Although the issue is beyond the scope of this thesis, it is likely that an assessment of the value of potential actions could expose clear potential for societal economic gain, therefore questioning why these actions have not been taken. So, at the most basic level, value of information frameworks can provide assessments of potential action, and grounds for questioning inaction.

Data-rich fisheries generally (but certainly not always) tend to perform better than data-poor ones at meeting desired policy goals (Costello et al. 2012), but improvements to data quality and type, or changes in the scope of societal considerations can still have drastic changes on optimal management strategy. While there, of course, tends to be a direct relationship between available data and management strength, I argue that these are not necessarily inextricably linked. In any case, there certainly seems to be a trend towards even wider inclusion of data and economic values within advanced management frameworks, perhaps the best example being the ‘ecosystem approach to fisheries’. This approach is an increasingly integral part of marine resource management and plays a particularly strong role in fisheries policy, particularly in moving beyond single-species management. Although the more important part of this approach is the qualitative consideration of the whole ecosystem when designing policies, quantitative ecosystem-based approaches remain a debated subject, largely because of perceived operational challenges (Hilborn 2011). For example, the integration of ecosystem services within management discussions is usually welcome and increasingly incorporated into legal management frameworks (Greiber et al. 2009), but the explicit inclusion of quantitative ecosystem-based analyses into policy remains contentious (De Groot et al. 2002).

Whether accurately quantifiable or not, it is clear that the contribution of fish species or groups to ecosystem functions can be considered along with their direct human use value. While relatively advanced models are necessary to provide a framework for ecosystem service valuation, the focus of this work is less on the development of the model itself, and rather on the potential changes in optimal management strategy that could stem from the explicit and quantitative inclusion of wider ecosystem services in decision frameworks. As ecosystem models in general require much more data than a more traditional single- or multi-species assessment, it is crucial from a management setting to know if these wider values would significantly change optimal strategy, thereby justifying the additional cost of research into improved data and structural knowledge.

Stakeholder engagement, co-management and the recognition of management needs and limitations are crucial for successful resource management, and often influence outcomes more than the management schemes themselves (Gutiérrez et al. 2011). In the case of marine resource management, general patterns of ecological benefits and the conditions under which they appear have been identified for some time (Clark 1973; Munro 1979; Lubchenco et al. 2003; Mawdsley et al. 2009), but the social and political aspects of implementation are seriously underrepresented in scientific studies. This is somewhat surprising given the recognition that these points ultimately determine effective implementation and enforcement of policies, both in theory and in practice (Alder et al. 1994; Kaiser 2005).

Given that stakeholders and managers have an array of objectives in mind, and equally varying time-horizons for assessing performance (Carr 2000), it becomes necessary to implement frameworks that capture uncertainty in weighing the potential benefits of competing management strategies. Recognizing the value of this information in formal assessments can help fill this need and hopefully provide users with a clearer view with which to make informed decisions on resource use.

Following from the discussion above, this work will address three primary questions.

1) What are the economic implications of fisheries data limitations?
2) How do economics-based fisheries policy recommendations change as wider data become available?
3) Do increases in expected benefits following from improved knowledge outweigh the costs of improving knowledge?

These questions regarding the implications of improved data availability are addressed at two stages, one of improvement at a basic level (Chapters 2 and 3) and one of improvements to an already-advanced level (Chapters 4 and 5). The unifying threads throughout the chapters are thus the implications of knowledge gaps in management, the rationale for exploring the value of knowledge acquisition in order to make wise management investments, and potential changes to optimal policy as wider information becomes available.

Chapter 2 focuses on marine fisheries in Mexico as a case of poor data monitoring, with limited reliable information on basic indicators such as catch per fishery or other abundance indexes. Using available information, the magnitude of (legal or illegal) unreported catch is estimated. Chapter 3 first identifies the economic implications of incomplete catch data in terms of societal costs (through lost downstream economic impacts, taxation and harmful subsidies) and resource costs (through overexploitation). It then develops a value of information framework to assess the gains expected from improved fishery information, using the reported and estimated catch per fishery in the previous chapter as alternative hypotheses on catch. Chapter 4 turns to issues of information value in data-rich settings. In this chapter, the market values of single species are confronted with the supporting services and downstream economic impact provided to various predator fisheries. This includes a simple method for assessing the unit market value of species within an ecosystem that accounts for value other than direct human consumption. Chapter 5 goes further by applying the concept of ecosystem service value to a multi-national fishery setting. In this case, previous game-theoretic analyses have been proposed to analyze fishing cooperation on a transboundary stock under climate-change. This chapter confronts these prior (single-species) models with wider ecosystem values, testing if cooperative responses would change significantly as a result.

Throughout this work, the fundamental question addressed is: given the expected economic benefits from competing sets of management assumptions and frameworks, is it worth it to invest in finding out which set is correct?

2. Extent and implications of deficient data monitoring in marine fisheries: Mexico as a case study[1]

2.1 Background

The most important lesson learned after a century of modern fishing is that the world's ocean resources are not inexhaustible, as previously held both in popular and academic circles (Melville 1851; Huxley 1883). Since this opportune realization, the main endeavor of the fisheries science community has been to develop quantitative methods by which fish stocks can be monitored and assessed in order to gauge their status with respect to given management reference points (e.g., Baranov 1918; Beverton & Holt 1957; Hilborn & Walters 1992; Walters & Martell 2004). The most important component of these status indicators is a measure of the catch of a given stock, and it thus has received the most attention in terms of data gathering both at the local and global scale, with a global database of catches since 1950 maintained by the FAO (Garibaldi 2012). Though the potential and limitations of catch as a stand-alone indicator of fishery status has been extensively discussed (e.g., Branch et al. 2010; Carruthers et al. 2011; Kleisner et al. 2013), it is the foundation for nearly all other assessment methods, and the only information freely collected by fishing fleets. The current sub-optimal state of most marine fish stocks (FAO 2012) has prompted organizations at international, regional and national levels to confront fisheries issues with management decisions, with the reliability of catch statistics being of particular concern.

[1]The following is adapted from the paper published as: Cisneros-Montemayor et al. (2013). Extent and implications of IUU catch in Mexico's marine fisheries. Marine Policy 39: 283-288

Fisheries in Mexico, reflecting the overarching political system, have historically been characterized by constant shifts in objectives and management schemes (OECD 2006). They have thus evolved from an overlooked sector, to a primary source of food and job creation, to a casualty of neo-liberal reform and now to the object of an apparent tug-of-war between *laissez-faire* management on the one hand and ecological conservation priorities on the other (Espinoza-Tenorio et al. 2011b). The participation and influence of scientists, academics and conservation organizations in fisheries management has also evolved towards a more holistic understanding of the social, political and ecological context of Mexican fisheries, with an increase in training in and application of novel quantitative methods to assess national fisheries' status (Hernandez & Kempton 2003). Unfortunately, a lack of effective fisheries governance in general, and catch monitoring in particular, has resulted in highly uncertain fishery statistics, which often lack the quality to be informatively used within quantitative assessments that reflect reality.

Illegal, unreported and unregulated (IUU) fishing is a significant issue all over the world, and can seriously misrepresent fish production at any level (Agnew et al. 2009; World Bank 2012). In Mexico, a large, mostly *de facto* open-access fishing sector (>300,000 fishers), versatile boats and gear, an extensive coastline, corruption, and a limited capacity for monitoring and enforcement result in significant IUU catch (Rodríguez-Valencia & Cisneros-Mata 2006). Even in the case of legal fishers, official statistics rely on the compulsory but unenforced submission of catch logs by fishers or buyers to the local fisheries office. In both cases, there is no further validation of catch, and catch logs are often filled in on the spot (and often for a fee) by fishery officers based on the fishers' memory of past catch (Espinosa-Romero et al. 2012). A survey of Mexican fishery experts including scientists, officials, fishers and others, found that in some fisheries, "irregular" fishing (unreported and illegal) currently represents 40-60% of reported catch (Cisneros-Mata et al. 2012).

This estimate does not account for discards in shrimp trawls, which historically have had a 1:10 shrimp to bycatch ratio and are widely regarded as the single most important source of unreported bycatch (Vázquez et al. 2004).

In light of the apparent disconnect between the recognized importance of catch statistics for management and the state of data monitoring in Mexico, alternative methods must be used in order to provide better estimates. Catch reconstructions have been employed extensively to address this issue e.g., Zeller et al. (2007a, 2011), under the fundamental thesis that 'unknown catch' does not equal 'zero catch' (Pauly 1998a). Although this is a simple and logical observation, attaching numbers to qualitative knowledge is powerful in conveying the seriousness of the issue and the need for action; this is indeed the main objective of the present study. Following this principle, I provide the first comprehensive estimate of unreported fisheries catches in Mexico, from 1950 to 2010.

2.2 Methods

The philosophical core of the reconstruction method is that, when it is recognized that catch in official statistics is incomplete but the magnitude of missing catch unknown, a well-informed estimate should replace a zero value (Pauly 1998a). Information can come from a variety of sources, including peer-reviewed literature, grey literature, and expert knowledge, but every attempt is made to employ it in a conservative manner (Zeller & Pauly 2007). The main difference between the methods used in this study with respect to those used in the past is that the focus is on estimating catch series by particular species, rather than by a fishery sector. The estimation of Mexico's total marine fisheries catch was thus undertaken within a structured database as explained below. Specific estimation methods for each fishery are presented in Appendices A and B.

Statistics for marine fisheries catch by Mexico within its EEZ from 1950-2010 were extracted from the FAO database (http://www.fao.org/fishery/statistics/software/fishstat/en), where catch is specified by FAO area. Due to significant inconsistencies identified in data available directly from the national fisheries agency (see Discussion), these FAO catch series formed the basis for subsequent estimations.

Mexico's subset of the FAO database consisted of 192 individual catch series (96 each for the Pacific and Atlantic Oceans) of varying taxonomical precision, with catch reported by year from 1950-2010. A series of descriptive categories were assigned to each catch series, and to every reconstructed series, and included:

a. Taxon: scientific name for the group, as precise as possible;
b. Group: elasmobranchs (e.g., sharks, rays), large pelagic fish (e.g., tunas, jacks), small pelagic fish (e.g., anchovies, sardines), benthopelagic fish (e.g., snappers, triggerfish), benthic fish (e.g., flounders), cephalopods (e.g., octopus, squids), gastropods (e.g., abalone, snails), bivalves (e.g., clams, mussels), echinoderm (e.g., sea cucumbers, sea urchins), other (e.g., seaweeds);
c. Target: main target of fishery (e.g., the "tuna" or "shrimp" fisheries use specific gears but catch many species other than shrimps and tunas, both targeted and as bycatch);
d. Sector: artisanal (open deck, outboard or no engine), industrial (covered deck, inboard engine), recreational (food or sale are *not* the main motive for fishing), subsistence (catch kept for consumption in the household);
e. Type: reported (FAO statistics), unreported legal (non-quantified catch by fishers operating legally), unreported illegal (non-quantified catch by domestic fishers operating illegally in any way), unreported discard (non-quantified discarded catch);

f. Input: FAO, reconstructed;

g. Area: Pacific, Atlantic;

h. FAO Name: the name for the species or species group as it appears in the FAO data;

i. Individual Reference: a binary variable denoting whether specific information related to unreported catch was found for a given fishery;

j. Interpolated: a binary variable denoting whether a time series of catch was interpolated to fill data gaps.

Once the initial database was compiled as outlined above, the reconstruction was undertaken within its framework. For each catch series in the FAO data, the first step was to seek all available information related to the fishery, including gear types employed, observed bycatch (and discard) rates and species, and governance characteristics. Two initial sources of information were invaluable in this respect. The Mexican National Fisheries Charts (DOF 2004, 2010, 2012) are official documents that list all species recognized as fished, and include a brief summary on every major commercial fishery by area; the assessment and management "Red Book" (Instituto Nacional de la Pesca 2006) contains reports on all currently assessed species. If no information was found to justify clear gaps in a catch series, these were linearly interpolated. This included missing data in the first years of recorded catch. For example, if the first four years of a catch series were missing and the fifth was 500 tonnes, the first year was assigned half the value of the fifth (thus assuming the fishery had not grown from zero catch in 1950) and the other years linearly interpolated. Or, if catch records were missing from, say, 1960-1965, these were linearly interpolated from reported catch in 1959 and 1966. Interpolated catch was designated as unreported and used as the new baseline for subsequent estimations of unreported catch.

Whatever specific information was found for a given catch series was used to estimate the magnitude of unreported catch, expressed as a ratio relative to reported catch and then converted into (metric) tonnes (t) per year and entered as new catch series in the database (including the appropriate descriptors). This method is consistent with similar studies attempting to estimate such unreported catch for other regions of the world (Pitcher et al. 2002; Zeller & Pauly 2007). According to an extensive survey of fishery experts in Mexico, on average (over several fisheries) unreported ("irregular") fishing contributes a further 45% of catch (90% of which is illegal) relative to reported landings (Cisneros-Mata et al. 2012). Around half of illegal catch is subsequently bought by processors and reported with legal catches (second author's pers. obs.), so these would appear in FAO statistics. A conservative ratio (relative to reported catch) of 15% for unreported legal catch and 22% for unreported illegal catch were added to current reported catches when no other information was available for a specific fishery, or in the case of the broadly defined finfish (*escama*) fishery. According to fishers and buyers, legal unreported catches have decreased during the last decades due to improvements in monitoring, while unreported illegal catch has increased due to a growing number of fishers and the addition of fishery regulations. Therefore, the ratio of unreported legal and illegal catch from 1950-2010 were assumed to vary linearly, from 40-15% and from 10-22%, respectively. Due to a general lack of data, I was not able to apply sensitivity analyses directly; however, I calculated and report confidence intervals of +/- 15% applied to resulting aggregate catch estimates (based on variance of expert opinions reported in Cisneros-Mata et al. 2012).

A major component of unreported catches in Mexico is bycatch in the shrimp fishery, particularly by industrial bottom trawlers. The high economic value of shrimp results in discarding of bycatch species, which are high due to the tropical environment in which shrimp are caught, and the unselective gears that are used. Catches were first separated into artisanal and industrial sectors based on the historical number of vessels by sector (1970-2007 from CONAPESCA 2007, 2013, other years linearly extrapolated) and current catch ratio (DOF 2010). Shrimp catches (which are often reported in aggregate) were split into species based on available yearly catch ratios (DOF 2004, 2010) and the average ratio when data were unavailable. Shrimp to bycatch ratios for industrial fisheries were 1:10 and 1:3 for the Pacific and Atlantic Ocean, respectively (Bojórquez et al. 1999) and, for artisanal fisheries, 1:3 for legal gears and 1:10.5 for illegal gear in both oceans (Amezcua et al. 2006). Bycatch composition and discard rates were variable, with the discard rate reported as being higher in the Pacific and in the industrial fishery.

Specific estimation procedures for each fishery are included in Appendix A (Pacific Ocean) and B (Atlantic Ocean).

Published references regarding unreported catch in Mexican fisheries are scarce, so assumptions on their magnitude were necessary in several cases and are acknowledged as such. This study is intended to be the first iteration in an ongoing effort to improve Mexican fisheries catch statistics, and the resulting catch database is freely available upon request. Proposed revisions to one or several catch series by other researchers can then be discussed and the database (and documentation) updated.

2.3 Results

From 1950-2010, total unreported catch was estimated at over 44 million t, equal to 91% of official landings as reported to the FAO (48.4 million t). Even with our conservative estimation methods and allowing for potential error in the ratios applied, total reconstructed catch was and remains almost two times higher than official catch as reported to the FAO (Fig. 2.1). On average during the past 61 years, total reconstructed catch (reported + unreported) was over 1.5 million t/year, compared to 796,000 t/year in the official statistics (Table 2.1).

Table 2.1. Summary of fishery catch statistics by type in Mexico, 1950-2010.

	Catch by period (t x 10^3)			
Type	**1950**	**2010**	**Total (1950-2010)**	**Average/Year (1950-2010)**
Reported	97	1,504	48,556	796
Total Unreported	416	683	44,308	727
Unreported Legal	322	255	14,480	233
Unreported Illegal	76	170	5,278	86
Unreported Discards	17	258	24,648	404
Total	513	2,188	92,864	1,522

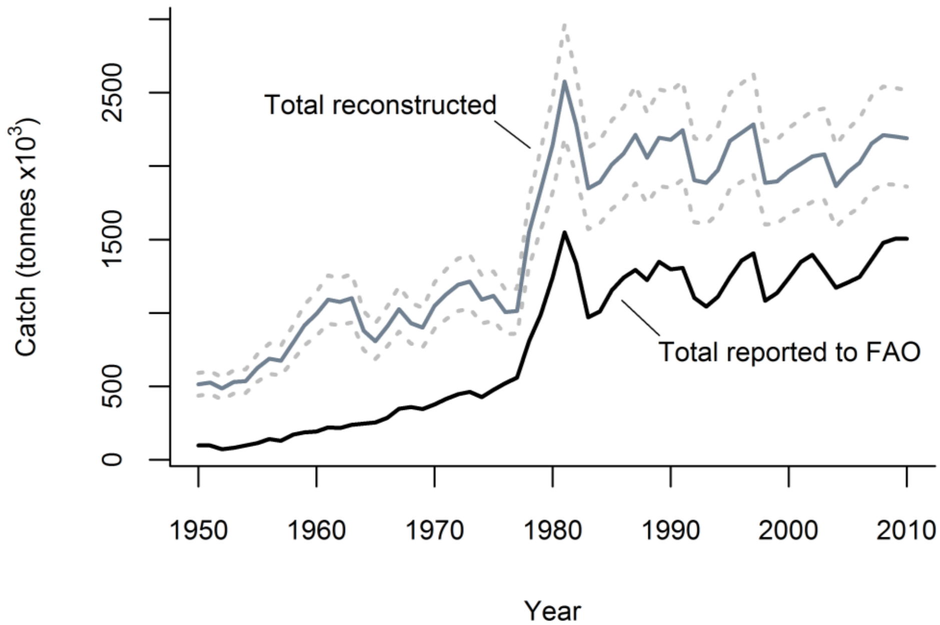

Figure 2.1. Total Mexican fishery catch reported to FAO compared to reconstructed catches estimated in this study. Confidence intervals (dashed lines) around estimate represent +/- 15% error.

A total of 192 entries, 96 per ocean, are reported in FAO catch statistics, corresponding to 148 taxa, though 5 corresponded to marine mammals and reptiles, not considered in this study. The resulting database of reconstructed catch includes 758 entries including reported and unreported legal, illegal and discarded catch by taxon, and a total of 243 taxa. Specific information regarding unreported catch was available for almost 40% of resulting time series, and 73 time series were interpolated to estimate obvious gaps in the time series, most in early years (see Appendix A). Applying both stationary and varying estimation ratios (e.g., unreported catch, bycatch), to reported catch by species resulted in fluctuating ratios of catch by type, but with an overall decreasing trend in the rate of unreported legal catch and an increasing trend in unreported illegal catch.

In aggregate, bottom trawls targeting shrimp have historically accounted for the highest total estimated catch (reported, unreported, illegal and discarded), with over 37 million t (54% of which was discarded) from 1950-2010, followed by finfish gillnets (*escama*; 24 million t), small pelagic seiners (19 million t) and large pelagic seiners and longlines (3.7 million t) (Fig. 2.2). Over the same time period, all other fishing gears caught almost 11 million t.

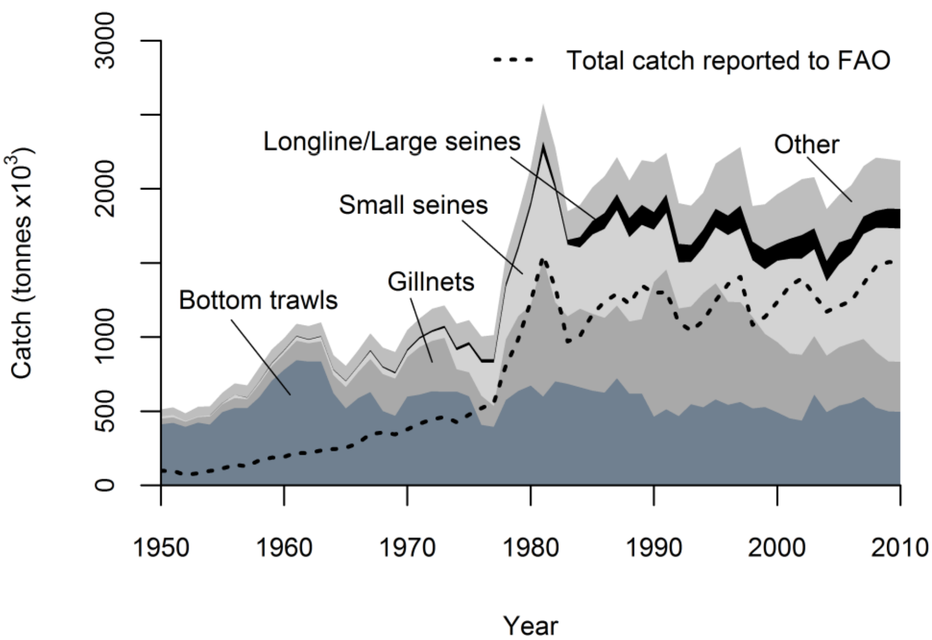

Figure 2.2. Total catch (reported + IUU) for gears with highest catch in Mexico (both oceans), 1950-2010. "Other" category includes hand-lines, hand-collection, traps and others.

In terms of catch by species group (Fig. 2.3), the highest total catch over the study period corresponded to benthopelagic fish (42.3 million t; all catches in metric tonnes), followed by small pelagic fish (19.6 million t), crustaceans (including crabs, lobsters and shrimps; 12.6 million t), large pelagic fish (6.4 million t), bivalves (3.1 million t), cephalopods (1.9 million t), elasmobranchs (1.8 million t), benthic fish (1.8 million t), seaweeds (1.7 million t), gastropods (1 million t), echinoderms (127 thousand t) and unidentified invertebrates (83 thousand t).

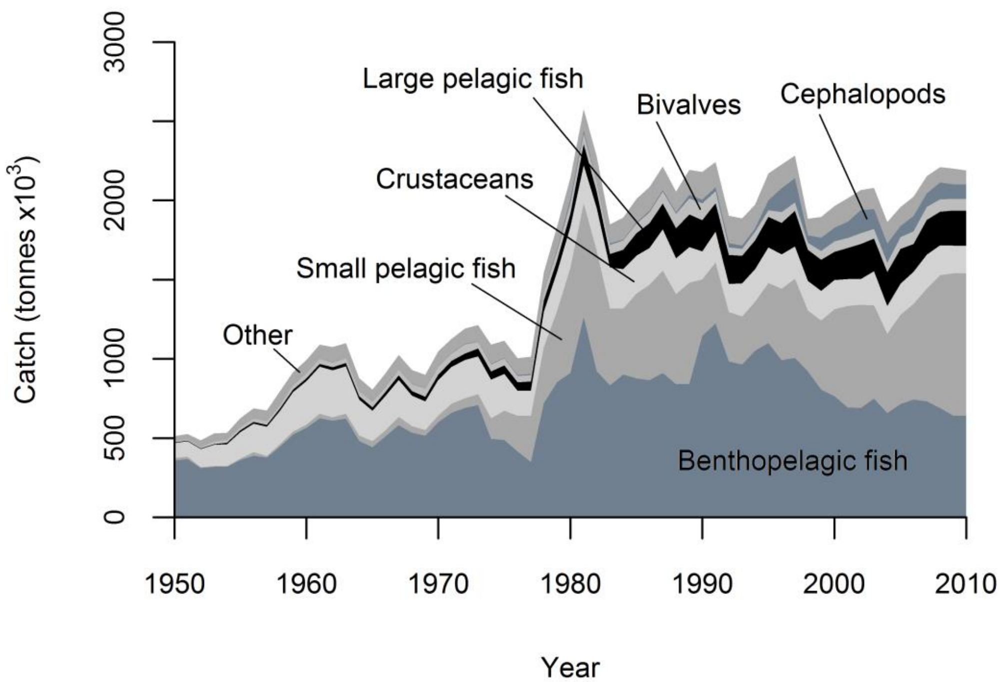

Figure 2.3. Total catch (reported + IUU) for taxa groups with highest catch in Mexico (both oceans), 1950-2010. “Other” category includes (in order of total catch) elasmobranchs, benthic fish, seaweeds, gastropods, echinoderms and unidentified invertebrates.

2.4 Discussion

Results show that from 1950-2010, total fisheries catch was almost twice as high as the official statistics as reported to the FAO. As expected from qualitative observation, unreported catch compared to reported catch was higher at the beginning of the study period (4.6:1 from 1950-1960). During this time, fishing cooperatives were granted exclusive fishing access, but there was little government interest or oversight of the sector until the creation of the National Fisheries Institute in 1962 (OECD 2006). Lack of regulation combined with the introduction of nylon netting and bottom trawl gear since the 1950s led to high unrecorded catch and discards (Figs. 2.1, 2.2), particularly in the Gulf of California shrimp and totoaba fisheries (Bahre et al. 2000). Management was strengthened by the onset of fisheries promotion programs in the 1970s (Fig. 2.2), which were highly successful in increasing fish catches, but did so largely through extensive government subsidies to the fisheries sector, mainly for technology, infrastructure and fuel (Espinoza-Tenorio et al. 2011b).

Four decades after the push for industrialized fisheries in Mexico, two main issues have arisen. First, as national fishing fleets are now large and relatively well-equipped, the subsidies that built-up these fleets now only serve to finance overfishing (around US$200 million in capacity-enhancing subsidies are currently conferred; Cisneros-Montemayor 2013a), undermining the resource base and jeopardizing future ecological function and economic benefits (Munro & Sumaila 2002). Second, accustomed to ongoing economic incentives, the fishing industry's attitude and strategy follow the expectation of government support without accountability, which results in limited private innovation and investment in efficiency, not to mention a lack of effective management control (OECD 2006; Espinoza-Tenorio et al. 2011a). Thus, the addition of potentially helpful policies intended to limit catch instead results in more unreported catch, now 'illegal' (Table 2.1).

Nevertheless, the overall ratio of unreported to reported catch has decreased over time, from over 4:1 in 1950 to 0.45:1 in 2010 (Table 2.1). This partly follows from declines in overall catches, lower discarding ratios as more species are retained and landed, and the explosive growth of fisheries for small pelagics, where almost all catch is reported. But, this also reflects improvements in monitoring capacity and disposition on the part of government agencies, and the work of research centers and non-government agencies within fishing communities to encourage documentation of landings and other pertinent information (Hernandez & Kempton 2003; Sáenz–Arroyo et al. 2005). Total catch has remained relatively stable for the last three decades, though catches have diversified over time, with 40% of taxa present in catches in 1950 compared to 2010. The addition of these new fisheries (notably for jellyfish, squid and swimming crabs), along with recent increases in the abundance of small pelagic fish, have masked declines in catch of benthopelagic fishes and other groups for the last two decades (Fig. 2.3).

The simplest conclusion of our results is that Mexican fisheries catches are currently not fully captured within government statistics that are subsequently provided to, but differ from, the FAO data (Fig. 2.1; Table 2.1). The decision to use FAO data as a baseline for estimations followed from a thorough analysis of national data freely accessible from CONAPESCA (the national governing body for fisheries and aquaculture) in its statistical yearbooks, which revealed clear errors (e.g., identical reported catches for different groups, or abrupt and drastic spikes in catch series). As these discrepancies are largely absent from the FAO data for Mexico, the reporting process from dockside to national to FAO statistics is unclear. However, the fact that statistics are collected at a national level, compiled in a comprehensive manner (errors notwithstanding), and furthermore made freely available over the internet, is an important development in the management of national fisheries and allowed for a study of this scope to take place at all. In many cases, this included the ability to allocate catches by taxa of varying precision, which is invaluable for the application of informative stock assessments.

Quantitative fisheries analysis in Mexico has made significant advances over the last decades as better training and technology are more readily available. Indeed, all but two of the 17 marine fisheries in the official assessment and management reference book (Instituto Nacional de la Pesca 2006) incorporate stock assessment methods including age-structured surplus-production models, virtual population analyses, and bioeconomic models. Together with a wider inclusion of stakeholders into the management process (Hernandez & Kempton 2003), moving towards a quantitative understanding of the dynamics of fish stocks certainly aids monitoring of stocks and ecosystem status. However, the current deficiencies in recorded catch statistics as highlighted in this study raise questions about the results of confronting structured statistical models with highly uncertain data.

Some metric of fisheries catch is the most important component of any stock assessment relying on fishery data (Hilborn & Walters 1992), so large discrepancies in recorded and true catch can result in erroneous estimates of the parameters and reference points that help inform management action. Furthermore, high uncertainty in parameter estimations following from errors can overwhelm inter- and intra-species interactions, negating the validity of the model itself (Walters & Ludwig 1981). Most of the species that are currently assessed do have relatively better catch monitoring in place, but an investment in recording full and accurate catch statistics (not to mention an updated estimate of nominal artisanal fishing effort, reported as static for the last 15 years) is *sine qua non* for the future expansion of stock assessment efforts. In the meantime, it would be highly advisable for any quantitative assessment to consider and present results for a wide range of potential parameter assumptions (Schnute & Hilborn 1993), even those as basic as the actual catch taken by a fishery.

Though discrepancies in reported and real catch have many implications for fisheries status assessments and management strategy, it is perhaps most troubling that in a country where 20 million people are undernourished (95% children; Olaiz et al. 2006), over 25% of fisheries catch over the last 60 years (currently 400,000 t/year) has been subsequently thrown overboard (Table 2.1). This highlights a pressing need for economic incentives that re-align these fishing strategies;

it is here that subsidies could indeed play a role through development of novel processing methods (Allsopp 1980), or in helping avoid bycatch, or enforce its retention while boosting prices for "trash" fish that could then be transported and sold at a discount in key regions of the country.

Implementation of turtle and fish exclusion devices on trawl gear, which had the highest catch of any gear type (37 million t; Fig. 3), can significantly reduce catch of large fish and turtles (Rodríguez-Valencia & Cisneros-Mata 2006), but reported bycatch ratios have nonetheless remained high during the entire study period (Bojórquez et al. 1999; Vázquez et al. 2004; Meltzer et al. 2012) and exclusion devices are often de-activated at sea by fishing crews (Cox et al. 2007). Bottom trawling is by no means the only gear type in Mexico with discards (e.g., Amezcua et al. 2006; Rodríguez-Valencia & Cisneros-Mata 2006; Ramírez-López 2009; Santana-Hernández et al. 2009; Shester & Micheli 2011), but it is likely where the first efforts to combat this wasteful practice, both through avoidance and retention of bycatch, would be most fruitful (FAO & International Development Research Centre 1982). Current Mexican law prescribes that strict bycatch limits must be set for all fisheries, yet thus far this has only been applied to billfish in commercial shark longliners (DOF 2007; Cisneros-Montemayor et al. 2012). As more fish stocks become fully or over-exploited, Mexico's fisheries will likely move toward a more efficient use of technology and enforcement to eliminate and/or efficiently use bycatch and discards. Our results provide a first estimate of the magnitude of these currently wasted resources.

This study provides the first estimate of total catch extracted by Mexican fisheries since the middle of the last century. Clearly, many assumptions are required for this type of undertaking (Pauly 1998a), though every attempt was made to provide estimates that were both substantiated by available information and erred on the conservative side. The main foreseeable obstacle was a shortage of first-hand information about particular species or fisheries, but in the end, 40% out of a total of 243 taxa were supported by specific information, and sources for aggregate groups (e.g., finfish) most likely adequately represent many others (DOF 2004, 2010, 2012). The uncertainty associated with estimations given limited information requires that methods be clearly stated and every assumption made clear, hence the inclusion of methods and sources for each fishery (Appendices A and B). Others are encouraged to question the methods used for a given fishery, analyze the raw results, and propose revisions to estimations if better information is available. Ideally, such revisions would update the current database and be included in a living document to that end. For this study, available information precluded the use of more detailed sensitivity analyses, but overall results are presented with confidence intervals derived directly from expert opinions (Fig.). Even in the case of the lower-bound, most conservative estimate, total unreported catch would represent over 500 thousand tonnes per year (Table 1).

From 1950-2010, total fisheries catch in Mexico, including both unreported legal and illegal catch and discarded bycatch, was almost twice as high as official statistics. This reflects a lack of clear policy to discourage such ill practices, as well as deficiencies in the reporting, monitoring and recording process which cannot be attributed to a single responsible party. Nevertheless, the fact that such a study was possible owes to advances in participation and interest in the sustainable use of the marine ecosystem, which I hope will continue and strengthen in the future, helping attain potential societal benefits. For this to become a reality, a change in culture must ensue including fishers, fishing leaders, field and administrative officials, technicians, researchers and all those involved in generating, collecting, processing, storing and publishing data and information. This study highlights an urgent need for reform in the fisheries sector; the question now is, where do we begin?

3. Economic implications of informal fisheries catch: a value of information approach[2]

3.1 Background

Informal fish catch, which includes unreported legal, illegal and discarded catch, has an array of economic consequences that are particularly prevalent in developing countries. First are the straightforward implications of unmonitored production for taxation and other rent-capturing mechanisms. There are always disincentives to paying taxes or otherwise contributing to management (Srinivasan 1973), yet the public ownership of fishery resources makes evasion more egregious, particularly if the resource base is concomitantly eroded given poor regulatory capacity (Baksi & Bose 2010). Similarly, market-based management schemes including maximizing economic yield, conferring subsidies or setting transferable quotas, are ineffectual without accurate data on the value of production (Munro & Sumaila 2002; Munro et al. 2009).

Perhaps more importantly, I argue that informal fisheries catch, if difficult to "launder" into formal production chains, is much more likely to be sold quickly and with reduced value-added activities such as processing and marketing that increase retail sales and trade (van der Meer 2012). Thus, downstream economic impacts and employment from landings, that easily outweigh landed value (Dyck & Sumaila 2010; Christensen et al. 2014), can be significantly and negatively impacted. Similarly, informal processing centers would be expected to generate a lower-quality (and value)

[2] The following is adapted from the manuscript submitted as: Cisneros-Montemayor et al. (*in review*). Economic implications and optimal monitoring investment given informal fisheries catch. Fish and Fisheries.

product, while providing less stable employment wages and benefits to their workers (Kucera & Roncolato 2008). Note that the informal fisheries sector already provides significant current benefits; informal economies in general represent around 40% of GDP in developing countries (Schneider 2002). I nevertheless argue that there is a loss of potential economic benefits from informal fisheries, and that economic growth in the sector could therefore be promoted by investing in formalizing current informal fishing (de Soto 2001).

Illegal, unregulated and unreported (IUU) fish catch has been recognized as a key concern in global fisheries (Pitcher et al. 2002; Agnew et al. 2009), and includes issues such as illegal fishing (Sumaila et al. 2006), vessel flags of convenience (Miller & Sumaila 2014), and foreign fleets operating in nation's EEZs (Pauly et al. 2002). I focus on 'informal catch' in the context of informal economies that contribute to socioeconomic benefits in a nation (or region) but are not reported in official statistics (Portes 1983). The FAO has collected and maintained global fisheries landings records since 1950 (Garibaldi 2012), yet such unreported catch has been roughly estimated at 18 million tonnes/year (Agnew et al. 2009). Ongoing formal efforts to re-estimate fisheries catch around the world have found (often significant) unreported catch in all of the territories analyzed to date, sometimes doubling the official regional catch data (Zeller & Pauly 2007).

Here, I use a value of information method (Walters 1986; Hilborn & Walters 1992) to evaluate the outcomes of alternative monitoring investment policies when there is uncertainty regarding the initial size of the informal sector, its associated economic losses, and the effectiveness of management. The expected economic value of resolving parameter or structural uncertainties thus becomes a benchmark for investment decisions (Brennan et al. 2007). Formal value of information methods have not been applied extensively in fisheries management (Hansen & Jones 2008), but have nonetheless been useful in resolving issues in stock-recruitment dynamics, abundance and spatial fish stock data, and the optimal placement of marine protected areas (Walters 1986; Frederick & Peterman 1995; Mäntyniemi et al. 2009; Costello et al. 2010).

The analytical framework developed here is straightforward, but a formal recognition of uncertainty in policy choices given competing assumptions and expected payoffs is quite significant, particularly when management budgets are limited (Walters 1986; Hansen & Jones 2008). I use a case-study database of official and estimated total landings and landed value over time for a developing country (Mexico; Chapter 2, Swartz et al. 2013). Aside from case-study results, the discussion focuses on the value of applying value of information methods themselves to confront basic fisheries governance issues, particularly in developing countries.

3.2 Methods

This study develops a method for estimating the economic value of formalizing fishery catches, i.e. it does not address potentially beneficial changes in the amount of catch. A key assumption of this study is that a formal, legal, and regulated catch and processing sector for fisheries outputs generates the most value and subsequent economic impacts from landings (Christensen 2010; Dyck & Sumaila 2010). Therefore, informally-sold fish catch that does not enter first-best processing and marketing results in lost potential economic impacts. I use available data whenever possible, and apply sensitivity analyses to parameter assumptions. Future research should focus on providing more empirical parameters, though sensitivity analyses should nonetheless be applied.

The value of information framework in this study uses three main inputs that can be either assumed or derived based on available information. These are: *i*) the current amount of informal catch; *ii*) the lost potential economic benefits associated with informal catch; and *iii*) the cost-effectiveness of investments in catch monitoring.

Amount of informal fisheries catch

The informal fishing sector, as defined here, comprises all fisheries catch that is not reported in official statistics. This includes unreported legal (unrecorded but adhering to existing fisheries regulations), illegal (breaking existing fisheries regulations, including fishing without a permit and/or during seasonal/spatial closures), and discarded (caught but discarded at sea) catch. This definition draws from previous work on unreported fisheries catch (Chapter 2; Zeller & Pauly 2007), though I move beyond the implications of unreported catch for fisheries management (Pauly 1998a) and into the broader context of informal economies (Portes 1983).

So, in any given country or region there will be formal fisheries catch that is recorded and reported by the relevant administrative body, and often will appear in FAO statistics, and an unknown but estimable amount of catch that is not officially recorded, yet contributes to informal markets at various scales. For subsequent calculations, key inputs are thus the formal (i.e., reported) and estimated total catch for the unit of analysis.

Economic losses from informal fisheries catch

Resource management policy analyses should follow specific sets of goals set by society; in this application, I assume that the goal of management is to maximize economic impacts from current landings, accounting for the costs of monitoring itself. Current economic impact (EI) from total national fisheries catch given the sector (*s*= formal, and informal legal, illegal and discarded catch) is estimated as:

$$EI = \left[\sum_{S,T} \left(p_{\tau} \cdot C_{\tau,S}\right) \cdot m_S^* \right] - c \mid u \qquad [1]$$

where,

$$m^* = \begin{cases} m \text{ if } s = \text{formal} \\ \alpha \cdot m \text{ if } s = \text{informal legal} \\ \beta \cdot m \text{ if } s = \text{informal illegal} \\ 0 \text{ if } s = \text{informal discarded} \end{cases} \quad [2]$$

and c is the management cost of formalizing fisheries production given a ratio u of informal (unreported) catch. The first term in equation 1 is the landed value of catch, with the ex-vessel unit price (p) for each taxonomic group (τ) applied to the total catch (C) per taxonomic group and sector. The economic impact multiplier (m^*) of landed value is also determined by the catch sector, where adjustment factors (α, β) for each type of informal catch are entered as a proportion of the economic multiplier (m) when catch is formal (Eq. 2).

I assume that EI^{max} is achieved when all catch is formally landed, recorded and processed (i.e. all s= formal and m^* is the maximum; Eq. 2), so that it can freely enter processing chains and downstream value is maximized. Conversely, EI^{min} would occur if all catch is informal, resulting in the lowest possible economic impacts. In reality, EI from catch in a given country or region would occur between these two extremes, depending on the proportion of catch for each sector and assuming that their multipliers are lower than formal catch (i.e., α and β are >0 and <1; Eq. 2).

I assume that unreported catch, while legally able to, is unlikely to fully enter formal production processes, as it would have been recorded and reported there (for example, by formal wholesalers and exporters). Nevertheless, this catch can be sold openly and with some post-catch value at informal fish markets or restaurants. Illegal catch is unlikely to be processed formally as it must usually be sold quickly to avoid detection. To illustrate the extreme case, a vessel fishing illegally in another nation's waters would take that fish and process it elsewhere, thereby "exporting" all potential value (Sumaila et al. 2006). 'Laundering' of illegal catch into formal processing and marketing chains does occur, but upon doing so it would cease to be informal catch and thus appear in official statistics.

Discards obviously have no landed value, yet I calculate their potential landed value and economic impacts under the assumption that the fishes normally thrown back due to high-grading could potentially have grown and/or been caught by other fishers who could then sell them at their market price. Some prices for discarded fish can nonetheless be low, which can be expected to reduce the economic value of discards compared to their volume relative to reported catch.

Cost-effectiveness of alternative management strategies

After estimating lost potential economic impacts, I evaluate the outcomes of competing monitoring investment policies, where monitoring is defined as enforcing (rather than merely measuring) policies that formalize catch. The effectiveness (Φ) of each monitoring investment policy (ι) in terms of the resulting catch reporting rate is assumed to follow an asymptotic pattern, and is calculated as:

$$\Phi_i = (1 - e^{-k \cdot \iota}) \cdot (\Phi_{max} - \Phi_{min}) + \Phi_{min} \qquad [3]$$

where ι represents the monitoring investment policy (the amount of money spent on formalizing catch, e.g., US$ millions) and *k* is the efficiency of monitoring investment (i.e., by how much is monitoring improved for each dollar invested). Effectiveness is bounded by minimum and maximum (Φ*min*, *max*) reporting rates at any level of investment; no matter how small an investment there will always be some catch that is reported (perhaps the fisher must comply with external regulations such as certification requirements), and no matter how large an investment there will always be fishers who circumvent monitoring (Sutinen et al. 1990).

Using the cost and expected effectiveness of each investment policy, I estimate the economic impacts of each policy (ι) given alternative real-world rates of unreported (informal) catch (u= [0,1]), and the expected value (EV) of each policy over a range of probabilities (ρ) that each u is true as:

$$EI_{\iota,u} = [(1 - u + u * \Phi_\iota) \cdot EI^{max}] + \left[u \cdot (1 - \Phi_\iota) \cdot EI^{min}\right] - c_\iota \qquad [4]$$

$$EV_\iota = \textstyle\sum_u EI_{\iota,u} \cdot \rho_u \qquad [5]$$

In Eq. 4, the first and second terms are, respectively, the economic impacts of formal and informal catch given the effectiveness of the monitoring investment policy (i.e. the proportion of catch that is reported), and c is the cost of each policy. In Eq. 5, EV is the expected economic impact of making a given management choice (in this case monitoring investment) over the range of possible assumptions on the real rate of informal (unreported) catch (u) (Walters 1986). Thus, there is an optimal policy EV_ι^* given each assumption of uncertainty (in this case, the proportion of informal catch) being true; EV^* would then be the overall optimal policy if we do not expect to resolve this uncertainty (Walters 1986; Costello et al. 2010).

Finally, the expected value of perfect information (EVPI; (Walters 1986)), which represents the marginal value of resolving uncertainty and subsequently choosing the optimal policy, rather than EV^*, is:

$$EVPI = \textstyle\sum (EV_\iota^* | \rho_u) \cdot \rho_u - EV^* \qquad [6]$$

Results are tested over a range of parameter assumptions, including monitoring efficiency, uncertainty around real unreporting rate, and the assumed loss in potential economic impacts associated with informal catch. In particular, the focus is on identifying parameters that change the optimal management policy itself (i.e., how much should be invested), as opposed to changing the magnitude of estimated economic impacts following a given policy (i.e., how large the gains will be).

Case study: Mexico

As a case study, I use recent (2010) data available for Mexico, where official catch data were updated using available information to estimate total informal catch from 1950-2010 (Chapter 2). Catch was reported by taxonomic group (τ in Eq. 1) and for distinct sectors (s in Eq. 1): formal (reported), and informal (unreported legal, illegal and discarded catch). All catches were matched to taxon and year-specific ex-vessel price data for Mexico, extracted from a global price database (Swartz et al. 2013). Using available official Mexican data (CONAPESCA 2013), taxon prices were adjusted to account for differences in the price estimation model compared to on-site reports. Overall, estimated prices were (median) 35% higher (mean= 17%) than Mexican government reports, and this adjustment factor was used when official taxon-specific prices were not available. Taxa that appeared only in discards (<30% of all taxa) were checked individually and assigned a price of zero if there is no current market for them (almost all were small benthic crustaceans).

All landings, regardless of sector, were assumed to receive the same ex-vessel price. Restricting landings (supply) could potentially result in a higher unit price on the informal market, but the inelastic price of fisheries products (Swartz et al. 2013) and an unwillingness of formal (regulated) processors to deal with illegal products likely keep prices relatively stable at the regional level. This is, however, a potential issue to address when working at global scales or with species that cannot be easily substituted; differential prices per sector can be easily integrated into the estimation method.

On average from 2005-2010, 48% of total catch – without including current large industrial catch of small pelagic fishes that masks overall Mexican fisheries catch trends – did not enter formal records (Chapter 2). In the same time period, average investment in enforcement and monitoring in Mexico was US$34 million (Lara & Guevara-Sanginés 2012). Therefore, I assume a baseline rate of informal catch u= 0.48, given a current cost of monitoring c= US$34 million (Eq. 1).

There is understandably little information on the economic impacts of informal fisheries in Mexico, though this was available for formal fisheries catch (m= 1.72; (Dyck & Sumaila 2010). I make conservative assumptions on the proportion of formal economic impact multipliers achieved by unreported legal and illegal catch, and varied these parameters widely during sensitivity analyses to test our results. As unreported legal catch can be sold openly in Mexico, but did not enter formal (first-best) markets where it would have entered official records, its multiplier is therefore assumed to be 50% of maximum (α= 0.5; Eq. 2). Illegal fishing by foreign fleets is common throughout the world, and often involves at-sea transfer and processing of catch (Kaczynski & Fluharty 2002; Pauly et al. 2013). In Mexico, I assume that this is not a significant issue and that illegal catch is landed in the country mostly by Mexican fishers, though does not enter formal processing and has an impact that is 25% of maximum (β= 0.25; Eq. 2). If catch were fished by foreign vessels and not landed in the country, β would equal zero. These assumptions can be validated or revised based on future empirical work.

Monitoring efficiency was derived empirically using the current monitoring investment (c) and reporting rate (u), $\Phi_{min,max}$, and solving for k in Eq. 3. Conservative bounds of Φ_{min}= 0.1 and Φ_{max}= 0.9 were used (i.e. at zero investment in monitoring, 10% of catch is reported, and at infinity investment, 90% is reported) based on empirical data reported for US fisheries (Sutinen & Kuperan

1999). Probabilities (ρ) were initially assumed to be normally distributed around the baseline u (0.48, σ= 0.07; scaled confidence bound reported in Chapter 2).

This study aims to illustrate the main arguments, testing results over a broad range of assumptions. Applications should use empirical data as much as possible, yet always provide sensitivity analyses due to the high uncertainty associated with these types of data. If available and required, the framework also allows for use of economic impact multipliers and prices specific to fishery types, such as industrial, artisanal or recreational.

3.3 Results

Using available information and the methods described above, the informal fisheries sector in Mexico is currently estimated to generate over US$900 million per year in economic impact (Table 3.1). Nevertheless, there is a current estimated yearly 20% (US$961 million) loss in total potential economic impacts (including both formal and informal catch) due to informal fisheries catch (Table 3.1). The highest loss in potential value is from discards, followed by illegal, and legal unreported catch (Table 3.1). Around 70% of catch in discards corresponded to taxa that also appear in formal catch, and have a current market price.

Table 3.1. Current and potential estimated economic impact of Mexican fisheries, by type, based on available data (CONAPESCA 2013; Chapter 2; Swartz et al. 2013), and methods above. All values are in 2014 USD millions.

	Economic impact (USD millions)	
Sector	**Current**	**Potential**
Formal	2,497	2,497
Informal subtotal	923	1,884
Unreported	573	725
Illegal	350	511
Discard	0	648
Total	3,421	4,382
Total – Monitoring costs	3,387	4,348

The expected effectiveness of a monitoring policy is key to determining how much to spend on formalizing catch. Under baseline parameters, the optimal monitoring investment policy (i.e., enforcing formalization of catch) when we do not expect to resolve uncertainty (i.e., EV^*) is US$130 million, with 85% of catch entering the formal sector and resulting in an economic impact of US$4.1 billion. This represents an increase of US$720 million over current economic impacts (Table 3.1). The optimal monitoring policy increases with assumed current proportion of informal catch (u), though these policies are highly sensitive to the efficiency of monitoring investments (k in Eq. 3; Fig. 3.1). Under the current rate of informal catch and monitoring investment (US$34 million) in Mexico, doubling the current monitoring efficiency decreases the spending required for the optimal monitoring policy to US$80 million (Fig. 3.1).

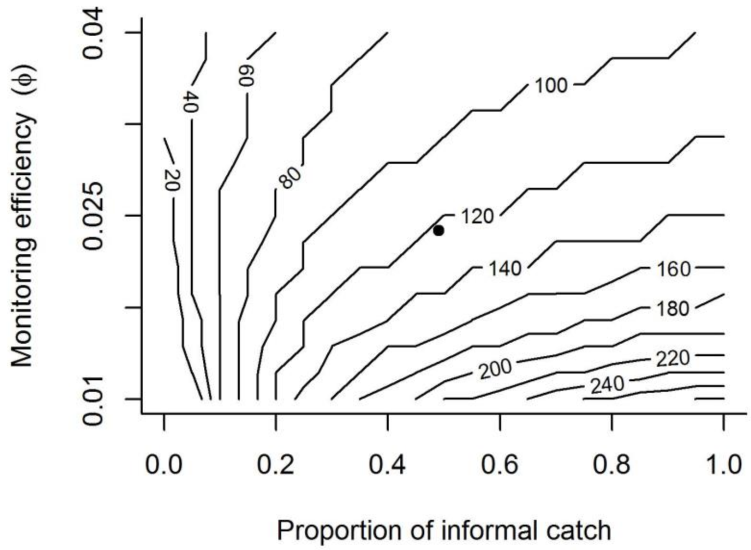

Figure 3.1. Optimal monitoring investment (EV*; US$ millions) given alternative assumptions on monitoring efficiency (Φ) and real proportion of informal catch (u). Solid point marks EV* at baseline parameters.

Uncertainty around the real proportion of informal catch (standard deviation of ρ in Eq. 5, 6; Fig. 3.2A) is inversely related to the EVPI, as higher uncertainty means that more could be gained by investing in resolving it. In the baseline case (σ=0.07), the EVPI was US$525 thousand, meaning that choosing the policy that maximizes expected economic impacts under current uncertainty would result in significant gains. If we were less confident about the real proportion of informal catch, say at σ=1, EVPI would equal US$18 million. A key result, however, is that the actual optimal monitoring policy (EV^*) is insensitive to uncertainty (σ) around assumptions on real unreporting rate or its economic impacts (i.e., α, β); though the expected overall value changes, the optimal investment policy does not. As noted above, the results are thus much more sensitive to the efficiency of investment in monitoring (Fig. 3.2B), as well as the initial guess at the mean rate of informal catch (*u*).

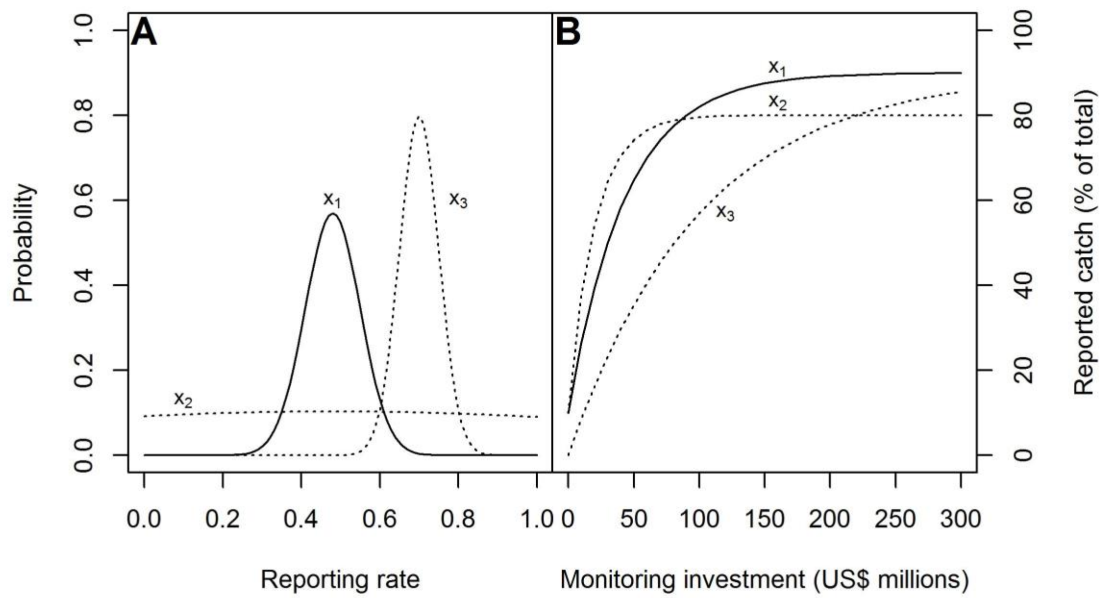

Figure 3.2. A) Probability distribution of real informal catch given alternative hypotheses on mean (u) and SD (sd) values. In x_1: u=0.48, sd=0.07; x_2: u=0.48, sd=1; x_3: u=0.7, sd=0.05. B) Effectiveness (Φ) of catch monitoring investment given alternative monitoring efficiency (k) values. In x_1: k=0.023, Φmin=0.1, Φmax=0.9; x_2: k=0.05, Φmin=0.1, Φmax=0.8; x_3: k=0.01, Φmin=0, Φmax=0.9. Solid lines are the baseline estimates.

Because the cost of monitoring to formalize catch is integrated into estimated economic impacts, expected returns decline as the marginal cost of monitoring outweighs its marginal benefit (Fig. 3.3). Investment at EV^* is inversely related to monitoring efficiency (Fig. 3.2B) and directly related to the expected increase in economic impacts as more catch is formally processed (Fig. 3.3). Assuming that the costs associated with informal catch are lower than the baseline assumption implies that less monitoring is required to maximize economic impacts (x_4 in Fig. 3.3). For example, if we assume that informal catch does not result in any economic loss compared to formal catch, Figure 3.3 would show a straight line with the intercept at estimated potential economic impacts (Table 3.1) and a negative slope equal to the cost of monitoring.

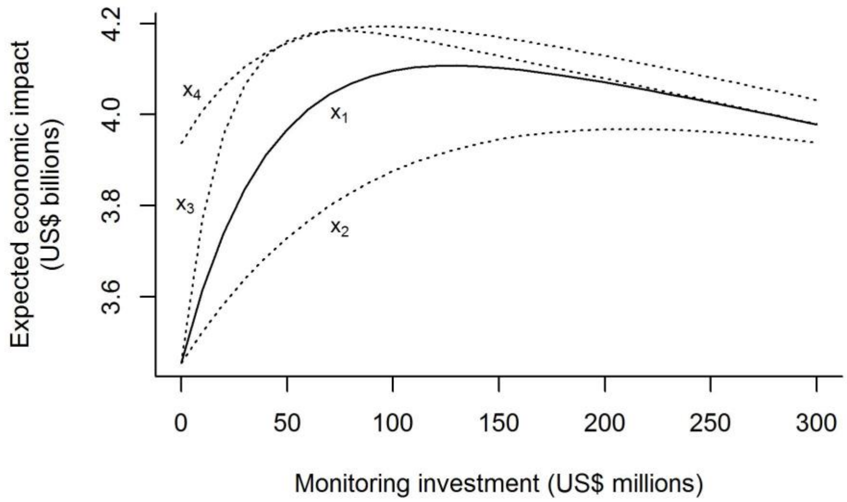

Figure 3.3. Estimated economic impacts of alternative fisheries catch monitoring investment policies in Mexico (US$ millions) given different hypotheses on the extent of losses from informal (relative to formal) catch, and monitoring efficiency (Φ) values. In x_1: Φ=0.023; x_2: Φ=0.01; x_3: Φ=0.05; x_4: Φ=0.023 and EI_{min} is 50% higher than the baseline estimate (e.g., higher assumed α and β). Solid line is the baseline estimate.

3.4 Discussion

Informal fish catch is a serious issue for resource management and sustainability; this study offers a framework to evaluate investments in monitoring and enforcement. Informal economies have significant benefits for communities that official statistics often ignore (Portes 1983), yet I argue there is an associated loss in potential value when fish are processed and marketed informally (Section 3.2). In the case study used here (Mexico), it is estimated that if all current catch was landed and processed formally, it would generate an additional yearly US$960 million in economic impacts (Table 3.1).

Given the current estimated size of the informal sector (Chapter 2), the optimal investment in monitoring is estimated at around US$130 million (Fig. 3.1), compared with the current investment of US$34 million ((Lara & Guevara-Sanginés 2012). While management in developing countries is often financially-limited, in the case of Mexico over US$200 million in fuel, gear, and infrastructure subsidies were conferred in 2011 (Lara & Guevara-Sanginés 2012), which presents an interesting opportunity for subsidy reorienting away from capacity-enhancement and towards monitoring and enforcement, as has been previously proposed (Cisneros-Montemayor 2013a).

It must be stressed that our argument here does not address improvements in the management or enforcement of fisheries policies themselves (Hilborn & Walters 1992; Pauly et al. 2002), only the economic effects of current informal catch. This allows for a straightforward calculation as proposed here, but achieving ecologically and economically sustainable fisheries must include not only analyses of current performance, but estimates of potential performance given management improvements to rebuild stocks and prevent economic waste (World Bank 2009; Worm et al. 2009; Sumaila et al. 2012).

Discarded catch is a significant component of lost potential economic impacts in Mexico (Table 3.1), and likely for other countries (Zeller & Pauly 2005). This follows from our assumption that discarded catch could have grown and/or been caught by other fishers and subsequently sold at its market price. In multi-species, heavily-fished settings such as found in Mexico and many tropical developing countries, discards are often juveniles of otherwise valuable fishes (Hall & Mainprize 2005; Meltzer et al. 2012), and there are increasingly less species without a market price (Pauly 1998b; Sala et al. 2004; Branch et al. 2010). Indeed, almost 70% of taxa in discards (representing 72% of discarded catch) also appeared in formal landings, meaning that they were discarded due to high-grading and not because there was no market for them. The remaining taxa were assigned a price of zero, though it is likely that this will change in the future if current trends continue and previously-“trash” fish are sold (Sala et al. 2004).

The value of information approach used here explicitly incorporates uncertainty in the estimated magnitude of informal catch and subsequent economic losses, arriving at an optimal investment policy given competing parameter hypotheses (Eq. 5). An important benefit of this method is that it allows us to test which uncertain parameters we should be more concerned with. In this case, the estimated optimal spending on monitoring was most sensitive to the efficiency of these investments in terms of actually formalizing catch, unless the real rate of informal catch is very low (Fig. 3.1). As with current estimated informal catch in Mexico (~50%), the latter is usually not the case (Zeller & Pauly 2007). Our confidence in estimates of mean current informal catch does not impact the estimated optimal investment policy (EV^* over the full range of uncertainty), though it does change the expected value of resolving uncertainty and choosing the best policy (EVPI; Eq. 6).

Increased regulations on fish products increase incentives for informal marketing (Schneider 2002) and, without prior enforcement and monitoring plans, can simply lead to higher unreported catch. This already occurs in fisheries as nations approve new policies for high-level treaties, but have inadequate capacity for true enforcement (Rigg et al. 2004). Therefore, a lot could be gained by ensuring that money spent on monitoring fisheries truly performs its function (Figs. 3.1, 3.3). This can include battling corruption of inspectors and managers (Kolstad & Soreide 2009), but also designing monitoring and enforcement frameworks that explicitly incentivize self-compliance (Hauck 2008). It is impossible to monitor every single fish caught (Sutinen & Kuperan 1999), yet our results suggest that improvements in monitoring to formalize catch can have significant economic benefits (Table 3.1).

Despite regulatory issues, the informal sector (in any activity) can be highly significant for the national economy (Schneider 2002; Gerxhani 2004). In Mexico, aside from lost potential value, the informal fisheries sector is currently estimated to generate US$1 billion/year, 30% of total fisheries' economic impacts in the country. However, losses from informal catch are worsened if foreign fleets are the ones fishing. This is uncommon in Mexico but occurs around the world, often involving the use of mother ships to transfer fish for processing at sea (Pauly et al. 2013; Kaczynski and Fluharty 2002). In these cases, potential economic impacts from national fish resources decrease as value is exported along with the fish (Sumaila et al. 2006).

This study focuses on the economic implications of informal catch, but other benefits of improving fishery data have been extensively discussed, including revising basic perceptions of the contribution of fisheries to society (Zeller & Pauly 2007; Agnew et al. 2009). In developing countries, socioeconomic benefits from artisanal fisheries may actually outweigh those from more politically-visible industrial ones, justifying increased attention (Zeller et al. 2007b; Teh et al. 2011; Le Manach et al. 2012; Christensen et al. 2014). More accurate data also improves biological and ecological assessments that help prioritize management investment and action (Zeller & Pauly 2005).

The economic focus presented here adds to these arguments by placing fisheries within wider policy issues around informal economies, and can be conveyed to decision-makers who may not be well acquainted with fishery-specific issues. Nevertheless, analyzing the implications of increased formalized catch for supply-demand and other market dynamics was beyond the scope of this analysis. The price of fish tends to be inelastic for most species and at large spatial scales (Swartz et al. 2013), but this may not hold at regional scales. This may lead to lower ex-vessel prices and profitability for fishers, though also to lower consumer prices linked to higher supply (a dynamic currently observed in periods of high demand during Lent; PROFECO 2011). Perhaps more importantly, formalizing fish processing and marketing in many developing countries would involve dealing with well-established informal markets and the interests of those who profit from their informality (or illegality) (Peña 1999; Kolstad & Soreide 2009). These broader but highly significant policy implications should be considered in conjunction with analytical studies such as the one presented here.

Note that if there are absolutely no regulations in place, i.e. there is no “formal” fisheries sector, formalizing the sector could still have positive impacts for economic growth, as formally-recognized fishers and processors could access more sources of loans for investing in their industry, and higher-value markets for their products (Bromley 1990; de Soto 2001). However, one would need to look to formal markets for similar industries, or other countries, to estimate potential economic benefits. One potential way to address informal catch is by promoting formal cooperative systems for fishers, particularly if these include some form of resource rights (e.g. committing to combat illegal fishing). Organized groups can more effectively bargain for adequate prices and access to value-added processing, as well as resolve potential issues among individuals so that the group complies with and has a greater say in policy (Peña 1999).

Improvements to information and decision frameworks increase the probability of meeting management objectives, however they are defined (Hilborn & Mangel 1997; Walters & Martell 2004). Economics-based approaches are useful for contextualizing management issues, incorporating key variables, anticipating outcomes and conveying options to an array of stakeholders (Clark 2006). As in virtually all countries (Zeller & Pauly 2007), informal fishing in Mexico is significant, but there are already efforts that explicitly recognize the issue and call for initial solutions (Centro de Colaboración Cívica, A.C. et al. 2013; Cisneros-Montemayor et al. 2013). Providing estimates of lost economic benefits resulting from poor governance can help elicit positive actions, by revealing the cost of inaction.

4. Integrating ecosystem data in fishery policy: economic contributions of forage fishes[3]

4.1 Integration of ecological and economic data in management

There is an evident trend towards greater inclusion of ecological and economic data and values within management frameworks, usually nested within 'ecosystem-based management' (EBM). For example, the integration of ecosystem services (that is, the benefits derived by humans from ecosystem components or functions) is generally a welcome addition to management framework language (Greiber et al. 2009). Their explicit inclusion in strategy, however, particularly in marine affairs, is not yet fully accepted (De Groot et al. 2002; Guerry et al. 2012).

This likely stems from an ongoing debate surrounding the quantitative aspects of EBM, and the perceived operational limits to its implementation (Hilborn 2011). Though the most important part of EBM is the qualitative consideration of a whole ecosystem when designing management policies and strategies, new techniques, including ecosystem modelling, have indeed it possible to formally evaluate ecosystem outcomes quantitatively in a number of ways (Plagányi 2007). While it is highly unlikely that we will ever be able to completely and accurately represent ecological dynamics in prediction models, these analyses can highlight situations in which single-species

[3]The following is adapted from the book chapter published as: Cisneros-Montemayor, A.M. (2013). Ecosystem models to explore the economic role of forage fishes. [Modelos ecológicos para analizar el papel económico de los peces de forrage]. In: Blanco, J.A. (Ed.) Application of ecosystem models to natural resource management. [Aplicaciones de modelos ecológicos a la gestión de recursos naturales] (pp. 67-76). Barcelona: OmniaScience.]

views may not fully capture ecological and economic dynamics that affect management outcomes. In particular, we might also identify areas where the suggested divergence between single-species and ecosystem-based outcomes and strategies warrants targeted investments in resolving key uncertainties, as discussed in Chapters 2 and 3. This chapter presents one such analysis highlighting the differences between perceived economic benefits of a set of species, from single-species and ecosystem views. This is then expanded in Chapter 5 to a more complete method including strategy evaluations over a range of climate scenarios.

4.2 Management of forage fishes: ecological context

Resource management may involve maximizing profits from a fish stock, maximizing catch for food consumption, or obtaining a combination of benefits toward mixed objectives including the economy, food, employment and conservation. While management goals are defined by society, production limits are bounded by nature, so resource conservation becomes an implicit objective for any sustainable management framework. In this context, there are times when particular species have a quite different ecosystem service value than that perceived by human industry, so that their optimal management becomes more complicated. One such example are forage fishes.

Forage fishes (FF) as defined here are not necessarily fish, as they generally also include groups such as squids along with sardines, herrings, anchovies, etc. Their defining characteristics are fast individual growth, high fecundity and relatively short life spans, which all contribute to the capacity to react quickly to prevailing climate conditions. Thus, their population abundance can increase explosively when conditions are favorable, but can also rapidly collapse when they are unfavorable (Pikitch et al. 2012). While it is thought that the mechanism of interaction between climate and FF abundance is mainly through juvenile survival and recruitment, it has also been recognized that overfishing of adults can exacerbate negative effects from an unfavorable climate regime (e.g., Cisneros-Mata et al. 1995b; Chavez 2003; Shannon et al. 2004).

A key characteristic of FF from an ecological (and as argued here, human) standpoint is that they provide a main energy pathway from plankton to predator species that make up most fishery stocks (Pikitch et al. 2012). Thus, tunas, jacks, sharks, etc., need FF for their energy intake during different ontogenetic stages. Although they play this fundamental ecological role, the historical abundance of FF has resulted in a low economic value (e.g., the price for a kg of sardine is much lower than one of tuna), offering a perfect example of disparity between contribution to ecosystem function and perceived economic value (Hannesson & Herrick Jr 2010).

4.3 Ecosystem-based forage fish management

Following from the above, it is important to conceptualize the management of FF differently than that of other species. Management recommendations would likely be quite different if a FF is conceptualized as a discrete species, as opposed to a support for many other species with commercial value. Although it is difficult to quantify the true economic value of FF in an ecosystem precisely, it is clear that somehow including their broader economic contribution is necessary for improved management. More information is needed to represent an ecosystem rather than a single species, yet current tools make it possible to explicitly integrate economic and ecological values within models that move towards applied ecosystem-based management. It is worth noting that this type of management is desirable in any complex natural system in addition to the marine realm.

Some management plans designed to maximize benefits from forage fish do incorporate ecological needs in the form of 'set-aside' biomass, but are mainly concerned with optimal single-species management (Herrick et al. 2007). That these strategies incorporate both biological and economic components is an important improvement on previous frameworks, but could benefit significantly from the explicit inclusion of ecological issues within analyses, which has become simpler thanks to new available quantitative methods (Pauly et al. 2000). For example, studies using ecosystem models have found that, given the market price of sardine compared to other fish, it might be economically optimal to limit sardine fisheries in order to provide forage for other, more valuable species (Hannesson et al. 2009). Though that study had a more theoretical focus in mind, the fact is that it is now feasible to estimate the total value of a species in terms of its contribution to others, potentially changing management strategy (Sumaila 1997b).

This study proposes a simple method to explicitly incorporate wider economic and ecological values in management models. Forage fish are used as a case study, but the main point is that it is possible, and necessary, to incorporate wider values within management frameworks in order to better appreciate and maximize ecosystem benefits.

Conceptual model

Forage fish (FF) are vitally important along the eastern Pacific coast; this includes Mexico and this case's study area. The most important FF species in this area are Pacific sardine (*Sardinops sagax*), anchovy (*Engraulis mordax*), and other species such as thread herring (*Ophistonema libertate*), mackerels (e.g., *Scomber japonicas*) and squids (*Loligo* spp.). Current scientific studies and fishery data suggest that, aside from fishing effort, the abundance of FF in this area has historically been driven by climate regimes, with an apparent (not necessarily causal) relationship with the Pacific Decadal Oscillation (see e.g., Radovich 1982; Baumgartner et al. 1992; Cisneros-Mata et al. 1995b; McFarlane & Beamish 1999; Chavez 2003).

Irrespective of the mechanisms for these fluctuations in population abundance, for the purposes of the arguments presented here it is more important to represent the ecosystem structure, particularly with a fisheries focus. An ecosystem model is therefore used to represent the pelagic ecosystem in the northwest Mexican Pacific that was originally used to test management strategies for pelagic fisheries (Cisneros-Montemayor et al. 2012). Table 4.1 shows annual regional landings, including estimates of unreported catch in Chapter 2, and landed value based on official ex-vessel prices (CONAPESCA 2007) in 2013 USD. The trophic level of each group is calculated as 1 + the weighted average of the trophic level of its diet items, where primary producers are = 1. So, a zooplankton species feeding exclusively on phytoplankton would have a trophic level of 2.

Table 4.1. Trophic level (TL), catch (tons), price per ton and landed value for species included in the ecosystem model. Forage fish species are highlighted in grey.

Group	TL	Catch (t)	Price/t (USD)	Landed value (USD millions)
Oceanic Sharks	4.46	7,398	1,221	9.0
Coastal Sharks	4.40	19,266	1,485	28.6
Dolphins	4.42	0	-	0
Marlin	4.87	1,450	947	1.4
Tuna	4.24	109,643	1,117	122.5
Dorado	4.26	1,808	947	1.7
Skipjack	4.21	21,176	549	11.6
Sailfish	4.42	5,613	947	5.3
Large Scombrids	4.42	5,426	1,176	6.4
Carangids	4.05	6,430	641	4.1
Small Scombrids	3.80	8,315	75	0.6
Squids	3.69	63,687	191	12.2
Flyingfish	3.25	0	0	0
Small Pelagic Fish	3.08	884,568	56	49.5
Mesopelagic Fish	3.25	0	-	-
Zooplankton	2.25	0	-	-
Phytoplankton	1.00	0	-	-

The *Ecopath with Ecosim* (EwE) platform allows for the modelling and analysis of an ecosystem, under the key assumption that linkages between species can be represented in terms of their biological characteristics (biomass, growth and mortality rates) and the trophic interactions between predators and their prey. Of course, in every case except for primary producers, every species is both a predator and a prey. The development, applications and limitations of EwE have been published in detail (e.g. Christensen & Walters 2004; Christensen et al. 2005) and there is ample literature on applications to a variety of systems and questions.

In the simplest terms, under the assumption that there is mass-balance in an ecosystem, the production (P) of any species (*i*) is equal to:

$$P_i = Y_i + M_i + E_i + BA_i + MO_i \qquad [1]$$

where Y= fishing mortality, M= predation mortality rate, E= net migration rate, BA= biomass accumulation and MO= natural mortality other than predation (old age, disease, etc.). Predators and preys are linked through predation mortality, where M depends on the proportion of each species in the diet of others, the predators' consumption rate and the rate of conversion from food into biomass (efficiency). Figure 4.1 is a graphic representation of the base model used for the subsequent analysis.

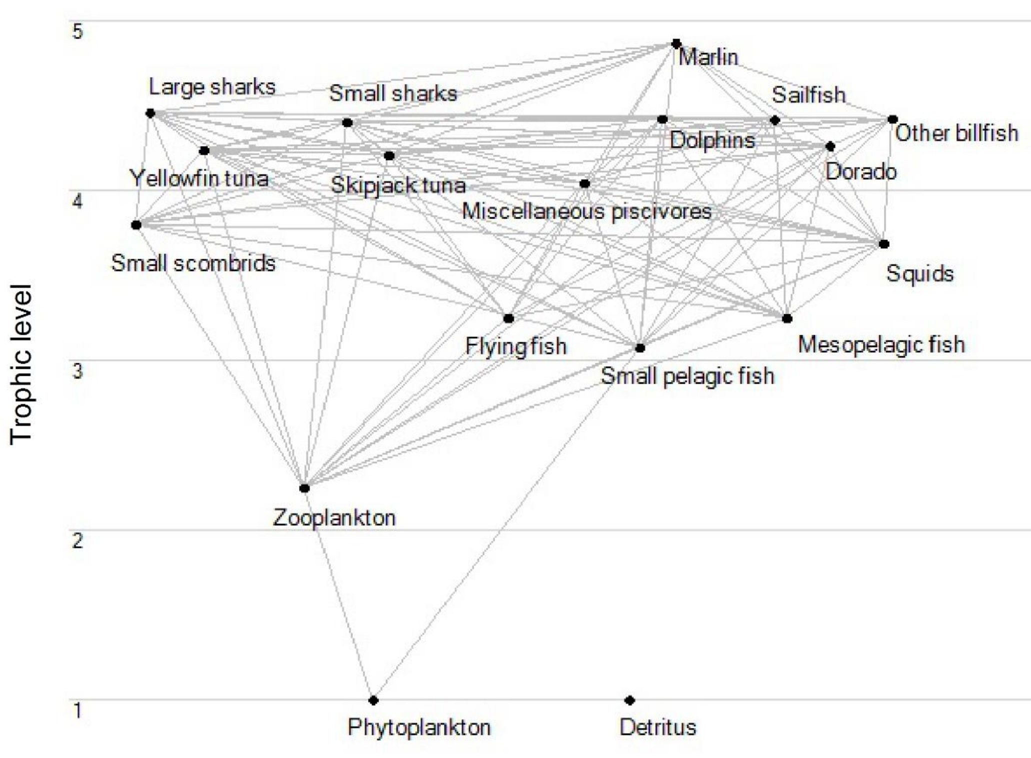

Figure 4.1. Trophic diagram of the Mexican Pacific pelagic ecosystem. Lines indicate predator-prey links. Model described in (Cisneros-Montemayor et al. 2012).

Once biological and diet parameters are established for each species, it is straightforward to calculate the economic value that each species contributes to the ecosystem. Thus, the indirect value (IV) of each prey species *i* to the commercial fishery value of each predator *j* becomes:

$$IV_{i,j} = DT_{i,j} \cdot CN_j \cdot CV_j \cdot LV_i \qquad [2]$$

where DT= the proportion of a species in the predator's diet, CN= consumption rate, CV= conversion rate of intake into biomass, and LV= fisheries landed value.

Economic value of forage fishes in Eastern Baja California Sur

The result of this relatively simple model is meant to be theoretical, as landed and supporting service values are not additive in a static model (this would represent double-counting). Nonetheless, it exemplifies the main argument of this study: many species within an ecosystem, and particularly those that function as forage fishes, are undervalued in the market (Table 4.2).

Table 4.2. Landed (“Direct”), supporting service (“Indirect”) and total market (“Total”) values by species group. Forage fishes highlighted in grey. Values are in 2013 USD millions.

	Market value (USD millions)		
Group	**Direct**	**Indirect**	**Total**
Oceanic Sharks	9.03	0	9.03
Coastal Sharks	28.6	0.197	28.8
Dolphins	0	0.265	0.265
Marlin	1.3	0.104	1.4
Tuna	122.4	0.123	122.5
Dorado	1.7	3.5	5.2
Skipjack	11.6	8.5	20.1
Sailfish	5.3	0.080	5.3
Large Scombrids	6.3	0.106	6.4
Carangids	4.1	3.8	8
Small Scombrids	0.623	19.7	20.3
Squids	12.1	36.8	49.05
Flyingfish	0	10.4	10.4
Small Pelagic Fish	49.5	49.8	99.3
Mesopelagic Fish	0	32.9	32.9
Zooplankton	0	78.02	78.02
Phytoplankton	0	8.2	8.2

Again, indirect value here represents the monetary value that a species has as food for other fished species. The relationship between the abundance of single forage fish species and the abundance of their predators is not expected to be linear, but there would be serious consequences for an ecosystem if forage fishes, and the energetic link they represent, were removed completely. These ecological linkages become vital to consider when designing management strategies, as assuming that species function independently from one another seriously violates our knowledge of the structure of marine ecosystems. In terms of price per ton of catch, which is often the only metric considered, the most valuable species in this ecosystem would be tunas and sharks (Table 4.1). However, when indirect values are included in the total economic (market) value, the most important fished species become sardines ('Small Pelagic Fish'), squids and mackerel ('Small Scombrids') (Table 4.2), all forage fish.

Given this new information and recognizing the value of species as food for others with higher market prices, a new strategy could decide to decrease their total catch during certain seasons. Or, a (sustainable) fishery could be started for species that limit system food abundance. This method is by no means complete, as it ignores the non-linear dynamics that shape ecosystems. For example, it does not account for top-down control, opportunistic feeding strategies, or foraging arena dynamics. These wider dynamics are incorporated in Chapter 5, but it is nonetheless clear that, with a more complete picture of the real economic value of species within a managed ecosystem, strategies must necessarily change to remain optimal. This is one key value of gathering and integrating information into a conceptual model that more closely represents complex and variable ecosystems.

4.4 Limitations and future research

Formally integrating broader economic and ecological values requires more information that necessary for a single-species assessment. However, as shown here, this should not preclude using a relatively simple model to capture key aspects of ecosystem-based management in an economic context. For example, the approach used here only considers bottom-up and not top-down dynamics, when both occur in nature. Most importantly, while it is easy to treat marine species as discrete entities for management, this simply does not align with reality. Dynamics within marine ecosystems, like any other, result from a complex web of connections between many species; at the very least, it is necessary to recognize this within management frameworks.

The explicit representation of recognized complexity within management has become easier thanks to advances in computing power and methods that allow for the development of ecosystem models that are as simple or complex as the information available. For example, the platform used here, *Ecopath with Ecosim*, is free and is trending towards open-source programming. There are already hundreds of publications, a user guide and online communities for support and application of ecosystem models to particular situations and contexts. In addition, there are other platforms, such as *Atlantis* (*http://atlantis.cmar.csiro.au*), that seek a much more complex representation of an ecosystem, for example integrating oceanographic, chemical and climatic information in models. Although this requires a much greater amount of data, the fact is that representing an ecosystem in a way that is useful and applicable to management is already possible (for a comparative review of various ecosystem modeling platforms, see Plagányi (2007).

A real limitation is the acceptance of operational ecosystem-based management by marine resource stakeholders including academics, administrators and the fishing industry, because, as outlined here, it can significantly change 'optimal' management recommendations. Any change to existing management schemes carries the challenge of convincing those affected that the new strategy is better than the previous one. For these cases, it is probably most convenient to use relatively simple models (like the one used here) to test the overall management recommendations brought on by a wider vision of the ecosystem and the economic benefits it provides. This could help identify situations where investments in achieving ecosystem-based management would be expected to bring increased benefits to those involved, thereby easing policy transition. More accurate models can then be developed to better capture these broader ecosystem dynamics.

A key point may be recognizing when broader ecological and economic values can and should be included within a mechanistic analysis like the one presented here. In this case, only direct and indirect market values from forage fishes, within a very limited model, have been included to support the main argument that the market often does not reflect the true value of species in an ecosystem. The same argument could be extended to other values, such as recreation (e.g. sport fishing or whale watching), option (e.g. letting a forest stand so our children have the option of using it as they will) or culture (e.g. the many plants and animals with traditional values to various human groups). While it is true that all of these values cannot always be well represented by quantitative models, it should not preclude their inclusion at the discussion table between different stakeholders concerned with gaining the most benefits from natural resources.

5. Single-species vs. ecosystem views in transboundary fishery strategies

5.1 Background

Transboundary stocks add complexity to traditional fisheries management, as agents react to the actions of others as well as to their own needs (Munro 2009; Munro & Sumaila 2012). The case of Pacific sardine (*Sardinops sagax*) in the California Current is a useful analytical study species, as it is shared by three major agents, Canada, the US and Mexico, and is subject to climate variations that are both difficult to predict and can counteract fishing strategy goals. The application of three-agent bionomic frameworks incorporating environmental effects on abundance and distribution have shown that this variability can prevent stable cooperative exploitation of the stock by all three countries (Ishimura et al. 2013a).

This study, however, moves beyond a single-species analysis and incorporates wider ecosystem benefits to each player. Because fisheries differ among the three countries, the ecological and economic contribution of a species to other fisheries, in the form of supporting services, could shift the balance of the game. Incorporating these ecosystem economic values is vital for the adequate development of fishing strategies, particularly in light of increasing mandated ecosystem approaches to management (Daily et al. 2009). In this context, ecosystem models are used as a platform to explicitly incorporate benefits to other fisheries following from single-stock management strategies. This follows from the analyses in Chapter 4, but using a method that better represents ecosystem dynamics. The results of a range of cooperative and non-cooperative fishing strategies are then tested to identify the ones that perform best under a range of potential climate scenarios.

Game theory in transboundary fisheries

The application of game theory in fisheries has provided insightful predictions on the stability of cooperation in transboundary stocks, as well as strategies to improve cooperation when it would not otherwise occur (Munro 1979). Most advances in game theory applied to cooperative fisheries management have been in the context of increased complexity of the game structure and player strategies (Bailey et al. 2010; Hannesson 2011b). Cooperative stability can thus be tested with various system shocks, such as changing prices, discount rates, open-access and coalition scenarios (Bjørndal et al. 2000; Pintassilgo & Lindroos 2006), and environmentally-driven fluctuation of fish stocks (Sumaila 1999; Ishimura et al. 2013b; Bailey et al. 2013).

This continued broadening of game theory from the theoretical to the applied may prove crucial for aiding cooperative management of the world's shared fish stocks (Miller & Munro 2004), and has already provided a platform for hypothesis testing at different scales (Bailey et al. 2013).

Ecosystem view of fishery benefits

Walters (1986) defined four "categories of interaction" within an ecosystem: production (food supply), natural mortality, physical structure and chemical environment. These categories are reflective of the various key types of ecosystem services as currently understood and 'valued'. With minor modifications and in the same order, they represent the range of services from the most easily quantifiable to the most difficult. First comes production, i.e., marketable fish (for food or recreation); then supporting services through predation by other marketable fish (Chapter 4); then physical habitat services, e.g., kelp forest and coral reef structure (Cesar 2002); and finally, the most difficult and least immediately intuitive services, such as O_2 production, CO_2 sequestration, chemical buffering and natural pollution treatment (De Groot et al. 2002). It is logical that inclusion of values beyond those currently incorporated in valuation frameworks will increase expected payoffs, but this may or may not shift single-stock-based cooperative strategies.

There have been previous frameworks developed to estimate net benefits derived indirectly from a fish stock given ecosystem trophic dynamics and/or game-theoretic fishing strategies. These include, for example, game-theoretical analyses with two linked species and two players (Sumaila 1997b) or one species and three players (Hannesson 2007; Ishimura et al. 2013b). The comparison between landed value and supporting service value has also been addressed, either focusing on one species (Hannesson & Herrick Jr 2010) or a set of species (usually forage fish) within an ecosystem (Chapter 4; Okey et al. 2014). To our knowledge, however, this has not yet been extended into game-theoretic models of transboundary cooperation within an ecosystem framework.

Preliminary analysis

According to the most recent available official data, the California Current supports fisheries with a total landed value of approximately US$950 million per year (Table 5.1). The three countries that border on this large marine ecosystem target an array of species, but in terms of species groups, the most valuable overall are invertebrates, followed by benthopelagic fishes and salmon. Sardine fisheries currently generate around US$30 million per year in landed value (Table 5.1).

Table 5.1. Current annual landed value by country off the California Current (CONAPESCA 2013; DFO 2014; NOAA 2014). Ex-vessel price estimated using total catch (metric tons) and value (2014 USD). Average profit margin calculated based on per-gear total fishing costs reported in Lam et al. (2013). For full table, see Appendix C.

	Landed Value (USD millions)				**Avg. Price**	**Avg. Profit**
Group	**Canada**	**USA**	**Mexico**	**Total**	**(USD ‘000/mt)**	**(%)**
Crustaceans	22	240	27	289	4.7	58
Mollusks	20	133	11	164	11.0	87
Benthopelagic Fish	43	48	0.2	91	2.7	14
Salmon	39	49	-	88	4.4	78
Squids	-	66	2	68	0.7	17
Tunas and Billfish	5	50	3	58	3.2	31
Flatfishes	32	21	1	54	2.1	24
Cod-likes	16	22	-	39	0.3	7
Other Invertebrates	6	16	13	35	2.7	57
Sardine	-	22	11	33	0.1	0
Small Pelagic Fish	11	3	0.2	13	0.6	12
Pelagic Fish	3	1	3	8	0.5	32
Other	-	-	5	5	0.6	62
Elasmobranchs	1	1	3	5	0.7	10
Total	**197**	**673**	**80**	**950**	**2.8**	**30**

There are limited available total abundance estimates by species in the California Current, but overall catch data suggests that a relationship exists between catch of Pacific sardine and the total catch of other species in the ecosystem (Fig. 5.1). If concurrent empirical abundance estimates were available for all groups in the system, a regression model could be used to calculate the net effect of one stock’s abundance on other species. However, it would be very difficult to determine the causal competitive or trophic relationships between groups, and if these were furthermore driven by or correlated with prevailing and varying environmental conditions. Furthermore, this estimation would be confounded by the fact that fishing effort and catchability — that also impact abundance — are almost certainly not constant over time.

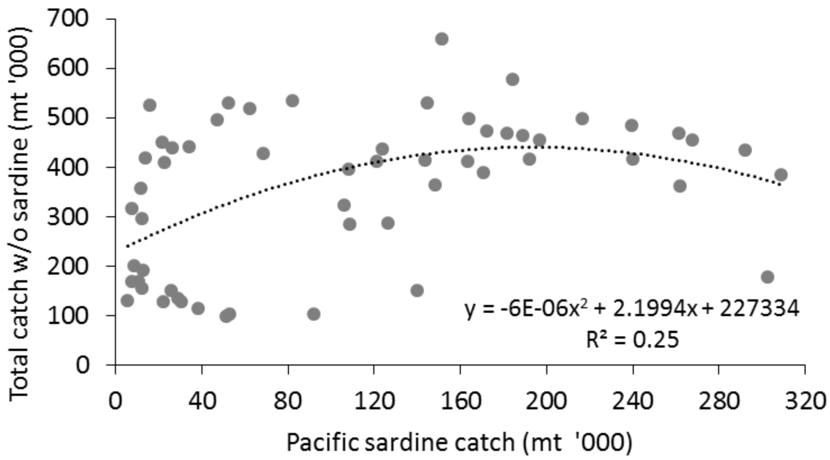

Figure 5.1. Catch of Pacific sardine (Sardinops sagax) versus other species in the California Current (Sea Around Us 2014; *seaaroundus.org*).

In this context, ecosystem models provide a virtual arena to explore inter-species interactions with all other factors held constant or controlled for. They are used in that capacity here, but it must be stressed that this is not an ecosystem modeling exercise *per se*. Rather, I use existing ecosystem models as a tool to test the effects of contributions from a given species to the biomass (and subsequent landed value) of other species in the ecosystem.

5.2 Methods

This study develops an integrated model which includes temperature-driven population dynamics for a single species (sardine in this case). The resulting stock distribution among the marine areas of the parties along the entire transboundary range is also dependent on climate and, after fishing rate policies acted on by each party, is used as an input to other species in a set of three ecosystem models. The results of fishing policy for each party are thus evaluated in terms of single and multiple species values, over a range of temperature and game-theoretic policy scenarios. The methods are first presented in general terms so they can be applied to other settings. I then explain the parameters used in this application, Pacific sardine (*Sardinops sagax*) in the California Current.

Single-species population dynamics

Following from previous studies (Hannesson 2007; Ishimura et al. 2013a), temperature (T) is the ultimate driver of the model, representing prevailing climate regimes. Other climatic indices can easily be employed instead of temperature, but this parameter has proven to be very useful, particularly for highly-variable forage fish populations. Scenarios include multiple possible temperature time series where temperature at each time step (t) depends on the previous temperature, a regime-specific trend parameter (μ), and a random error term (σ, [-1, 1]) (Eq. 1). This parameter is taken from a random uniform distribution and can be complemented with a scale parameter to modify the influence of this random error on temperature at a given time (Ishimura et al. 2013a).

$$T_{t+1} = T_t + \mu \cdot \sigma \qquad [1]$$

Because climate can affect not only the abundance, but also the distribution of fish stocks (Cheung et al. 2008), stakeholders along the distribution range can expect varying shares of total abundance. The distribution of the stock ($D_{x,t}$) is calculated as the proportion of the yearly biomass in each player's area, where x is one of three players and t is the time step, and is estimated as:

$$\begin{cases} for\ x = 1,\ D_{x,t} = max\,\{0\,, min\,[1\,, \left(T_{H,x} - T_t\right) / \,T_{H-L,x}\,]\,\} \\ for\ x = 2,\ D_{x,t} = \left(1 - D_{1,t}\right) \cdot\ max\,\{0\,, min\,[1\,, \left(T_{H,x} - T_t\right) / \,T_{H-L,x}\,]\,\} \\ for\ x = 3,\ D_{x,t} = 1 - (D_{1,t} + D_{2,t}) \end{cases} \qquad [2]$$

where T_H and $_L$ are the upper and lower temperature thresholds, respectively, for sardine in each country. This equation was developed by Ishimura et al. (2013a), and could be further generalized to allow for more than three players. For transboundary stocks, the share of the stock for any one player at a given time is between 0 and 1, and the sum over all players must always equal 1. For straddling stocks where a portion enters the high seas and might not be accessed by any of the players, this portion can be allocated to an extra "player" representing the inaccessible area.

At each time step, temperature is assumed to have a linear effect on sardine abundance (*B*) through environmental carrying capacity (*K*) (Eq. 3). Biomass and catch are subsequently estimated based on the initial abundance (B_0), distribution (*D*), and fishing rate (*F*) by country:

$$K_t = B_0 * T_t \,/\, T_{t=1} \quad [3]$$

$$B_{t+1} = B_t + r \cdot B_t \cdot (1 - B_t \,/\, K_t) - C_t \quad [4]$$

$$C_t = \sum_x B_t \cdot D_{x,t} \cdot F_{x,t} \quad [5]$$

Note that, while I assume a positive relationship between temperature and total sardine abundance here, this assumption can be relaxed, reversed, or substituted for another functional shape or driver by modifying Eq. 3. Likewise, the rate of population growth (*r*) or the form of the population growth function (Eq. 4) can be substituted for other types that better approximate known dynamics. In Eq. 5, catch of the transboundary stock (*C*) is assumed to occur at the end of each time step, where *F* is the country-specific fishing rate (see *Game structures* below).

Ecosystem-level fishery effects

The section above details single-species population dynamics, and can be used to test the payoffs of alternative cooperative strategies in terms of, for example, the landed value of the model organism (e.g. sardine) (Ishimura et al. 2013a, 2013b). However, I draw on an ecosystem-based approach to management and focus on the benefits derived from multiple species with direct and indirect interactions within an ecosystem. In particular, the role of forage fishes such as sardines, anchovies and herrings as an energy pathway from planktonic to larger predator species has been highlighted as a key ecosystem supporting service (Bakun et al. 2009; Pikitch et al. 2014), and is captured here.

Ecopath with Ecosim (EwE) is an ecosystem modelling platform that uses mass balance equations and specified trophic linkages to represent an ecosystem (Christensen & Walters 2004). It is an ideal platform for this type of analysis (Plagányi 2007) and, due to its popularity, there are hundreds of EwE models available that could be used in the framework provided here (Palomares et al. 2009). In Ecopath, the initial EwE module, the essential parameters for each functional group are biomass (B), production to biomass ratio (P/B), consumption to biomass ratio (Q/B) and ecotrophic efficiency (EE), and a diet matrix specifying the proportion of each functional group in the diet of others. Given these parameters, the production (P) of each functional group (i) is calculated as:

$$P_i = Y_i + \sum_j M_i \cdot B_j + E_i + BA_i + MO_i \qquad [6]$$

where Y is the total fishery catch rate, M is the instantaneous predation rate, E is the net emigration rate (emigration – immigration), BA is the biomass accumulation rate, MO is sources of mortality other than fishing and predation and *j* are predator species (Christensen et al. 2005). Dynamic biomass growth B in time interval t is calculated in the Ecosim module of EwE as:

$$\delta B_i / \delta t = g_i \cdot \sum Q_{ij} - \sum Q_{ji} + I_i - (MO_i + F_i + e_i) \cdot B_i \qquad [7]$$

where, for each prey group *i* and predator group *j*, g is the net growth efficiency, F is the fishing mortality rate, and e and I are the emigration and immigration rates (Christensen et al. 2005). 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 & Walters (2004).

Using the built-in biomass forcing tool in Ecosim (EwE version 6.4.2, July 2014; *ecopath.org*), the relative abundance of a single species (in this case sardine, see *Case study* below) was varied from 0 to 10 times the baseline value, and the equilibrium effects on the biomass of all other species groups were noted for each scenario.

The resulting data represent the abundance of each species group given the abundance of sardine (or whatever group is being varied) in the system, with all other things being equal. This allows for the application of a regression (second-order polynomial) model to calculate the net impacts on each species. This equation and its parameters are central to the subsequent results because they determine the direction and magnitude of relationships between the stock being varied and the other species in the model ecosystem, and this type of regression was selected here to capture the non-linear dynamics observed in realistic ecosystem models. Unlike approaches that gauge this contribution using only Ecopath data (e.g., Chapter 4; Hannesson et al. 2009), using Ecosim scenarios captures both bottom-up and top-down dynamics (Shannon 2000), though remains limited to trophic interactions unless further specified in the model.

In Eq. 8, the biomass shares ($B_{x,t}$) of the transboundary stock are estimated at each time step based on the total biomass of the stock and the proportion of the stock for each player (Eq. 2 and 4):

$$B_{x,t} = B_t \cdot D_x \quad [8]$$

Using a corresponding ecosystem (or multispecies) model as explained above, the relative abundance (A) of each species group i in country x is calculated as:

$$A_{i,x,t} = \alpha_1 \cdot B_{x,t}^2 + \alpha_2 \cdot B_{x,t} + \alpha_3 + \varepsilon \qquad [9]$$

where the α terms are calculated from a second-order polynomial regression after varying the stock's biomass from 0 to n times the baseline (here, n=10). Combinations of negative and positive α_1 and α_2 values of various magnitudes can represent linear, concave or exponential functions, and α_3 represents the value at the origin, i.e. the abundance of a species when the stock being varied equals zero. Note that, while this study focuses on the role of a transboundary forage fish as a model organism, Eq. 9 can be applied to any species in an ecosystem model and would yield similar results to a keystone analysis comparing the overall impact of individual species' abundance changes, weighted by their initial abundance in the model system (Libralato et al. 2005).

Following from Eq. 9, and assuming that fishing effort remains constant, the yearly landed value (LV) for each species group and country is estimated based on the baseline landed value for each country as:

$$LV_{i,x,t} = A_{i,x,t} \cdot LV_{i,x,t=0} \qquad [10]$$

$$NPV_x = \sum_{i,t}(LV_{i,x,t} \cdot \pi_{i,x}) \ / \ (1+d)^t \qquad [11]$$

In Eq. 11, net present value (NPV) from fisheries in each country is estimated assuming a discount rate (d) and profit margins (π) for each species group and player (e.g., Table 5.1).

The inclusion of available cost data allows for comparison between net present value, total discounted value (the sum of discounted landed value over the time period) and average landed value. The latter is perhaps the most intuitive metric, and is usually much better understood by commercial fishers. These performance metrics are evaluated for each temperature scenario, fishing strategy and country. Note that the effects of different discount rates can have significant implications for policy recommendations, and have been found to be a key potential driver of fishing strategy (Sumaila 2004; Sumaila & Walters 2005). This study currently assumes a discount rate d of 3%, though this can be easily changed, preferably adopting different discount rates for each country.

Game structures

Cooperative coalition structures can be of three types, full (when all players cooperate), partial (when there is at least one non-cooperating free-rider) and non-cooperative (Lindroos et al. 2005). For a three-player game, this results in three possible coalition structures: a grand coalition $\{x_1, x_2, x_3\}$, partial cooperation $\{x_1, x_2\}$ $\{x_3\}$, $\{x_1, x_3\}$ $\{x_2\}$, $\{x_2, x_3\}$ $\{x_1\}$ and non-cooperation $\{x_1\}$ $\{x_2\}$ $\{x_3\}$.

For this study, I adapt fishing strategies proposed by Ishimura et al. (Ishimura et al. 2013b) as follows:

- Full cooperation: all players fish at F= fishing rate at maximum sustainable yield (F_{MSY});
- Partial cooperation: cooperating players fish at F= F_{MSY}; free rider fishes at F= 1 if its stock share < 0.5 and F_{MSY} if its stock share >= 0.5;
- Non-cooperation, pragmatic: each player fishes at F= 1 if its stock share < 0.5 and F_{MSY} if its stock share >= 0.5;
- Non-cooperation, windfall: each player fishes at F= 1 if its stock share > 0.5 and F_{MSY} if its stock share <= 0.5;
- No sardine fishing: F= 0 for all players.

Note that the non-cooperative structures are similar, but represent quite different fishing strategies. In the "pragmatic" strategy, players are assumed to fish at a sustainable rate (F_{MSY}) only when they have a large enough stock share and therefore an incentive to conserve the stock for the future (Hannesson 2007). If their stock share is small, players instead opt to "just fish whatever we can". On the other hand, the "windfall" strategy uses the same reference point, but players opt to fish more conservatively if they are confronted with a small stock share, fishing all they can when natural fluctuations increase abundance in their waters.

In all cases, it is assumed that only annual landed value, total discounted value, and net present value (NPV) are evaluated by players, for both single and multiple-species. The use of these three metrics is intended to represent the various viewpoints often expressed by resource stakeholders, whose interests may include how much revenue is being made annually, how much total revenue, or net value (revenue minus costs), can be expected over a given time period. Other metrics, such as minimum allowed population thresholds, are not implemented here, but have been explored in the past in similar modelling exercises as trade-offs to pure market performance (Ishimura et al. 2013b).

Case study: Pacific sardine in the California Current

The California Current is a Large Marine Ecosystem spanning from the southern coast of British Columbia, Canada, the US coast along Washington, Oregon and California, to the northern Baja California Peninsula in Mexico (Lynn & Simpson 1987). A transboundary stock of Pacific sardine (*Sardinops sagax*) is distributed along this ecosystem, and has been previously identified as an useful model organism due to its importance for ecosystems and fisheries, and its large climate and fishery-driven fluctuations in both distribution and abundance (Chavez 2003; Hannesson & Herrick Jr 2010).

There are many excellent reviews detailing various aspects of sardine research in the region, including its historical trends (Baumgartner et al. 1992), fishery dynamics (Radovich 1982; Wolf 1992), ecological importance (Hannesson et al. 2009; Hannesson & Herrick Jr 2010) and significance for international cooperative management (Ishimura et al. 2013a, 2013b). Following from Ishimura et al. (2013a, 2013b), this study assumes an initial biomass (B_0) of 1.2 million metric tons of sardine along the California Current, and a population growth rate (r) of 0.27. These starting parameters can easily be changed, or an alternative population dynamics function can be used to perhaps better describe the model species.

Using the framework detailed above, the ecosystem-wide economic impact to each player (Mexico, US, Canada) of regional sardine abundance as it relates to climate change is estimated and used as an alternative objective function to the single-species approach developed in Ishimura et al. 2013a. This is performed here using two separate ecosystem models for the California Current representing Canadian and US areas.

The Canada model was developed to represent the southwest coast of Vancouver Island, British Columbia, the main area where Pacific sardine is distributed in the region. Parameters were based on published models developed for the Strait of Georgia (Martell et al. 2002), southern BC shelf (Pauly & Christensen 1996), BC shelf (Preikshot 2005) and northern California Current (Field et al. 2006). The model was reviewed with experts including government and university scientists, and local fishers. Particular attention was placed on diets, which often have more local variability than other parameters such as species growth rates. Input parameters and trophic linkages are presented in Appendices D and E. Of the 33 groups represented in the model, 19 were linked directly to sardine either as predators or prey. The US model (Field et al. 2006; Field & Francis 2006) includes 25 species groups, though 6 are further represented by multiple life stages (i.e., larval, juvenile, adult) (Appendices F and G). In this model, only 4 groups are directly linked to sardine. There were no available models for Mexico, so values for the US are used. While not ideal, I argue that the coasts off Baja California are similar to the adjacent US, so that models for the latter are reasonably representative.

Available data on current catch and landed values for the three countries along the California Current was gathered from official sources (CONAPESCA 2013; DFO 2013; NMFS 2014) and aggregated by broader species groups to allow for better comparisons. The same species groupings were assigned to species in the ecosystem models after calculating the effects of sardine abundance changes (polynomial regression given 0 to 10 times baseline sardine abundance) to obtain the α parameters in Eq. 9.

Profit margins (π in Eq. 11) were calculated based on total (including fixed and variable) fishing costs per ton for corresponding fishing gears and players (Lam et al. 2011) (Table 5.1, Appendix C). Based on these cost estimates and official price data (CONAPESCA 2013; DFO 2013; NMFS 2014), in some cases the reported fishing costs per ton exceeded the reported ex-vessel price for the same species; profit margin was set to zero in these cases. This result may be partly due to the resolution of cost data (Lam et al. 2011), but could also reflect the effect of profit-enhancing subsidies (Sumaila et al. 2010) or vertically-integrated fishing firms operating their fishing sector at a loss that is made up in the processing sector (Sumaila et al. 2012). This calculation uses available data to provide an approximation to fishing cost, but more importantly does not include basic bioeconomic dynamics, specifically effort-dependent cost and abundance-dependent catch per unit effort. This is one key area for improvement in future work.

These parameters, together with baseline landed values and fishing strategy effects on sardine were used to estimate subsequent landed values and net present value by country and species (Eq. 10, 11). Fishing strategies were implemented as detailed above, with two partial cooperation scenarios where either Mexico or Canada are free riders. Biomass of other species given sardine abundance represent equilibrium model results.

5.3 Results

Biomass forcing functions were used in the Canada and US California Current models to obtain regression coefficients for species abundance given sardine abundance (Appendix H). A representative net impact on species (aggregated into species groups) over all countries is presented in Figure 5.2, where relative sardine abundance has been increased by 10 times.

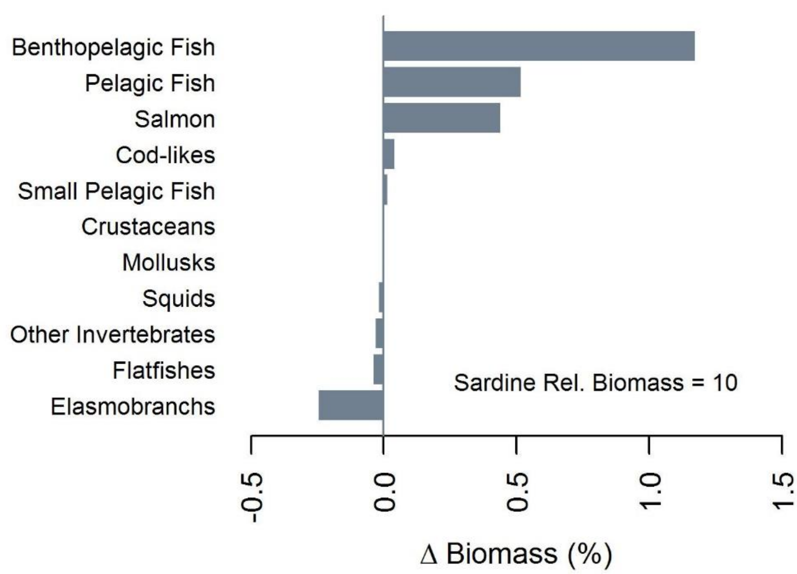

Figure 5.2. Net biomass change (including all countries) for fished groups in the California Current ecosystem models given change to sardine abundance.

All other things being equal, average sardine catch (and value) increases as fishing rate on sardine is increased from zero to a maximum at F_{MSY} (Fig. 5.3). The concurrent reduction in sardine biomass negatively impacts catch and value for other fisheries due to a net reduction in available prey (Fig. 5.2). This loss is offset by sardine landed value so that the total value is increased with sardine F up until just less than the sardine F_{MSY} (Fig. 5.3). Under current assumptions, a single-owner would thus maximize total sardine annual landed value by fishing at F= 0.17, total annual landed value at F= 0.15, and annual landed value of all species other than sardine at F= 0 (Fig. 5.3). At fishing rates higher than sardine F_{MSY}, however, reductions in sardine abundance due to overfishing result in a net decrease in sardine, other species', and total landed value (Fig. 5.3). Note that only species that appear both in official catch statistics (Table 5.1) and ecosystem models (Appendices C and E) are included, so that landed values in results are slightly less than the total from available data (Table 5.1). Furthermore, different assumptions on the initial biomass of sardine would change its estimated potential sustainable yield, and its value relative to that of other fisheries.

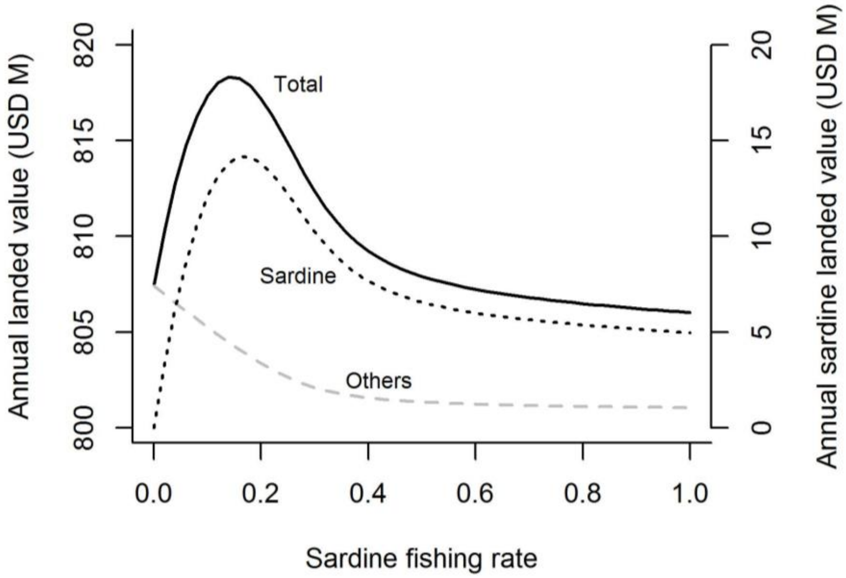

Figure 5.3. Total and sardine average landed value per year at baseline scenario given sardine fishing rate applied to entire stock, all other things being equal. Sardine annual landed value maximized at F= 0.17; Total annual landed value maximized at F= 0.15.

Each of the various game structures can have different performance given prevailing temperature trends, both for sardine and total landed values (Fig. 5.4). Among all strategies, cooperation between at least two players outperforms other management structures (Co, M, and C in Fig. 5.4).

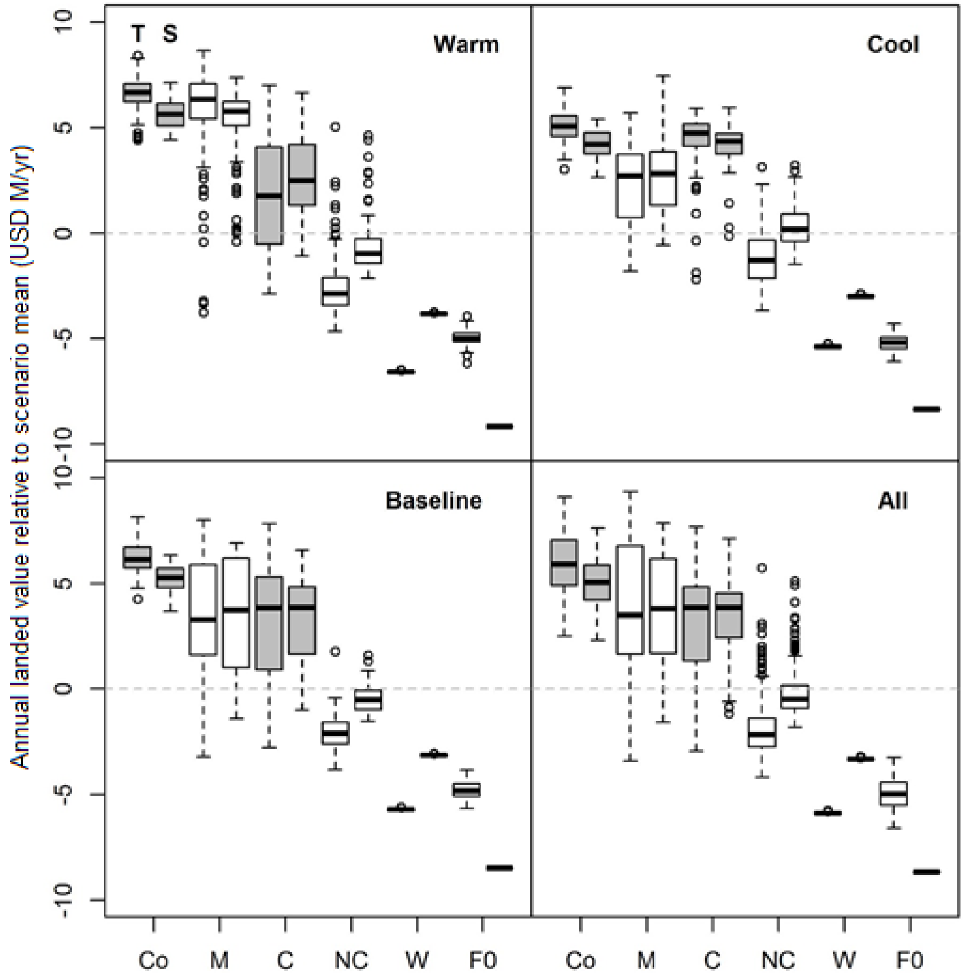

Figure 5.4. Total and sardine annual landed value per year given multi-player fishery strategy, relative to the overall mean for each temperature scenario (n=10,000). Co= full cooperation; M= Mexico free rider; C= Canada free rider; NC= non-cooperative pragmatic (players fish all available sardine when their stock share <0.5), W= non-cooperative windfall (players fish all available sardine when their stock share >0.5); F0= no sardine fishing.

Gains in average landed value of other species from leaving sardine unfished are not enough to offset losses to the sardine fishery itself (Table 5.2, 5.3). However, unchecked sardine fishing may result in a strategy to maximize total discounted value by quickly overfishing the stock as occurs in the non-cooperative scenarios, which negatively impacts other fisheries (Table 5.2, 5.3).

Because profits from sardine fishing are estimated to be zero (Lam et al. 2011), net present value from all fisheries is maximized when sardine is not fished (Table 5.2, 5.3). In Table 5.2, the above results are presented relative to the performance of the non-cooperative (pragmatic) fishing strategy, the most similar strategy to current management, over all temperature scenarios. This provides a point of reference for outcomes of moving away from current management. Table 5.3 presents results relative to all strategies, and represents the relative value, over all temperature scenarios, of selecting a particular strategy compared to choosing randomly among all strategies.

Table 5.2. Mean performance of management strategies for all players relative to non-cooperative (pragmatic) strategy, over all climate scenarios (n=1000). Values are in 2014 USD millions.

Total relative to non-cooperative	**Annual landed value (USD millions /year)**			**Total discounted value (USD millions)**			**Net present Value**
Strategy	**Sardine**	**Others**	**Total**	**Sardine**	**Others**	**Total**	**(USD millions)**
Full Cooperation	5.3	2.7	8	65	59	125	14.9
Mexico Free Rider	4.2	1.6	5.8	64	33	97	8.6
Canada Free Rider	3.7	1.5	5.1	53	31	84	8
Non-Cooperative (Windfall)	-3	-0.7	-3.7	-66	-19	-86	-3.9
No Sardine Fishing	-8.4	5.6	-2.8	-257	119	-138	30.3
Absolute mean (Non-cooperative)	**8**	**802**	**810**	**257**	**18,037**	**18,294**	**7,980**

Table 5.3. Mean performance of cooperative management strategies for all players relative to mean over all strategies and climate scenarios (n=1000). Values are in 2014 USD millions.

Total relative to all strategies	**Annual landed value (USD millions /year)**			**Total discounted value (USD millions)**			**Net present Value**
Strategy	**Sardine**	**Others**	**Total**	**Sardine**	**Others**	**Total**	**(USD millions)**
Full Cooperation	5	1	6	89	22	111	5.3
Mexico Free Rider	3.9	-0.2	3.7	87	-4	83	-1
Canada Free Rider	3.4	-0.3	3.1	77	-6	71	-1.7
Non-Cooperative (Pragmatic)	-0.3	-1.8	-2.1	23	-37	-13	-9.6
Non-Cooperative (Windfall)	-3.3	-2.5	-5.8	-43	-56	-100	-13.6
No Sardine Fishing	-8.7	3.8	-4.9	-233	82	-152	20.7
Absolute mean (All strategies)	**8.7**	**803**	**812**	**-233**	**18,074**	**18,308**	**7,982**

Results by player show similar trends to the totals, but vary significantly in relative magnitude between the three countries, Mexico (Table 5.4), Canada (Table 5.5) and the U.S. (Table 5.6). The outcomes in these tables are again shown relative to the mean over all strategies and temperature scenarios. Mexico (Table 5.3) gains the most from free-riding compared to the other players, who see greater losses to NPV from non-cooperation (Table 5.4, 5.5). Climate scenarios modify each player's optimal strategy, where Mexico or Canada benefit the most from non-cooperation when sardine distribution shifts into their water in the warming and cooling scenarios, respectively (Table 5.3, 5.4). Both of these strategies lower U.S. (Table 5.5) and overall payoffs, yet full cooperation results in the best payoffs in terms of sardine and total discounted value.

Table 5.4. Mean performance of management strategies for Mexico relative to mean over all strategies and climate scenarios (n=1000). Values are in 2014 USD millions.

Mexico relative to all strategies	**Annual landed value (USD millions /year)**			**Total discounted value (USD millions)**			**Net present Value**
Strategy	**Sardine**	**Others**	**Total**	**Sardine**	**Others**	**Total**	**(USD millions)**
Full Cooperation	1.1	0	1.1	13	0.1	13	0.1
Mexico Free Rider	2.5	0	2.5	72	0	72	0
Canada Free Rider	0.7	0	0.7	5	0	5	0
Non-Cooperative (Pragmatic)	0.6	-0.01	0.6	31	-0.1	31	-0.1
Non-Cooperative (Windfall)	-2.3	-0.01	-2.4	-58	-0.2	-58	-0.2
No Sardine Fishing	-2.5	0.01	-2.5	-63	0.2	-63	0.3
Absolute mean (All strategies)	**2.5**	**57**	**59.5**	**63**	**1,282**	**1,345**	**1,042**

Table 5.5. Mean performance of cooperative management strategies for Canada relative to mean over all strategies and climate scenarios (n=1000). Values are in 2014 USD millions.

Canada relative to all strategies	**Annual landed value (USD millions /year)**			**Total discounted value (USD millions)**			**Net present Value**
Strategy	**Sardine**	**Others**	**Total**	**Sardine**	**Others**	**Total**	**(USD millions)**
Full Cooperation	1.4	0.8	2.2	19	18	38	3.1
Mexico Free Rider	0.9	-0.2	0.7	9	-3.7	5.4	-0.6
Canada Free Rider	2.4	-0.3	2.1	68	-5.2	63	-1
Non-Cooperative (Pragmatic)	0.2	-1.5	-1.3	21	-32	-11	-5.6
Non-Cooperative (Windfall)	-2.4	-2.2	-4.6	-57	-49	-106	-7.8
No Sardine Fishing	-2.5	3.3	0.8	-60	-71	10	12
Absolute mean (All strategies)	**2.5**	**147**	**150**	**60**	**3,311**	**3,371**	**868**

Table 5.6. Mean performance of cooperative management strategies for the U.S. relative to mean over all strategies and climate scenarios (n=1000). Values are in 2014 USD millions.

U.S. relative to all strategies	**Average landed value (USD millions /year)**			**Total discounted value (USD millions)**			**Net present Value**
Strategy	**Sardine**	**Others**	**Total**	**Sardine**	**Others**	**Total**	**(USD millions)**
Full Cooperation	2.6	0.1	2.7	56	2.9	59	2.1
Mexico Free Rider	0.4	0	0.4	5.6	-0.5	5.1	-0.4
Canada Free Rider	0.3	0	0.3	3.4	-0.8	2.6	-0.7
Non-Cooperative (Pragmatic)	-1.1	-0.2	-1.3	-28	-4.9	-33	-3.9
Non-Cooperative (Windfall)	1.5	-0.3	1.2	72	-7.5	65	-5.6
No Sardine Fishing	-3.7	0.5	-3.2	-109	10.8	-99	8.4
Absolute mean (All strategies)	**3.7**	**599**	**603**	**-109**	**13,480**	**13,589**	**6,071**

5.4 Discussion

Results suggest that cooperative fishing strategies outperform others over a range of temperature scenarios when goals incorporate ecosystem-wide economic value. Fisheries along large marine ecosystems like the California Current rarely focus on a single species (e.g., Table 5.1). Management should therefore incorporate species interactions (at the very least) into policies for sustainable fishing. The key influence of forage fishes on overall production has been identified for various marine ecosystems (Shannon 2000; Hannesson & Herrick Jr 2010; Pikitch et al. 2012; Okey et al. 2014). There is some evidence of a relationship between sardine abundance and that of other species in the California Current, at least as reflected by reported catches (Fig. 5.1). This is clearly not necessarily causal, thought in the context of this analysis a manager might nonetheless be able to make some inference on expected overall ecosystem productivity using sardine as an indicator species.

The analyses using ecosystem models representing the California Current lend support for a net positive effect of sardine abundance (acting as an independent variable) on the abundance of other species and their subsequent landed values (Fig. 5.2). If this effect were quite pronounced, the optimal solution would be to leave sardine unfished so that other fisheries would benefit, thereby increasing total value (Sumaila 1997b). However, the magnitude of the relationships found in this study are not enough to offset losses from foregone sardine catch, so that annual total landed value is maximized at a fishing rate above zero, just below F_{MSY} (Fig. 5.3, Table 5.2, Table 5.3).

Alternative assumptions on initial biomass, and therefore estimated maximum sustainable yield (MSY) would change the magnitude of sardine benefits relative to those of other fisheries. For example, MSY using the assumptions in this study is between 70,000-100,000 tons (depending on the climate scenario), compared to the latest reported catch of 300,000 tons (Table 5.1). Future work could incorporate estimates of sardine fishing rates by country to better approximate current stock status, though this was beyond the scope of this study. In any case, overfishing sardine has the worst outcomes, as sardine catch but also biomass decrease, with negative effect on other species (Figs. 5.3, 5.4).

This conclusion holds for total landed value, but incorporating costs of fishing can change perceptions on optimal strategy. The net present value by country was calculated for all species based on available data on fishing costs per ton, which reports a zero per cent profit margin for sardine fisheries (Lam et al. 2011). This plays a large part in the result that, once fishing costs have been accounted for, the net present value for all fisheries is indeed maximized when sardine is left unfished (Table 5.3). However, the difference between this first-best strategy is much lower in Mexico (Table 5.4) than in the US (Table 5.6) or Canada (Table 5.5), thus modifying each individual player's incentives. For example, in terms of total landed value Mexico gains the most from free-riding, with little effect on NPV (Table 5.3), compared to the other players, that see greater losses to NPV from non-cooperation (Table 5.4, 5.5). Notably, the second-best strategy for any player's NPV remains full cooperation.

Alternative temperature (as a proxy for climate) regimes are a central component of the model used here, and can modify the performance of each game structure. It has been previously found that in this ecosystem, the incentives for the two geographically-extreme players, Canada and Mexico, result in a lack of stable cooperative structures given the natural shifts in sardine distribution (Ishimura et al. 2013a, 2013b), but this study extends the analysis to evaluate performance both for sardine and total landed values (Fig. 4). As expected from general trends (Figs. 5.2, 5.3) there is a tradeoff between catching sardine and allowing it to be consumed by other species that are subsequently fished. This tradeoff is not strong enough to offset losses to sardine fisheries if this species is left unfished (F0 in Fig. 5.4), so players would have to find strategies that allow for some sardine fishing.

Cooperative strategies even with at least two players consistently outperform other structures (Co, M, and C in Fig. 5.4), which lends support for potential side payments as a strategy to bring one player into full cooperation (Munro 1979). Side payments are a form of profit-sharing useful when the total payoffs from cooperative strategies are high, but individual payoffs for a given participant are low. So, a portion of total payoffs are re-distributed to incentivize those participants and facilitate cooperation. This strategy has been employed in a range of settings, from fisheries to water use, and could indeed be applied to the California Current case even with a single-species approach (Bailey et al. 2013). However, the wider analysis used here contributes to the calculation of required compensation, as a participant could argue that losses in terms of foregone sardine catch (for example if another player were asked to decrease their fishing rate in favor of the cooperative policy) should be tempered by the potential gains for other fisheries, thus reducing the payment amount. To my knowledge, this type of argument has thus far not been accounted for in transboundary fishing strategy negotiations.

This study highlights the complexity of management choices, yet only three players, six game strategies, and seven metrics (annual and total discounted value of sardine, others and total, and NPV) are evaluated. Transboundary, straddling or high-seas stocks often involve many more players (Munro 1979; Sumaila et al. 2007; Hannesson 2011a) and strategies are very rarely held over time as they have been modelled here. This is made significantly more complicated by the inclusion of ecosystem-level complexity, which could be expanded to multiple species and even non-market values (Daily et al. 2009). When analyzing results with a view to modifying policy, players should consider not just the estimated performance of the best strategies (Table 5.3), but also their benefits relative to current strategies (Table 5.2) and the potential costs of moving away from them. The results from this study suggest that incorporating supporting service values can indeed change optimal policy, but this is also true of a wider valuation that accounts for fishing costs, which may be a more cost-effective prior analysis to undertake.

Though this is not meant to be an ecosystem modelling exercise, the results presented here evidently hinge on the ecosystem models used, and it is quite possible that other model configurations would perhaps not change trends (that would signal markedly different model representations), but certainly magnitude. The source data used to inform the models used here can significantly influence their final structure in terms of, for example, determining diet compositions or the level of aggregation and groups included. Furthermore, because the regressions (Eq. 9) were performed using equilibrium biomass in Ecosim, our methods assume that species will respond to sardine abundance instantaneously each year, which clearly may not be the case. More importantly, the lack of explicit incorporation of non-trophic linkages between species is an issue that should be addressed in future applications. Sardine in particular has been known to have marked correlations (whether causal or not) with abundances of other small pelagics in the California Current (Baumgartner et al. 1992; Chavez 2003). Similar relationships have been observed in other pelagic systems (e.g., the Benguela Current; (Shannon et al. 2004)), and it is likely that many more have not yet been identified.

Aside from their limitations, it is increasingly practical to develop theoretical models incorporating ecosystem linkages and components that are useful for managers (Trites et al. 1999; Plagányi 2007; Fulton 2010; Cisneros-Montemayor et al. 2012). In the case of the California Current, incorporating ecosystem components adds to arguments for sustainable single-stock harvest of sardine and changes optimal cooperative fishing strategies. As we develop these more intricate models, the true complexities of the real world are even more evident. Given the limitations of resources for research and model development, the question of how useful a model is for management is becoming more important than how complex we can make it.

6. Conclusion

Managers increasingly must decide what data to use to make decisions. This extends to the allocation of their own research and management resources in order to collect the most useful information and design and enforce the most useful policies. Models that represent a given management context are a powerful tool to aid this decision-making, and yet they have been underutilized in most of the world. These models can be quite complex, incorporating our best current understanding of ecosystem and socio-economic linkages and structure. Often, however, the simple exercise of recognizing uncertainty and allowing for alternative hypotheses on the state of natural resources (and the industries that depend on them) can provide managers and stakeholders a broader vision with which to make decisions.

'Improvements to information' can mean different things (and have different implications) depending on the context in which they are made. As seen for fisheries in Chapter 2, in some cases this entails challenging our most basic assumptions about catch statistics. Note that, even as we have barely entered into a formal assessment of information value, there is a necessary paradigm shift towards viewing statistics, such as fisheries catch, as an 'assumption' rather than the unalterable truth. Of course, as information-gathering improves, the uncertainty around many such assumptions decreases (almost by definition). Nevertheless, for much of the world, these improvements have only just begun.

In the case of Mexico, a systematic re-estimation of fisheries catch using relatively simple methods suggests that total catch over the last fifty years (1950-2010) was actually twice that of official reports. Aside from corrections to magnitude, more worrying are the emergence of previously-unseen trends. Total catch in Mexico is stable, but the data estimated here shows that benthopelagic fishes (snappers, groupers, etc.) have significantly declined over the last decades, supporting anecdotes from almost any fisher along the coast. Furthermore, discarding of fish is likely more significant than previously thought, a serious issue in a country (like many others in the developing world) with regional food shortages.

Mexico in particular has a long and sound history of fisheries research, with interesting cases of management design despite the challenges of enforcing policies along a very large and at-times ungoverned coastline. A large number of trained individuals and strong research institutions could ideally take these results and improve on the estimation methods to produce catch statistics that are more informative to the models already being used. The fact that this had not been done already again speaks to the necessity of shifting paradigms and actively challenging and improving the current available data (including the estimations in this thesis).

One way of pushing for improvements is by placing a cost on the status quo, which is precisely one of the results of applying a formal value of information method. Given alternative hypotheses on the state of a production system and a range of management options with assumptions and uncertainty around their effectiveness, we can estimate where the best outcomes emerge. For Mexico, given the official and re-estimated catch, results suggest that an additional US$100 million investment in monitoring catch would net over US$700million in economic impacts to the country (Chapter 3). This analysis uses economic impacts to highlight implications of inadequate monitoring, but other metrics could include net present value, employment, consumer surplus, or wider welfare analyses that challenge existing natural resource investment strategies. Mexico in particular already confers US$200 million in capacity-enhancing subsidies that have been proved to be negative for resource management, so the issue in this case is one of restructured investments rather than lacking funds. Revealing these costs of the status quo provides more weight to calls for reform, or at the least should force agencies to provide evidence that their decision-making is informed by data and is in the best interest of the public.

When much data is given, much is expected. As data gathering continues and improves, and perhaps more importantly, becomes more widely accessible, we are able to develop much more complex models that incorporate wider information. A quintessential example are ecosystem models, which were previously thought to be almost unthinkably complicated, and have now become a powerful and accessible tool for many situations. As we continue to move into representing these complex systems, an interesting question arises regarding the *management*-effectiveness of departing from simpler models. In the context of optimally allocating management resources, is being able to build a complex model a reason for building and using it? My answer would be, it depends what the management question being asked is.

One key area where more explicit representations of an ecosystem are highly useful is in evaluating economic benefits beyond the landed value of a single stock, i.e., ecosystem services. In Chapter 4, a relatively simple ecosystem model (this oxymoron speaks to the shifting baselines of research techniques) is used to highlight the fact that the supporting value of some fish species as food for higher trophic level (and market value) species far outweighs their own landed value. This has clear policy implications, especially when managers are tasked with considering the total value of a regions' fisheries and other industries (e.g., sport fishing or ecotourism).

However, the magnitude of the tradeoff may or may not warrant drastic changes to active policy. The model developed in Chapter 5 is a relatively complicated exercise. A climate model feeds into a species distribution model, into a population dynamics model, into a game-theoretic fishing strategy model, into a set of ecosystem models, and back. Its development was very interesting and useful from an academic perspective (which is to say I learned a lot about modeling techniques), but its results are far more interesting in the context of valuing information itself. The results confirm previous findings, where the players on the edges of the stock's distribution only have an incentive to conserve the stock in extreme climate (either warming or cooling) scenarios, but goes further by asking how their single-stock-informed actions affect the other fisheries that take place in their ecosystems. The results suggest that managing a single stock (in this case a sardine species) "optimally" can indeed have negative impacts on the other species in the ecosystem, but that total fishery landed value would not be helped by simply shutting down the sardine fishery. Ultimately, a cooperative solution where the sardine stock is fished sustainably and its biomass is maintained at a high level in order to allow for other species to feed has the best outcomes over a range of potential climate trends. In terms of cooperative strategy, incorporating both multi-species values and fishing costs resulted in a reversal of the optimal fishing policy.

This work, as stated in Chapter 1, set out to address a handful of research questions, and an overarching problem: Given the expected economic benefits from competing sets of management assumptions and frameworks, is it worth it to invest in finding out which set is correct? The answer is that it is *always* worth it to honestly recognize what the competing assumptions are, and what the outcomes would be under alternative scenarios. Our choices should follow from there. The alternative, carrying on as usual and ignoring or hiding uncertainty, is a disservice to the public and to the natural ecosystems we aim to manage.

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Appendices

Appendix A. Methods and sources for estimation of unreported catch in Mexico, by fishery:

Pacific Ocean

Table A.1. Methods and sources for estimation of unreported catches in Mexico, by fishery: Pacific Ocean. Names are as appear in FAO catch database, except for multi-fishery reconstruction 'Finfish'; 'nei'= not elsewhere included.

Name	Method
Abalones nei	Species breakdown between *Haliotis fulgens* and *H. corrugata* based on current catch composition (~79% and 21%, respectively) (Sierra-Rodríguez et al. 2006). Illegal catch is therefore assumed to be 15%, lower than overall rate applied to artisanal fisheries (see 'Finfish').
Albacore	Not modified. Note: There is no mention of albacore (*Thunnus alalunga*) in the National Fisheries Charts.
Amberjacks nei	Interpolated catch. Illegal and unreported catch: see 'Finfish'. Gillnets targeting jacks are set on fish run routes; bycatch of other fishes is 31% of catch (DOF 2004).
Aquatic invertebrates nei	See 'Marine crabs nei'.
Ark clams nei	Interpolated catch (1950-1963, 1978-1984).
Barracudas nei	Interpolated catch (1950-1953, 1958, 1963, 1965-1972). Unreported and illegal catch: see 'Finfish'.
Bigeye scad	Not modified. This is an Atlantic Ocean fish, but appears in Pacific Ocean catch.
Bigeye tuna	Not modified.
Black drum	Not modified. This is an Atlantic Ocean fish, but appears in Pacific Ocean catch.
Black skipjack	There is a small, largely unmonitored artisanal fishery operating in the state of Oaxaca, with likely landings of around 10% of reported catch (pers. obs.). Catch statistics drop abruptly in 2006 compared to more recent years, and are missing before 2005.

Table A.1. Methods and sources for estimation of unreported catches in Mexico, by fishery: Pacific Ocean. Names are as appear in FAO catch database, except for multi-fishery reconstruction 'Finfish'; 'nei'= not elsewhere included.

Name	Method
Blue shark	Catch statistics were missing before 2005, so they were interpolated backward and subtracted from the 'Sharks, rays, nei' category. Unreported and illegal catch: see 'Finfish'.
Blue shrimp	See 'Penaeus shrimps nei'.
Bobo mullet	There is no specific mention of the bobo mullet (*Joturus pichardi*) in the National Fisheries Charts, though many types of mullet are included. Unreported and illegal catch: see 'Finfish'. Bycatch in the mullet fishery is high at 70% of target and consists of miscellaneous fish with 32 species listed (DOF 2004).
Brown seaweeds	Not modified. Note: catch statistics drop abruptly in late 2000s.
California pilchard	Some unreported catch may be taken directly to tuna pens near Ensenada. Unreported catch estimated as 20% of reported landings from Northern Baja California (Eva Cotero, Instituto Nacional de Pesca, pers. comm.).
Californian anchovy	Not modified.
Cannonball jellyfish	New series added from catch reported in DOF 2010. A conservative unreported catch of 10% relative to reported catch was added to this incipient fishery.
Carangids nei	Interpolated catch. Unreported and illegal catch: see 'Finfish'.
Chub mackerel	Not modified.
Clupeoids nei	Not modified.
Common dolphinfish	Unreported recreational landings and discard mortality estimated in (Cisneros-Montemayor et al. 2012). Illegal artisanal catch estimated as 20 times that of total recreational catch.
Croakers nei	Interpolated catch. Unreported and illegal catch: see 'Finfish'.
Croakers, drums nei	Interpolated catch. Unreported and illegal catch: see 'Finfish'.

Table A.1. Methods and sources for estimation of unreported catches in Mexico, by fishery: Pacific Ocean. Names are as appear in FAO catch database, except for multi-fishery reconstruction 'Finfish'; 'nei'= not elsewhere included.

Name	Method
Cupped oysters nei	Interpolated catch. Unreported and illegal catch: see 'Finfish'.
Demersal percomorphs nei	Unreported and illegal catch: see 'Finfish'.
Eastern Pacific bonito	Unreported and illegal catch: see 'Finfish'. In addition, unreported sport catch is assumed to be 2% of reported landings.
Echinoderms	See 'Sea urchins nei'.
Flatfishes nei	Interpolated catch (1950-1954). Unreported and illegal catch: see 'Finfish'.
Flathead grey mullet	See 'Mullets nei'.
Gadiformes nei	Unreported and illegal catch: see 'Finfish'.
Gastropods nei	This group was pooled with the 'Stromboid conchs nei' as these evidently correspond to the same time series. A split into two snail species, *Astraea undosa* and *Phyllonotus erhythostoma* was then made using reported catch composition (DOF 2004). Unreported and illegal catch rates assumed to be equal to 'Finfish' category.
Groupers nei	Interpolated catch (1950-1954, 1958-1963, 1984-2010). Unreported and illegal catch: see 'Finfish'.
Groupers, seabasses nei	Interpolated catch (1950-1964, 1979-1983). Unreported and illegal catch: see 'Finfish'.
Grunts, sweetlips nei	Unreported and illegal catch: see 'Finfish'.
Hairtails, scabbardfishes nei	Unreported and illegal catch: see 'Finfish'.
Hammerhead sharks, etc. nei	Catch statistics were missing before 2005, so they were interpolated backward and subtracted from the 'Sharks, rays, nei' category. Unreported and illegal catch: see 'Finfish'.
Jacks, crevalles nei	Unreported, illegal catch and bycatch: see 'Amberjacks nei'.

Table A.1. Methods and sources for estimation of unreported catches in Mexico, by fishery: Pacific Ocean. Names are as appear in FAO catch database, except for multi-fishery reconstruction 'Finfish'; 'nei'= not elsewhere included.

Name	Method
Jumbo flying squid	A conservative unreported 10% relative to reported catch was added to reflect artisanal catch.
Marine crabs nei	Three vague but related categories, 'Aquatic invertebrates nei', 'Marine crabs nei' and 'Marine crustaceans nei' were joined into a single category and subsequently used as a baseline to split out species groups for which data was found. When catch per year was available for a species, this was used in a new catch series for that species and subtracted from the baseline 'Invertebrates' group. The resulting species were *Cancer anthonyi*, *C. antennarius*, *C. productus*, *C. gracilis*, *Callinectes bellicosus*, *C. arcuatus* and *C. toxotes* (Molina-Ocampo et al. 2006; DOF 2010). Unreported and illegal catch rates were assumed to be equal to the 'Finfish' category.
Marine crustaceans nei	See 'Marine crabs nei'.
Marine fishes nei	Catch of other species reported within this category was split out (subtracted) whenever possible; this is noted in the corresponding species' entries. Unreported and illegal catch: see 'Finfish'.
Marine shells nei	Interpolated catch (1950-1957, 1960-1963, 1990-1994). A conservative unreported catch equal to 10% of reported catch was added to this vague category.
Marlins, sailfishes, etc. nei	Unreported recreational landings and discard mortality estimated based on (Cisneros-Montemayor et al. 2012). Bycatch in industrial longline fishery: see 'Requiem sharks'.
Mexican barred snapper	Unreported and illegal catch: see 'Snappers, jobfishes nei'.
Milkfish	Unreported and illegal catch: see 'Finfish'.
Mojarras, etc. nei	Unreported and illegal catch: see 'Finfish'.

Table A.1. Methods and sources for estimation of unreported catches in Mexico, by fishery: Pacific Ocean. Names are as appear in FAO catch database, except for multi-fishery reconstruction 'Finfish'; 'nei'= not elsewhere included.

Name	Method
Mullets nei	The fishery for mullets is under a specific management plan with full seasonal closures (NOM-016-PESC-1994), so better monitored and more difficult to 'launder' illegal catch that does take place. Thus, current unreported statistics relative to reported catch are assumed to be 10% from legal and 20% from illegal catch, less than that assumed for the 'Finfish' in general.
North Pacific hake	Unreported and illegal catch: see 'Finfish'.
Northern red snapper	Unreported and illegal catch: see 'Snappers, jobfishes nei'.
Ocean whitefish	Catch reported in FAO (1998-2010) was split among three species (*Caulolatilus princeps*, *C. affinis*, *C. hubbsi*), with catch of *C. princeps* making up 70% of total, and 15% each for others (DOF 2010). Total catch from 1990-1997 was available from official statistics (DOF 2010) and the same species split was used. Catch from 1950-1989 was interpolated based on 1990 catch (assuming 1950 catch was equal to 50% of 1990) and subtracted from 'Marine fish nei'. Unreported and illegal catch: see 'Finfish'.
Octopuses, etc. nei	Unreported and illegal catch: see 'Finfish'.
Opalescent inshore squid	Not modified.
Pacific anchoveta	Not modified.
Pacific angelshark	Catch statistics were missing before 2005, so they were interpolated backward and subtracted from the 'Sharks, rays, nei' category. Unreported and illegal catch: see 'Finfish'.
Pacific bluefin tuna	Not modified.

Table A.1. Methods and sources for estimation of unreported catches in Mexico, by fishery: Pacific Ocean. Names are as appear in FAO catch database, except for multi-fishery reconstruction 'Finfish'; 'nei'= not elsewhere included.

Name	Method
Pacific calico scallop	This fishery is under a specific management plan with full seasonal closures (NOM-004-PESC-1993), so better monitored and more difficult to 'launder' illegal catch that does take place. Thus, current unreported statistics relative to reported catch are assumed to be 10% from legal and 20% from illegal catch. Interpolated catch (1950-1983).
Pacific flatiron herring	Not modified.
Pacific jack mackerel	Unreported and illegal catch: see 'Finfish'.
Pacific piquitinga	Unreported and illegal catch: see 'Finfish'.
Pacific red snapper	Unreported and illegal catch: see 'Finfish'.
Pacific sierra	Interpolated catch (1950-1963). Unreported and illegal catch: see 'Finfish'.
Pacific thread herring	Not modified.
Paralabrax spp	Unreported and illegal catch: see 'Finfish'.

Table A.1. Methods and sources for estimation of unreported catches in Mexico, by fishery: Pacific Ocean. Names are as appear in FAO catch database, except for multi-fishery reconstruction 'Finfish'; 'nei'= not elsewhere included.

Name	Method
Penaeus shrimps nei	Species catch estimated for 1950-2004 based on '*Penaeus* shrimps' series and 1985-2000 catch composition (DOF 2004). Industrial and artisanal catch split from total based on historical number of vessels by sector (1970-2007, (CONAPESCA 2014); other years extrapolated) and current catch ratio (DOF 2004). Unreported and illegal catch relative to reported catch was estimated to be 10% and 20%, respectively, based on (Cisneros-Mata et al. 2012). Shrimp to bycatch ratio for industrial sector at 1:10 (Vázquez et al. 2004), and for artisanal sector as 1:3 for legal gears and 1:10.5 for illegal gear (Amezcua et al. 2006). Proportion of fishes, crustaceans and mollusks in bycatch at 70, 28 and 2%, respectively. Bycatch discard rate at 84% for fish and 100% for other groups (Bojórquez et al. 1999). Species breakdown of fishes in industrial bycatch based on (Bojórquez et al. 1999). Species breakdown of crustaceans in industrial bycatch based on (Rosales 1976; Paul & Hendrickx 1980).
Pompanos nei	Unreported and illegal catch: see 'Finfish'.
Rays, stingrays, mantas nei	Catch from this aggregate group was split into species groups based on official reported catch composition (Batoidea, *Dasyatis* spp., Rhinobatidae, *Rhinobatos productus*, *Rhinoptera steindachneri*) (DOF 2004). Unreported and illegal catch: see 'Finfish'.
Red seaweeds	Not modified.
Red-eye round herring	Not modified.

Table A.1. Methods and sources for estimation of unreported catches in Mexico, by fishery: Pacific Ocean. Names are as appear in FAO catch database, except for multi-fishery reconstruction 'Finfish'; 'nei'= not elsewhere included.

Name	Method
Requiem sharks nei	Total reported catch was split into three species groups, *Carcharhinus falciformis*, *C. limbatus* and *Carcharhinus* spp. (DOF 2004). As these catches correspond to the oceanic (longline) fishery, bycatch rates were applied based on reported catch composition (DOF 2004). As the catch statistics of the main bycatch, billfish, do not seem to reflect their bycatch in this fishery, and are reserved for sport fishing up to 50 nm from the shore, we assume that all catch is kept but is unreported. Reported bycatch species and their contribution to total catch are billfishes (40.8%), dolphinfish (4.6%), yellowfin tuna (3.5%) and other fishes (3%).
Sea catfishes nei	Unreported and illegal catch: see 'Finfish'.
Sea cucumbers nei	Unreported and illegal catch: see 'Finfish'.
Sea mussels nei	Though there is some unreported catch, mussels are not a particularly valuable fishery, so we assume only 10% unreported catch relative to reported.

Table A.1. Methods and sources for estimation of unreported catches in Mexico, by fishery: Pacific Ocean. Names are as appear in FAO catch database, except for multi-fishery reconstruction 'Finfish'; 'nei'= not elsewhere included.

Name	Method
Sea urchins nei	The category 'Echinoderms' includes sea urchins from 1990-2004; from 2005-2010, sea urchins are split out but not reported by species. However, catch by species for red (*Strongylocentrotus franciscanus*) and purple (*S. pupuratus*) sea urchin is reported in official documents from 1972-2008 (Palleiro-Nayar et al. 2006; DOF 2010). Species was split out using this information, and assuming that the species ratio from 2009-2010 was the same as 2008. There is an unknown component in the 'Equinoderms' catch series that does not correspond to starfish (this fishery does not appear in FAO data and was included independently; see 'Starfish'). For the years when sea urchins were included in the 'Equinoderms' catch series, the unknown component was maintained by linearly interpolating 'Equinoderm' catch from 1989-2006. This split into species did not affect total reported catch in any way. The resulting catch series are 'Equinoderms', and separate series for each sea urchin species, one for when each was reported as 'Equinoderms', one for when they were reported as 'Sea urchins', and one for catch of red sea urchin unreported in FAO data but appearing in official statistics. This fishery is under a specific management plan with full seasonal closures (NOM-009-PESC-1993), so better monitored and more difficult to 'launder' illegal catch that does take place. Thus, current unreported statistics relative to reported catch are assumed to be 10% from legal and 20% from illegal catch.
Sharks, rays, skates, etc. nei	Interpolated catches of available shark species were subtracted from this aggregate group (see 'Blue shark', 'Hammerhead sharks, etc. nei', 'Pacific angelshark'). Unreported and illegal catch: see 'Finfish'.
Shortjaw leatherjacket	Unreported and illegal catch: see 'Finfish'.
Silversides(=Sand smelts) nei	Unreported and illegal catch: see 'Finfish'.
Skipjack tuna	Not modified.

Table A.1. Methods and sources for estimation of unreported catches in Mexico, by fishery: Pacific Ocean. Names are as appear in FAO catch database, except for multi-fishery reconstruction 'Finfish'; 'nei'= not elsewhere included.

Name	Method
Snappers, jobfishes nei	Category split into eighteen different species based on listing in (DOF 2010). There was no information on the relative abundance (either point estimate or through time) of particular species, so total catch was apportioned equally among them. Directed snapper fisheries most commonly use hook and line gear, so unwanted bycatch is generally lower than the broad finfish fisheries, which usually use nets of different types. Unreported and illegal catch rates were assumed to be equal to those used for 'Finfish' (15% and 22%, respectively), but we conservatively assume that no discarding takes place. Interpolated catch (1950-1954, 1962).
Snooks (=Robalos) nei	Species breakdown based on (DOF 2010). *Centropomus vidris* is cited as most abundant in Chiapas and Nayarit fishery (which makes up ~70% of total catch), and *C. medius* in the rest of fishing areas. These species were assumed to contribute 70% of catch in their respective areas, resulting in ratio of 49% and 21% for *C. vidris* and *C. medius*, respectively. Other catch was apportioned equally among remaining species, *C. nigriscens*, *C. armatus*, *C. robalito* (10% each). Unreported and illegal catch: see 'Finfish'.
Spotted rose snapper	Unreported and illegal catch: see 'Snappers, jobfishes nei'.
Spotted weakfish	Not modified. This fish is from the Atlantic Ocean.
Starfish	New series added from catch reported in (DOF 2010). A conservative unreported catch of 10% relative to reported catch was added.
Stromboid conchs nei	See 'Gastropods nei'.
Swordfish	Interpolated catch (1950-1953, 1958-1963).

Table A.1. Methods and sources for estimation of unreported catches in Mexico, by fishery: Pacific Ocean. Names are as appear in FAO catch database, except for multi-fishery reconstruction 'Finfish'; 'nei'= not elsewhere included.

Name	Method
Totoaba	Illegal catch of adults after 1972 moratorium estimated as 161.7 t per year (Cisneros-Mata et al. 1995a); after 1993 enactment of the Upper Gulf Reserve (DOF 1993) up to 2010, catch is assumed to decline from 50% to 25%. Unreported bycatch in shrimp trawls estimated based on number of shrimp trawlers per year and the estimated catch of juveniles in 1988 (Cisneros-Mata et al. 1995a), and assumed to have ended in 1993 after enforcement of the Reserve. Interpolated catch (1950-1954). Unreported catch in directed fishery: see 'Finfish'.
Triggerfishes, durgons nei	Interpolated catch (2003-2004). Unreported and illegal catch: see 'Finfish'.
Tropical spiny lobsters nei	Interpolated catch (1950-1953). Unreported, illegal and incidental catch outside of Baja California (using nets) estimated based on catch outside Baja California (7.6%) (CONAPESCA 2014) and catch rates for that gear type (Pérez-González et al. 2006). Unreported catch (of non-marketed lobster) and bycatch in trap fishery estimated based on catch rates and species breakdown for Baja California (Shester 2008). This fishery is under a specific management plan with full seasonal closures (NOM-009-PESC-1993), so better monitored and more difficult to 'launder' illegal catch that does take place. Thus, current unreported statistics relative to reported catch are assumed to be 10% from legal and 20% from illegal catch.
Venus clams nei	This aggregate category was broken down into seven species groups (*Anadara tuberculosa*, *Chione californiensis*, *Lyropecten subnodosus*, *Megapitaria squalida*, *Pecten vogdesi*, *Pinna rugosa* and Veneridae) based on catch composition reported in official documents (DOF 2004, 2010). Unreported and illegal catch rates were assumed to be equal to the 'Finfish' category.
Weakfishes nei	Unreported and illegal catch: see 'Finfish'.
White mullet	Unreported and illegal catch: see 'Finfish'.
Whiteleg shrimp	See 'Penaeus shrimps nei'.

Table A.1. Methods and sources for estimation of unreported catches in Mexico, by fishery: Pacific Ocean. Names are as appear in FAO catch database, except for multi-fishery reconstruction 'Finfish'; 'nei'= not elsewhere included.

Name	Method
Yellow snapper	Interpolated catch (1950-1957). Unreported and illegal catch: see 'Snappers, jobfishes nei'.
Yellowfin tuna	Not modified.
Yellowleg shrimp	See 'Penaeus shrimps nei'.
Finfish	The *escama*, or finfish, fishery in Mexico is both a common name for miscellaneous fishes and an official management category. It is essentially a permit catch anything other than a few specific target species (e.g., shrimp, lobster, crab, tuna, etc.). Monitoring and reporting is consequently vague, so all catch series under the finfish category (and/or those for which individual data was not found) were estimated together under the conservative assumption that 15% of legal catch is not captured in official statistics, having decreased from around 40% since the 1950s due to some improvement in reporting schemes (María Espinosa-Romero, pers. comm.). A survey of experts reported that a further 45% is captured illegally (Cisneros-Mata et al. 2012), though around half of this catch is subsequently bought by processors and reported with legal catches (second author's pers. obs.). We assume that illegal catch has increased (by a factor of 2) over the study period as more regulations are enacted (i.e., illegal catch rate in 1950 is assumed to be half of current, increasing linearly). Thus, unreported (including legal and illegal) catch is currently a further 37% of reported landings. Where applicable, catch is interpolated before estimating unreported and illegal catches. As finfish is by nature a multi-species fishery, it is assumed that bycatch of species that are not the main target was kept. However, many individuals are discarded due to small size or low value; bycatch discards for the finfish fishery as a whole were estimated based on a 34.3% discard rate (relative to total catch) reported in (Shester & Micheli 2011).

Appendix B. Methods and sources for estimation of unreported catch in Mexico, by fishery:

Atlantic Ocean

Table B.1. Methods and sources for estimation of unreported catches in Mexico, by fishery: Atlantic Ocean. Names are as appear in FAO catch database, except for multi-fishery reconstruction 'Finfish'; 'nei'='not elsewhere included'.

Name	Method
Amberjacks nei	Interpolated catch (1950-1963, 1979-1989). Illegal and unreported catch: see 'Finfish'. Gillnets targeting jacks are set on fish run routes; bycatch of other fishes is 31% of catch (DOF 2004).
American cupped oyster	Fishery is reported as run by well-managed cooperatives (DOF 2004), so only a conservative illegal catch of 5% relative to reported catch was added. Interpolated catch (1950-1952).
American eel	Unreported and illegal catch: see 'Finfish'.
Anchovies, etc. nei	Interpolated catch (1950-1953, 1958-1963).
Aquatic invertebrates nei	Unreported and illegal catch: see 'Finfish'.
Atlantic bluefin tuna	Not modified.
Atlantic bonito	Not modified.
Atlantic sailfish	Commercial fishery catch was not modified, but 10% of catch relative to reported commercial landings was added corresponding to the sport fishery in the Gulf of Mexico.
Atlantic seabob	Not modified.
Atlantic sharpnose shark	Unreported and illegal catch: see 'Finfish'.
Atlantic Spanish mackerel	Unreported and illegal catch: see 'Finfish'.
Barracudas nei	Unreported and illegal catch: see 'Finfish'.
Barred grunt	Unreported and illegal catch: see 'Finfish'.
Bigeye thresher	Not modified.
Bigeye tuna	Not modified.

Table B.1. Methods and sources for estimation of unreported catches in Mexico, by fishery: Atlantic Ocean. Names are as appear in FAO catch database, except for multi-fishery reconstruction 'Finfish'; 'nei'='not elsewhere included'.

Name	Method
Black drum	Unreported and illegal catch: see 'Finfish'.
Black stone crab	Interpolated catch (1950-1963). Unreported and illegal catch: see 'Finfish'.
Blackfin tuna	Not modified.
Blue crab	Interpolated catch (1950-1963). Unreported and illegal catch: see 'Finfish'.
Blue runner	Unreported and illegal catch: see 'Finfish'.
Bobo mullet	Interpolated catch (1950-1963). Unreported and illegal catch: see 'Finfish'.
Brazilian groupers nei	Unreported and illegal catch: see 'Finfish'.
Carangids nei	Interpolated catch (1950-1963, 1979-1989). Unreported and illegal catch: see 'Finfish'.
Caribbean spiny lobster	Interpolated catch (1950-1963). Unreported and illegal catch: see 'Finfish'.
Chub mackerel	Not modified.
Clupeoids nei	Not modified.
Cobia	Unreported and illegal catch: see 'Finfish'.
Common dolphinfish	Unreported and illegal catch: see 'Finfish'.
Common octopus	Unreported and illegal catch: see 'Finfish'.
Common snook	Interpolated catch (1950-1962). Unreported and illegal catch: see 'Finfish'.
Common squids nei	Unreported and illegal catch: see 'Finfish'.
Croakers, drums nei	Interpolated catch (1950-1961). Unreported and illegal catch: see 'Finfish'.

Table B.1. Methods and sources for estimation of unreported catches in Mexico, by fishery: Atlantic Ocean. Names are as appear in FAO catch database, except for multi-fishery reconstruction 'Finfish'; 'nei'='not elsewhere included'.

Name	Method
Cubera snapper	See 'Snappers, jobfishes nei'.
Demersal percomorphs nei	Unreported and illegal catch: see 'Finfish'.
Echinoderms	This is a fairly small dedicated fishery in the Atlantic coast, so we assumed a relatively low rate of unreported catch, of 5%.
Flatfishes nei	Unreported and illegal catch: see 'Finfish'.
Flathead grey mullet	Unreported and illegal catch: see 'Finfish'.
Gastropods nei	Unreported and illegal catch: see 'Finfish'.
Grey snapper	Unreported and illegal catch: see 'Snappers, jobfishes nei'.
Groupers nei	Unreported and illegal catch: see 'Finfish'.
Groupers, seabasses nei	Unreported and illegal catch: see 'Finfish'.
Grunts, sweetlips nei	Unreported and illegal catch: see 'Finfish'.
Gulf kingcroaker	Interpolated catch (1950-1961). Unreported and illegal catch: see 'Finfish'.
Hairtails, scabbardfishes nei	Unreported and illegal catch: see 'Finfish'.
Hammerhead sharks, etc. nei	Unreported and illegal catch: see 'Finfish'.
Hogfish	Unreported and illegal catch: see 'Finfish'.
Jacks, crevalles nei	Unreported, illegal catch and bycatch, see 'Amberjacks nei'.
King mackerel	Unreported and illegal catch: see 'Finfish'.
Ladyfish	Unreported and illegal catch: see 'Finfish'.
Lane snapper	Unreported and illegal catch: see 'Snappers, jobfishes nei'.

Table B.1. Methods and sources for estimation of unreported catches in Mexico, by fishery: Atlantic Ocean. Names are as appear in FAO catch database, except for multi-fishery reconstruction ‘Finfish’; ‘nei’=‘not elsewhere included’.

Name	Method
Marine crabs nei	Unreported and illegal catch: see ‘Finfish’.
Marine fishes nei	Unreported and illegal catch: see ‘Finfish’.
Marine shells nei	Unreported and illegal catch: see ‘Finfish’.
Marlins, sailfishes, etc. nei	Not modified.
Mexican four-eyed octopus	This fishery is the most commercially important in the Mexican Atlantic and is under a specific management plan (NOM-008-PESC-1993). Current unreported statistics relative to reported catch are assumed to be 15% of reported catch.
Milkfish	Unreported and illegal catch: see ‘Finfish’.
Mojarras, etc. nei	Unreported and illegal catch: see ‘Finfish’.
Mullets nei	Unreported and illegal catch: see ‘Finfish’.
Northern brown shrimp	See ‘Penaeus shrimps nei’.
Northern pink shrimp	See ‘Penaeus shrimps nei’.
Northern red snapper	Unreported and illegal catch: see ‘Snappers, jobfishes nei’.
Northern white shrimp	See ‘Penaeus shrimps nei’.
Nurse shark	Unreported and illegal catch: see ‘Finfish’.

Table B.1. Methods and sources for estimation of unreported catches in Mexico, by fishery: Atlantic Ocean. Names are as appear in FAO catch database, except for multi-fishery reconstruction 'Finfish'; 'nei'='not elsewhere included'.

Name	Method
Penaeus shrimps nei	All shrimp catch in the Atlantic was assigned to one of six species based on information from the National Fisheries Chart (DOF 2010). These species are the seabob (*Xiphopenaeus kroyeri*), brown (*Farfantepenaeus aztecus*), white (*Litopenaeus setiferus*), pink (*F. duodarum*), rock (*Syciona brevirostris*) and red (*F. brasiliensis*) shrimp. Catch ratios per species for years without species-specific catch data were calculated based on average for available years. Shrimp to bycatch ratio for industrial sector of 1:3 (Bojórquez et al. 1999) was assumed for all recorded landings. The proportion of fishes, crustaceans and mollusks in bycatch were assumed to be 65, 33 and 2%, respectively, with a bycatch discard rate at 57% for fish and 100% for other groups (Bojórquez et al. 1999).
Pompanos nei	Unreported and illegal catch relative to reported catch was estimated to be 10% and 20%, respectively, based on (Cisneros-Mata et al. 2012).
Porgies	Interpolated catch (1950-1961). Unreported and illegal catch: see 'Finfish'.
Porgies, seabreams nei	Unreported and illegal catch: see 'Finfish'.
Rays, stingrays, mantas nei	Unreported and illegal catch: see 'Finfish'.
Requiem sharks nei	Interpolated catch (1950-1961). Unreported and illegal catch: see 'Finfish'.
Round sardinella	A further 5% of reported catch was added, as this is an occasional bycatch species.
Scaled sardines	A further 5% of reported catch was added, as this is an occasional bycatch species.
Sea catfishes nei	Unreported and illegal catch: see 'Finfish'.
Sea cucumbers nei	Unreported and illegal catch: see 'Finfish'.

Table B.1. Methods and sources for estimation of unreported catches in Mexico, by fishery: Atlantic Ocean. Names are as appear in FAO catch database, except for multi-fishery reconstruction 'Finfish'; 'nei'='not elsewhere included'.

Name	Method
Sharks, rays, skates, etc. nei	Unreported and illegal catch: see 'Finfish'.
Shortfin mako	Unreported and illegal catch: see 'Finfish'.
Skipjack tuna	Not modified.
Snappers, jobfishes nei	Directed snapper fisheries most commonly use hook and line gear, so unwanted bycatch is generally lower than the broad finfish fisheries, which usually use nets of different types. Unreported and illegal catch rates were assumed to be equal to those used for 'Finfish' (15% and 22%, respectively), but we conservatively assume that no discarding takes place.
Snooks(=Robalos) nei	Unreported and illegal catch: see 'Finfish'.
Southern stingray	Unreported and illegal catch: see 'Finfish'.
Spotted weakfish	Interpolated catch (1950-1957, 1962). Unreported and illegal catch: see 'Finfish'.
Stromboid conchs nei	Catch data was split into *Strombus gigas*, *S. costatus*, and Strombidae based on catch composition information (DOF 2010) . In light of the management plan in place for this fishery (NOM-013-PESC-1994), unreported statistics relative to reported catch are conservatively assumed to be 10% from legal and 20% from illegal catch.
Swordfish	Not modified.
Tarpon	Unreported and illegal catch: see 'Finfish'.
Tiger shark	Unreported and illegal catch: see 'Finfish'.
Tilefishes nei	Unreported and illegal catch: see 'Finfish'.
Triggerfishes, durgons nei	Interpolated catch (1950-1963, 2003-2004). Unreported and illegal catch: see 'Finfish'.

Table B.1. Methods and sources for estimation of unreported catches in Mexico, by fishery: Atlantic Ocean. Names are as appear in FAO catch database, except for multi-fishery reconstruction 'Finfish'; 'nei'='not elsewhere included'.

Name	Method
Venus clams nei	Catch data was split equally among the main species in the catch composition, *Rangia cuneata*, *R. flexuosa*, *Polimesoda carolineana*, *Anadara baughmani* and *Mercenaria mercenaria* (DOF 2004). It is reported that lack of monitoring results in significant underreporting, so it was assumed that 50% of all catch is not reported.
Vermilion snapper	Unreported and illegal catch: see 'Snappers, jobfishes nei'.
Weakfishes nei	Unreported and illegal catch: see 'Finfish'.
White grunt	Unreported and illegal catch: see 'Finfish'.
White mullet	Unreported and illegal catch: see 'Finfish'.
Yellowfin tuna	Not modified.
Yellowtail snapper	Unreported and illegal catch: see 'Snappers, jobfishes nei'.

Table B.1. Methods and sources for estimation of unreported catches in Mexico, by fishery: Atlantic Ocean. Names are as appear in FAO catch database, except for multi-fishery reconstruction ‘Finfish’; ‘nei’=‘not elsewhere included’.

Name	Method
Finfish	The ‘escama’, or finfish, fishery in Mexico is both a common name for miscellaneous fishes and an official management category. It is essentially a permit catch anything other than a few specific target species (e.g., shrimp, lobster, crab, tuna, etc.). Monitoring and reporting is consequently vague, so all catch series under the finfish category (and/or those for which individual data was not found) were estimated together under the conservative assumption that 15% of legal catch is not captured in official statistics, having decreased from around 40% since the 1950s due to some improvement in reporting schemes (María Espinosa-Romero, pers. comm.). A survey of experts reported that a further 45% is captured illegally (Cisneros-Mata et al. 2012), though around half of this catch is subsequently bought by processors and reported with legal catches (pers. obs.). We assume that illegal catch has increased (by a factor of 2) over the study period as more regulations are enacted (i.e., illegal catch rate in 1950 is assumed to be half of current, increasing linearly). Thus, unreported (including legal and illegal) catch is currently a further 37% of reported landings. Where applicable, catch is interpolated before estimating unreported and illegal catches. As finfish is by nature a multi-species fishery, it is assumed that bycatch of species that are not the main target was kept. However, many individuals are discarded due to small size or low value; bycatch discards for the finfish fishery as a whole were estimated based on a 34.3% discard rate (relative to total catch) reported in (Shester & Micheli 2011).

Appendix C. Current yearly landed value by country off the California Current.

Table C.1. Current yearly landed value by country off the California Current (CONAPESCA 2013; DFO 2014; NOAA 2014). Ex-vessel price estimated using total catch (metric tons) and value (2014 USD). Average profit margin calculated based on per-gear total fishing costs reported in Lam et al. (2013).

Annual fishery catch and value, California Current					
Group	**Country**	**Catch (mt '000)**	**Landed value (USD millions)**	**Price (USD '000)**	**Profit margin (%)**
Benthopelagic Fish	Canada	20.4	42.6	2.1	0
Cod-likes	Canada	53.3	16.5	0.3	2
Crustaceans	Canada	3.4	22.0	6.5	54
Elasmobranchs	Canada	1.9	0.7	0.4	0
Flatfishes	Canada	13.8	31.9	2.3	10
Mollusks	Canada	1.4	20.3	14.3	91
Other	Canada	0.0	0.0	0.7	0
Other Invertebrates	Canada	3.1	5.9	1.9	33
Pelagic Fish	Canada	12.6	3.0	0.2	0
Salmon	Canada	9.0	38.5	4.3	77
Sardine	Canada	0.0	0.0		0
Small Pelagic Fish	Canada	9.4	10.7	1.1	36
Squids	Canada	0.0	0.0		0
Tunas and Billfish	Canada	1.4	4.7	3.4	26
Benthopelagic Fish	US	13.2	47.9	3.6	42
Cod-likes	US	64.7	22.1	0.3	11
Crustaceans	US	54.4	240.1	4.4	32
Elasmobranchs	US	1.4	1.4	1.0	0
Flatfishes	US	11.9	21.4	1.8	0
Mollusks	US	6.3	132.5	21.0	94
Other	US	0.0	0.0		0
Other Invertebrates	US	6.4	16.1	2.5	49
Pelagic Fish	US	0.9	1.1	1.2	20
Salmon	US	11.0	49.4	4.5	78
Sardine	US	99.9	22.0	0.2	0
Small Pelagic Fish	US	9.6	2.7	0.3	0
Squids	US	97.5	66.4	0.7	0
Tunas and Billfish	US	14.4	50.3	3.5	28
Benthopelagic Fish	Mexico	0.3	0.2	0.5	0
Cod-likes	Mexico	0.0	0.0		0
Crustaceans	Mexico	3.3	26.9	8.1	89
Elasmobranchs	Mexico	4.2	3.1	0.7	30
Flatfishes	Mexico	0.4	0.6	1.6	62

Annual fishery catch and value, California Current					
Group	**Country**	**Catch (mt ‘000)**	**Landed value (USD millions)**	**Price (USD ‘000)**	**Profit margin (%)**
Mollusks	Mexico	7.2	11.4	1.6	77
Other	Mexico	9.0	5.4	0.6	62
Other Invertebrates	Mexico	3.5	13.2	3.8	90
Pelagic Fish	Mexico	2.9	3.5	1.2	76
Salmon	Mexico	0.0	0.0		0
Sardine	Mexico	143.1	11.4	0.1	0
Small Pelagic Fish	Mexico	2.3	0.2	0.1	0
Squids	Mexico	4.9	1.5	0.3	34
Tunas and Billfish	Mexico	2.1	2.6	1.2	40

Appendix D. Ecopath input data for California Current Canada model.

Table D.1. Ecopath input data for US California Current model. TL= trophic level; B= biomass (mt/km2); P/B= production / biomass (/year); Q/B= consumption / biomass (/year); EE= ecotrophic efficiency.

	Group	TL	B	P/B	Q/B	EE
1	Orcas	4.24	0.00	0.02	7.40	0.00
2	Dolphins	3.74	0.04	0.02	7.30	0.00
3	Birds	3.85	0.00	0.10	91.70	0.92
4	Baleen whales	3.13	0.16	0.02	13.37	0.97
5	Steller sea lion	3.90	0.02	0.06	12.70	0.80
6	Harbor seal	4.10	0.01	0.08	17.40	0.63
7	Pelagic sharks	4.20	0.03	0.14	1.00	0.71
8	Dogfish	3.27	1.30	0.20	2.70	0.04
9	Squid	3.34	1.94	3.00	6.00	0.73
10	Chinook salmon	3.73	0.39	2.30	6.91	0.87
11	Coho salmon	3.91	0.25	2.30	6.52	0.99
12	POP	3.20	0.50	0.10	0.40	0.49
13	Shelf rockfish	3.23	0.83	0.13	2.20	0.73
14	Slope rockfish	3.27	0.59	0.06	1.90	0.48
15	Sablefish	3.45	1.47	0.36	2.00	0.05
16	Pollock	3.07	0.72	0.41	1.96	0.58
17	Halibut	3.35	0.20	0.32	1.18	0.51
18	Flatfishes	3.07	1.45	0.35	2.80	1.00
19	Hake	3.13	10.06	0.75	5.80	0.34
20	Mackerel	3.26	0.27	0.35	6.00	0.11
21	Sardine	2.10	1.00	3.50	5.00	0.29
22	Herring	2.60	3.23	1.00	7.17	0.92
23	Eulachon	2.40	0.40	2.00	18.00	0.50
24	Mesopelagic fish	3.05	7.58	1.50	3.00	0.30
25	Jellyfish	3.06	15.00	3.00	12.00	0.01
26	Euphasids	2.10	27.04	8.00	40.00	0.85
27	Copepods	2.00	16.61	14.00	70.00	0.55
28	Zooplankton	2.00	10.00	25.00	183.30	0.35
29	Benthos	2.01	7.00	1.00	3.00	0.98
30	Diatoms	1.00	22.80	130.00	0.00	0.45
31	Large phytoplankton	1.00	9.33	180.00	0.00	0.58
32	Small phytoplankton	1.00	9.33	200.00	0.00	0.74
33	Detritus	1.00	1.00			0.07

Appendix E. Ecopath diet composition for California Current Canada model.

Table E.1. Diet composition, Canada California Current Ecopath model. Diets sum to 1.

Group	Prey\Predator	1	2	3	4	5	6	7	8
1	Orcas	0	0	0	0	0	0	0	0
2	Dolphins	0	0	0	0	0	0	0	0
3	Birds	0	0	0.00	0	0	0	0	0
4	Baleen whales	0.00	0	0	0	0	0	0	0
5	Steller sea lion	0.03	0	0	0	0	0	0	0
6	Harbor seal	0.00	0	0	0	0	0	0	0
7	Pelagic sharks	0	0	0	0	0	0	0	0
8	Dogfish	0.07	0	0	0	0	0	0	0
9	Squid	0.47	0.16	0.05	0	0.05	0.04	0.60	0
10	Chinook salmon	0.01	0.05	0	0	0.07	0.19	0.01	0.00
11	Coho salmon	0.01	0.05	0	0	0.07	0.19	0.01	0.00
12	POP	0.01	0	0	0	0.00	0.00	0	0
13	Shelf rockfish	0.01	0.05	0	0	0.00	0.00	0	0.00
14	Slope rockfish	0.01	0.01	0	0	0	0	0	0.00
15	Sablefish	0.00	0	0	0	0.00	0	0	0
16	Pollock	0.15	0	0.02	0	0.28	0.21	0	0
17	Halibut	0.00	0	0	0	0.00	0	0.05	0
18	Flatfishes	0.15	0.05	0	0	0.16	0.04	0.05	0.04
19	Hake	0	0	0	0	0.00	0.02	0	0.06
20	Mackerel	0.03	0	0	0	0.03	0	0.02	0
21	Sardine	0	0.11	0.16	0.01	0.27	0.24	0.02	0
22	Herring	0.04	0.32	0.07	0.11	0.01	0.04	0.02	0.09
23	Eulachon	0	0	0	0	0.04	0.02	0	0
24	Mesopelagic fish	0	0	0.60	0	0	0	0.03	0
25	Jellyfish	0	0	0	0	0	0	0	0.04
26	Euphasids	0	0	0.01	0.67	0	0	0.05	0.56
27	Copepods	0	0	0	0.10	0	0	0	0
28	Zooplankton	0	0	0	0.11	0	0	0	0.01
29	Benthos	0	0.20	0	0	0	0	0	0.19
30	Diatoms	0	0	0	0	0	0	0	0
31	Large phytoplankton	0	0	0	0	0	0	0	0
32	Small phytoplankton	0	0	0	0	0	0	0	0
33	Detritus	0	0	0	0	0	0	0	0
34	Import	0	0	0.09	0	0	0	0.15	0

Table E.1. Diet composition, Canada California Current Ecopath model. Diets sum to 1.

Prey/Predator	**9**	**10**	**11**	**12**	**13**	**14**	**15**	**16**	**17**	**18**	**19**
1	0	0	0	0	0	0	0	0	0	0	0
2	0	0	0	0	0	0	0	0	0	0	0
3	0	0	0	0	0	0	0	0	0	0	0
4	0	0	0	0	0	0	0	0	0	0	0
5	0	0	0	0	0	0	0	0	0	0	0
6	0	0	0	0	0	0	0	0	0	0	0
7	0	0	0	0	0	0	0	0	0	0	0
8	0	0	0	0	0	0	0	0	0	0	0
9	0.19	0.21	0.44	0.03	0.01	0.10	0.17	0.01	0.02	0.00	0.00
10	0	0	0	0	0	0	0.00	0	0	0	0
11	0	0	0	0	0	0	0.00	0	0	0	0
12	0	0.00	0	0	0	0	0.00	0	0	0	0
13	0	0.00	0	0	0.01	0.02	0.00	0	0	0	0
14	0	0	0	0	0	0	0.00	0	0.01	0	0
15	0	0	0	0	0	0	0	0.00	0.01	0	0
16	0	0	0	0.00	0	0	0	0.00	0.12	0.00	0
17	0	0	0	0	0	0	0	0	0	0	0
18	0	0.00	0.01	0.01	0.05	0.08	0.00	0	0.12	0.02	0
19	0	0	0	0	0	0	0.00	0	0	0	0.00
20	0	0	0	0	0	0	0	0	0.01	0	0
21	0.00	0.08	0.06	0.03	0.08	0	0.02	0.02	0	0.01	0
22	0.02	0.19	0.02	0	0.16	0	0.08	0.02	0.01	0.06	0.01
23	0.00	0.10	0.04	0.03	0	0	0.02	0	0	0	0
24	0.05	0.24	0.25	0.06	0.00	0	0.03	0.00	0.01	0	0.02
25	0	0	0	0	0.01	0	0.09	0	0	0	0
26	0.28	0.16	0.16	0.67	0.38	0.20	0.26	0.30	0	0.01	0.92
27	0.28	0	0	0	0	0	0	0.10	0	0.08	0
28	0.19	0	0.03	0.01	0.05	0	0.09	0.15	0	0	0
29	0	0	0	0.16	0.25	0.60	0.25	0.40	0.69	0.82	0
30	0	0	0	0	0	0	0	0	0	0	0
31	0	0	0	0	0	0	0	0	0	0	0
32	0	0	0	0	0	0	0	0	0	0	0
33	0	0	0	0	0	0	0	0	0	0	0
34	0	0	0	0	0	0	0	0	0	0	0.05

Table E.1. Diet composition, Canada California Current Ecopath model. Diets sum to 1.

Prey/Predator	20	21	22	23	24	25	26	27	28	29
1	0	0	0	0	0	0	0	0	0	0
2	0	0	0	0	0	0	0	0	0	0
3	0	0	0	0	0	0	0	0	0	0
4	0	0	0	0	0	0	0	0	0	0
5	0	0	0	0	0	0	0	0	0	0
6	0	0	0	0	0	0	0	0	0	0
7	0	0	0	0	0	0	0	0	0	0
8	0	0	0	0	0	0	0	0	0	0
9	0.01	0	0	0	0	0	0	0	0	0
10	0	0	0	0	0	0	0	0	0	0
11	0	0	0	0	0	0	0	0	0	0
12	0	0	0	0	0	0	0	0	0	0
13	0	0	0	0	0	0	0	0	0	0
14	0	0	0	0	0	0	0	0	0	0
15	0	0	0	0	0	0	0	0	0	0
16	0	0	0	0	0	0	0	0	0	0
17	0	0	0	0	0	0	0	0	0	0
18	0	0	0	0	0	0	0	0	0	0
19	0	0	0	0	0	0	0	0	0	0
20	0	0	0	0	0	0	0	0	0	0
21	0.06	0	0	0	0	0	0	0	0	0
22	0.08	0	0	0	0	0	0	0	0	0
23	0.00	0	0	0	0	0	0	0	0	0
24	0.12	0	0	0	0	0	0	0	0	0
25	0	0	0	0	0	0	0	0	0	0
26	0.70	0	0	0	0.5	0.6	0	0	0	0
27	0.02	0	0.3	0.2	0.3	0.3	0.05	0	0	0
28	0.01	0.1	0.3	0.2	0.2	0.1	0.05	0	0	0
29	0	0	0	0	0	0	0	0	0	0.01
30	0	0.3	0.1	0.2	0	0	0.3	0.7	0.1	0
31	0	0.3	0.15	0.2	0	0	0.3	0	0.35	0
32	0	0.3	0.15	0.2	0	0	0.3	0.2	0.45	0
33	0	0	0	0	0	0	0	0.1	0.1	0.99
34	0	0	0	0	0	0	0	0	0	0

Appendix F. Ecopath input data for California Current U.S. model.

Table F.1. Ecopath input data for US California Current model. TL= trophic level; B= biomass (mt/km2); P/B= production / biomass (/year); Q/B= consumption / biomass (/year); EE= ecotrophic efficiency.

	Group name	TL	B	P/B	Q/B	EE
1	Seals and sealions	4.09	0.02	0.08	16.00	0.00
2	Lingcod					
2.1	Lingcod larvae	3.00	0.00		44.97	0.00
2.2	Lingcod juvenile	3.11	0.08		5.00	0.00
2.3	Lingcod adult	4.10	0.50		2.20	0.15
3	Cabezon					
3.1	Cabezon larvae	3.00	0.00		71.72	0.00
3.2	Cabezon juvenile	3.66	0.16		3.63	0.00
3.3	Cabezon adult	3.66	0.40		1.50	0.06
4	Shortbelly rockfish					
4.1	Shortbelly rockfish larvae	3.00	0.00		105.77	0.00
4.2	Shortbelly rockfish juvenile	3.71	0.04		5.35	0.00
4.3	Shortbelly rockfish adult	3.71	0.40		1.95	0.29
5	Nearshore rockfish					
5.1	Nearshore rockfish larvae	3.00	0.00		48.88	0.00
5.2	Nearshore rockfish juvenile	3.40	0.25		4.34	0.00
5.3	Nearshore rockfish adult	3.40	2.00		1.60	0.25
6	Widow rockfish					
6.1	Widow rockfish larvae	3.00	0.00		74.41	0.00
6.2	Widow rockfish juvenile	3.31	0.10		7.16	0.01
6.3	Widow rockfish adult	3.31	2.80		2.10	0.31
7	Flatfish	3.01	8.70	0.30	2.50	0.78
8	Sablefish	3.88	2.70	0.06	1.95	0.00
9	Dogfish	3.85	1.00	0.20	2.50	0.00
10	Hake	3.39	25.00	0.23	2.50	0.78
11	Salmon	3.73	0.40	0.93	5.80	0.54
12	Mackerel	3.28	0.30	0.35	6.00	0.00
13	Sardine	2.70	0.70	0.50	5.00	0.58
14	Benthic fish	3.08	4.70	0.40	2.00	0.88
15	Forage fish and mesopelagics	3.06	35.00	1.00	6.00	0.82
16	Cephalopods	2.65	2.00	2.00	6.00	0.77
17	Crabs	3.09	1.70	0.50	3.00	0.92
18	Shrimps	2.60	3.10	2.50	11.00	0.80
19	Jellyfish	3.00	2.40	5.50	20.00	0.96
20	Zooplankton	2.00	52.00	8.00	40.00	0.72

Table F.1. Ecopath input data for US California Current model. TL= trophic level; B= biomass (mt/km2); P/B= production / biomass (/year); Q/B= consumption / biomass (/year); EE= ecotrophic efficiency.

21	Abalone					
21.1	Abalone juvenile	2.00	0.00		104.42	0.00
21.2	Abalone adult	2.00	1.00		4.00	0.50
22	Benthic invertebrates	2.00	52.00	3.50	15.00	0.33
23	Macroalgae	1.00	10.00	10.00	0.00	0.02
24	Phytoplankton	1.00	55.00	120.00	0.00	0.32
25	Detritus	1.00	50.00			0.14

Appendix G. Ecopath diet composition for California Current U.S. model.

Table G.1. Diet composition, U.S. California Current Ecopath model. Diets sum to 1.

Group	Prey/Predator	1	2	3	4	5	6
1	Seals and sealions	0	0	0	0	0	0
2	Lingcod larvae	0	0	0	0	0	0
3	Lingcod juvenile	0	0	0	0	0	0
4	Lingcod adult	0	0	0	0	0	0
5	Cabezon larvae	0	0	0	0	0	0
6	Cabezon juvenile	0	0	0	0	0	0
7	Cabezon adult	0	0	0	0	0	0
8	Shortbelly rockfish larvae	0	0	0	0	0	0
9	Shortbelly rockfish juvenile	0	0	0	0	0	0
10	Shortbelly rockfish adult	0	0	0	0	0	0
11	Nearshore rockfish larvae	0	0	0	0	0	0
12	Nearshore rockfish juvenile	0	0	0	0	0	0
13	Nearshore rockfish adult	0	0	0	0	0	0
14	Widow rockfish larvae	0	0	0	0	0	0
15	Widow rockfish juvenile	0	0	0	0	0	0
16	Widow rockfish adult	0	0	0	0	0	0
17	Flatfish	0.15	0	0	0	0	0
18	Sablefish	0	0	0	0	0	0
19	Dogfish	0	0	0	0	0	0
20	Hake	0.25	0	0	0.1	0	0.15
21	Salmon	0	0	0	0	0	0
22	Mackerel	0	0	0	0	0	0
23	Sardine	0	0	0	0	0	0
24	Benthic fish	0	0	0	0.4	0	0.05
25	Forage fish and mesopels	0.5	0	0.1	0.5	0	0
26	Cephalopods	0.1	0	0	0	0	0
27	Crabs	0	0	0	0	0	0.2
28	Shrimps	0	0	0	0	0	0.3
29	Jellyfish	0	0	0	0	0	0
30	Zooplankton	0	1	0	0	1	0
31	Abalone juvenile	0	0	0	0	0	0
32	Abalone adult	0	0	0	0	0	0
33	Benthic invertebrates	0	0	0.9	0	0	0.3
34	Macroalgae	0	0	0	0	0	0
35	Phytoplankton	0	0	0	0	0	0
36	Detritus	0	0	0	0	0	0

Table G.1. Diet composition, U.S. California Current Ecopath model.

Prey/Predator	7	8	9	10	11	12	13	14	15
1	0	0	0	0	0	0	0	0	0
2	0	0	0	0	0	0	0	0	0
3	0	0	0	0	0	0	0	0	0
4	0	0	0	0	0	0	0	0	0
5	0	0	0	0	0	0	0	0	0
6	0	0	0	0	0	0	0	0	0
7	0	0	0	0	0	0	0	0	0
8	0	0	0	0	0	0	0	0	0
9	0	0	0	0	0	0	0	0	0
10	0	0	0	0	0	0	0	0	0
11	0	0	0	0	0	0	0	0	0
12	0	0	0	0	0	0	0	0	0
13	0	0	0	0	0	0	0	0	0
14	0	0	0	0	0	0	0	0	0
15	0	0	0	0.001	0	0	0	0	0
16	0	0	0	0	0	0	0	0	0
17	0	0	0	0	0	0.02	0.02	0	0
18	0	0	0	0	0	0	0	0	0
19	0	0	0	0	0	0	0	0	0
20	0.15	0	0	0	0	0.01	0.01	0	0.02
21	0	0	0	0	0	0	0	0	0
22	0	0	0	0	0	0	0	0	0
23	0	0	0	0	0	0	0	0	0
24	0.05	0	0.05	0.049	0	0.02	0.02	0	0.03
25	0	0	0.5	0.5	0	0.25	0.25	0	0.05
26	0	0	0	0	0	0.05	0.05	0	0
27	0.2	0	0	0	0	0	0	0	0
28	0.3	0	0.05	0.05	0	0	0	0	0
29	0	0	0.1	0.1	0	0.05	0.05	0	0.2
30	0	1	0.3	0.3	1	0.6	0.6	1	0.65
31	0	0	0	0	0	0	0	0	0
32	0	0	0	0	0	0	0	0	0
33	0.3	0	0	0	0	0	0	0	0.05
34	0	0	0	0	0	0	0	0	0
35	0	0	0	0	0	0	0	0	0
36	0	0	0	0	0	0	0	0	0

Table G.1. Diet composition, U.S. California Current Ecopath model.

Prey/Predator	16	17	18	19	20	21	22	23	24
1	0	0	0	0	0	0	0	0	0
2	0	0	0	0	0	0	0	0	0
3	0	0	0	0	0	0	0	0	0
4	0	0	0	0	0	0	0	0	0
5	0	0	0	0	0	0	0	0	0
6	0	0	0.000101	0	0	0	0	0	0
7	0	0	0	0	0	0	0	0	0
8	0	0	0	0	0	0	0	0	0
9	0	0	0	0	0	0	0	0	0
10	0	0	0	0	0	0	0	0	0
11	0	0	0	0	0	0	0	0	0
12	0	0	0	0	0	0	0	0	0
13	0	0	0	0	0	0	0	0	0
14	0	0	0	0	0	0	0	0	0
15	0	0	0	0	0	0	0	0	0
16	0	0	0	0	0	0	0	0	0
17	0	0	0.0505	0.1	0	0.02	0	0	0.05
18	0	0	0	0	0	0	0	0	0
19	0	0	0	0	0	0	0	0	0
20	0.02	0	0.1616	0.25	0	0	0	0	0
21	0	0	0	0	0	0	0	0	0
22	0	0	0	0	0	0	0	0	0
23	0	0	0	0	0	0.01	0.1	0	0
24	0.03	0	0.0808	0.13	0	0.03	0	0	0
25	0.05	0	0.3333	0.2	0.35	0.62	0.2	0	0
26	0	0	0.0909	0	0	0.02	0	0	0
27	0	0	0	0.02	0	0	0	0	0
28	0	0.02	0.0202	0.05	0.03	0	0	0	0.05
29	0.2	0	0.0909	0	0	0	0	0	0
30	0.65	0	0.0707	0.2	0.62	0.3	0.7	0.7	0
31	0	0	0	0	0	0	0	0	0
32	0	0	0	0	0	0	0	0	0
33	0.05	0.98	0.101	0.05	0	0	0	0	0.9
34	0	0	0	0	0	0	0	0	0
35	0	0	0	0	0	0	0	0.3	0
36	0	0	0	0	0	0	0	0	0

Table G.1. Diet composition, U.S. California Current Ecopath model.

Prey/Predator	25	26	27	28	29	30	31	32	33
1	0	0	0	0	0	0	0	0	0
2	0	0	0	0	0	0	0	0	0
3	0	0	0	0	0	0	0	0	0
4	0	0	0	0	0	0	0	0	0
5	0	0	0	0	0	0	0	0	0
6	0	0	0	0	0	0	0	0	0
7	0	0	0	0	0	0	0	0	0
8	0	0	0	0	0	0	0	0	0
9	0	0	0	0	0	0	0	0	0
10	0	0	0	0	0	0	0	0	0
11	0	0	0	0	0	0	0	0	0
12	0	0	0	0	0	0	0	0	0
13	0	0	0	0	0	0	0	0	0
14	0	0	0	0	0	0	0	0	0
15	0	0	0	0	0	0	0	0	0
16	0	0	0	0	0	0	0	0	0
17	0	0	0	0	0	0	0	0	0
18	0	0	0	0	0	0	0	0	0
19	0	0	0	0	0	0	0	0	0
20	0	0	0	0	0	0	0	0	0
21	0	0	0	0	0	0	0	0	0
22	0	0	0	0	0	0	0	0	0
23	0	0	0	0	0	0	0	0	0
24	0	0	0	0	0	0	0	0	0
25	0	0	0	0	0	0	0	0	0
26	0.01	0	0	0	0	0	0	0	0
27	0	0	0	0	0	0	0	0	0
28	0	0	0.15	0	0	0	0	0	0
29	0.05	0	0	0	0	0	0	0	0
30	0.91	0.35	0	0.2	1	0	0	0	0
31	0	0	0	0	0	0	0	0	0
32	0	0	0	0	0	0	0	0	0
33	0.03	0.3	0.85	0.4	0	0	0	0	0
34	0	0	0	0	0	0	0.5	0.5	0
35	0	0	0	0	0	0.9	0	0	0.3
36	0	0.35	0	0.4	0	0.1	0.5	0.5	0.7

Appendix H. Net multipliers of sardine abundance by species group and country.

Table H.1. Net multipliers of sardine abundance by species group and country. Parameters are for second-order polynomial regression equation, estimated for relative biomass.

	Canada			US and Mexico		
Group	α_1	α_2	α_3	α_1	α_2	α_3
Benthopelagic Fish	1.8×10^{-3}	0.110	0.897	-3×10^{-6}	-1×10^{-5}	1.000
Cod-likes	6×10^{-4}	-0.006	1.006	3×10^{-4}	-4×10^{-4}	1.005
Crustaceans	-	-	-	-5×10^{-5}	8×10^{-4}	0.999
Elasmobranchs	1.9×10^{-3}	-0.047	1.046	2×10^{-4}	-0.003	1.004
Flatfishes	7×10^{-4}	-0.012	1.015	-6×10^{-5}	7×10^{-4}	0.999
Mollusks	-	-	-	-2×10^{-5}	3×10^{-4}	0.999
Other	-	-	-	-	-	-
Other Invertebrates	3×10^{-7}	-0.003	1.003	-3×10^{-5}	5×10^{-4}	0.999
Pelagic Fish	-3×10^{-4}	0.059	0.952	-	-	-
Salmon	-2×10^{-4}	0.033	0.973	3×10^{-4}	0.014	0.987
Sardine	-	-	-	-	-	-
Small Pelagic Fish	-2×10^{-4}	-0.001	1.002	2×10^{-4}	-0.002	1.002
Squids	1×10^{-4}	-0.006	1.006	-6×10^{-5}	-0.001	0.999
Tunas and Billfish	-	-	-	-	-	-