MARINE ECOSYSTEM RESTORATION WITH A FOCUS ON CORAL REEF ECOSYSTEMS

by

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Abstract

The declines of fish populations in ecosystems around the globe have triggered considerable interest in marine ecosystem restoration. In addition to focusing on individual fish populations, there is increased emphasis on understanding inter-species interactions and on understanding the human relationships with the ecosystems. My thesis approaches marine restoration from (a) practical aspects of considering multispecies interactions in the ecosystem (Ecopath with Ecosim models), estimating unreported and illegal catches (influence tables) and policy that considers the concerns of multiple stakeholders (Bayesian influence diagram modeling); (b) theoretical aspects of carrying capacity and fish life history analyzed using life history parameters (Population dynamics modeling).

I begin my thesis by exploring the technological, socio-economic, and political history of Raja Ampat in Eastern Indonesia (my geographical focus) to understand resource management challenges and to calculate the trends in relative misreporting of fisheries catch. The unreported fish catch exceeds the reported fish catch by a factor of 1.5. My next chapter explores the ecological benefits of establishing marine protected areas for coral reef ecosystems in Raja Ampat using Ecopath, Ecosim and Ecospace models. I estimate an ideal minimum size of no-take areas— the size of no-take area at which the biomass density of reef fish reached an asymptote—to be 16 to 25 km^2. Analysis of biomass density of reef fish in MPAs led to questions about ecosystem carrying capacity. To explore carrying capacity, I reconstruct ancient snapper population biomass using archaeological data obtained from fish middens using equilibrium age structure model. The results show that the ancient snapper population was about 2 to 4 times higher than the modern population biomass. To model the differing utilities of different stakeholders, in the next chapter, I develop a bayesian influence diagram model. The results indicate that restricting net fisheries and implementing 25% fisheries closure are robust scenarios favored under several combinations of the modeled variables and utility functions. The final chapter explores how the life history parameters of fish species affect the population response to restoration. It is expected that slow growing species would show a greater response to protection than fast growing species.

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Dedication

To my parents

&

parents-in-law

Co-Authorship Statement

Chapter 1 and Chapter 7 are the introductory and concluding chapters of the thesis. All the other chapters are written as publishable manuscripts. Chapter 2 is published, Chapter 3 has been submitted and Chapters 4 to 6 are yet to be submitted to for publication. I am the lead author on all the manuscripts and assume responsibility for the analyses and the results presented.

Chapter 2 is co-authored with Cameron Ainsworth, Tony Pitcher, Yohannes Goram, and Rashid Sumaila. The methodology was based on previous work by Tony Pitcher and Cameron Ainsworth. Yohannes Goram provided several key insights into local aspects of the fisheries in Raja Ampat. Rashid Sumaila provided guidance with the economic aspects of the analyses.

Chapter 3 is co-authored with Cameron Ainsworth and Tony Pitcher. The Ecospace models are built upon Ecopath and Ecosim models for Raja Ampat. Cameron Ainsworth and I built the Ecopath and Ecosim models. Tony Pitcher provided guidance in all aspects of the research.

Chapter 4 is co-authored with Tony Pitcher, Foss Leach and Alison MacDiarmid. The original idea for this work was developed by Tony Pitcher and Foss Leach. Foss Leach also provided the archaeological data used in the analyses. Alison MacDiarmid provided information and guidance on the modern aspects of the snapper fisheries in New Zealand.

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Chapter 6 is co-authored with Tony Pitcher who provided guidance and edits at different stages of the work.

1 Introduction

This thesis asks the following questions with regard to marine ecosystem restoration. (1) What are the management challenges in a tropical coral reef ecosystem (Raja Ampat, Indonesia)? (2) What is an ideal minimum size for a marine protected area (MPA) from an ecological perspective? (3) What is the carrying capacity of a species in a system? (4) How can multiple uses from the ecosystem be considered together? And lastly, (5) how do life-history parameters of fish influence the response of a population to restoration? This introductory chapter begins with a brief description of the history of fisheries management; mainly the events and changes that have led to the current focus on restoration. The chapter also narrates the history of marine resource use and the current management status in Raja Ampat. Raja Ampat is a regency (the administrative hierarchy of a regency is one level below the province and roughly corresponds to a district) located adjacent to the northwest tip of the province Papua in Eastern Indonesia. Finally, this chapter discusses the key questions asked in each chapter of the thesis.

1.1 Rationale: brief history of fisheries management

At the great international Birkbeck's fishery exhibition held in 1883 to celebrate success of British fisheries (Nature News 1883), it was debated whether fisheries were exhaustible or not, and whether there was need of fisheries management (Smith 1994). Though with many qualifications—referring only to pelagic fish and to the then present mode of fishing —Thomas Huxley maintained that fisheries were inexhaustible and nothing humans did could affect the numbers of fish in the oceans (Smith 1994). Huxley's arguments were countered by Ray Lankester (Smith 1994), who mainly raised concerns about recruitment overfishing. Complaints against the destructive nature of trawlers were older, but inquiry commissions and researchers repeatedly exonerated trawl fisheries mainly based on inconclusive evidence; all forms of fisheries were allowed ‘unbridled expansion’ (Roberts 2007). When evidence[1] was made available in the form of declining catch per unit effort (CPUE), it was not heeded enough (Smith 1994).

Shifting baselines (Pauly et al. 1998) ensured that even in the 1950s, the perception of inexhaustibility continued[2]. However, in the period from the late-1800s to the mid-1990s, the science of the study of fish populations and fisheries management had grown, and more evidence was collected on the potential of fisheries to impact fish populations.[3] It was recognized that fisheries management was advantageous; this itself was a significant step forward from the previous century when fisheries had been allowed to grow unhindered (Roberts 2007). Some of the important scientific works that continue to be extensively used today include the catch equation given by Baranov (1918) and the growth model given by von Bertalanffy (1938). Population growth was described as a sigmoid curve (Sigmoid curve theory by Graham, 1935), and further progress on the same idea led to the development of the ‘surplus production model’ (Schaefer 1954). The Schaefer surplus production model probably, on account of its ease of application and focus on maximizing yield, became very popular, and was applied to almost every fishery. The Schaefer surplus production model was probably the first scientific work that theoretically showed that excessive effort could lead to population declines.

Major changes in fisheries management happened in the period 1950 to 2000. Large, long-distance fleets spread to distant coastlines; perceptions of fisheries declines gained strength, and several nations proceeded to secure their coastlines. To quell the increasing conflict over the seas, The United Nations Convention on the Law of the Sea (UNCLOS) established the ‘Exclusive Economic Zone’, and most countries signed the UNCLOS in 1982. Each coastal state had sovereign rights over the resources in the adjacent continental shelves and was responsible for managing and conserving the same resources. All states subsidized fisheries and fisheries catch increased all over the world. The

[1] The impoverishment of the Sea’ by Walter Garstang published in 1900, Walter Garstang evidently stated that fisheries were exhaustible and were in the process of being exhausted, cited from Smith (1994).

[2] The inexhaustible sea’ by H. Daniel and F. Minot, published in 1954, cited from Roberts (2007).

[3] Russell’s 1942 lecture on ‘Overfishing’, and 2 post-war increase in fish abundance in North Sea, cited from Beverton and Holt (1957).

maximum sustainable yield (MSY), the maximum of the surplus production from a stock that can be sustainably harvested each year, became the "key paradigm" and "played a central role" in management (Punt and Smith 2001). The MSY was estimated most commonly using Schaefer's surplus production model given in the 1950s (Schaefer 1954). There were several problems with the MSY approach including the assumption of equilibrium and the assumption of CPUE being directly proportional to abundance (Larkin 1977; Sissenwine 1978; Punt and Smith 2001). When surplus production models were applied to growing fisheries, the debt associated with exploiting standing stocks of populations was overlooked, and by the time the problem was recognized, several fisheries had exceeded their MSY levels, and the fishing industry had become overcapitalized. By late 1980s it was recognized that fisheries could not "sustain uncontrolled exploitation and development" (FAO 1995).

Consequent to overcapitalization, the goal of fisheries management was to control fishing capacity. Stock assessment models more complex than surplus production models were developed. The virtual population models which included both catch and age information had originated earlier (Derzhavin 1922; cited in Sparre and Venema 1998), but they became popular from 1950 to 2000 (Megrey 1989; cited in Sparre and Venema 1998). Dynamic pool models were introduced in the 1950s (Beverton and Holt 1957). Complex fisheries stock assessments methods continued to evolve to provide accurate management advice on quotas, harvest control rules, fixed escapement rules, and reference points (Pauly and Morgan 1987; Hilborn and Waiters 1992; Hannesson 1993; Walters and Pearse 1996; McAllister and Kirkwood 1998; Cooke 1999; Walters and Martell 2004). However, the extraction levels suggested by science were often negotiated upwards at political negotiations (Daw and Gray 2005). Restrictive regulations met with resistance from the fishers—the fishers adopted the restrictions only when their opportunity cost for fishing elsewhere or altogether leaving fishing were higher (Clark 2006). In addition, the lack of compliance led to underreporting and issues related to illegal, unreported and unregulated (IUU) fishing (Pitcher et al. 2002; Sumaila et al. 2006) became serious. Inaccurate catch data can lead to inaccurate management recommendations (Patterson et al. 2001). Several schemes were introduced to control fishing capacity, but most of the schemes were made toothless by the clever strategies of the fishers. Vessel buy back schemes, especially when they were anticipated, were "economically equivalent to direct vessel subsidies" because the fishers invested in capacity and used these schemes to get rid of inefficient vessels (Clark 2006). Measures to limit entry led fishers to invest in increasing fishing power by increasing engine, gear or fishing hold capacity (capital stuffing) (Clark 2006). The difficulty in reducing fishing capacity was further exacerbated by non-malleable (fishing vessels that could not be easily converted to other uses) or only partially malleable fishing fleets (Clark 2006).

The precautionary principle became popular in fisheries science in the early 1990s to reduce the chance of collapse of exploited species and to limit impacts on non-target species and habitats (Garcia 1994; Costanza et al. 1999). The FAO Code of Conduct for Responsible Fisheries (FAO-CCRF) adopted in 1995 provided principles for conservation, management and development of fisheries resources: the conservation guidelines promoted precautionary approach, advocated limit reference points, protection of critical habitats, and recovery of depleted stocks (FAO 1995). An extensive study (Pitcher et al. 2009) a decade after the FAO-CCRF was adopted found that several developed and developing countries performed poorly with respect to the adopted standards. The last two decades (1990-2010) documented the tragedies of overfishing and raised concern regarding the health of the oceans. Unchecked overcapacity led several fisheries to collapse; one of the most notable was the unimaginable setback from the collapse of the North Atlantic cod. Fish populations declined worldwide (Myers et al. 1996; Rose and Kulka 1999; Morris et al. 2000; Dulvy et al. 2003; Hutchings and Reynolds 2004). Declining fish populations pushed several marine ecosystems towards collapse (Hughes 1994; Pauly et al. 1998; Jackson et al. 2001; Pandolfi et al. 2003). Dulvy et al. (2003) documented 133 local, regional, and global extinctions.

1.2 Context

This section includes a brief description of the study area, the history of marine resource use, and the current status of management in Raja Ampat.

1.2.1 Brief description of study area - Raja Ampat Islands

Raja Ampat is a Regency located within the northwest tip of the province Papua in eastern Indonesia. The region is an archipelago that extends over 45,000 km^2, it includes 4 large islands (Waigeo, Batanta, Salawati and Misool), and around 600 small islands. The archipelago is located in the 'Coral Triangle' (Donnelly et al. 2003). The area encompasses a variety of marine habitats including some of the most biodiverse coral reef areas on Earth (McKenna et al. 2002a; Donnelly et al. 2003). It is estimated that Raja Ampat possesses over 75 percent of the world's known coral species (Halim and Mous 2006). More than 1000 fish species, manta rays, sharks, and short finned pilot whales and turtle rookeries, are the other highlights of marine life in the region. Several authors (Allen 2002; Erdmann and Pet 2002) have referred to the exceptional habitat diversity and consequent rich biodiversity of the region.

The most abundant reef fish families in the region are gobies (Gobiidae), damselfishes (Pomacentridae), wrasses (Labridae), cardinalfishes (Apogonidae), groupers (Serranidae), butterflyfishes (Chaetodontidae), surgeonfishes (Acanthuridae), blennies (Blenniidae), parrotfishes (Scaridae), and snappers (Lutjanidae). These 10 families represent 61% of reef fish species in Raja Ampat (Allen 2002). A survey across 45 reef sites revealed that most of the reef sites were in 'excellent' to 'good' condition (measured based on reef condition index), but few sites were observed to be in poor condition (McKenna et al. 2002b). Stress and damage was observed on 85% of the surveyed sites; the predominant stressors were fishing pressure (including destructive fishing methods), siltation, and eutrophication/pollution.

From 1960 to 1993, the human population has increased in the region at an average rate of 3% per year (2.7% in 1960 to 1980 (McNicoll 1982); 3.41% in 1980 to 1990 (Surbakti et al. 2000)). Small-scale fisheries operations on the reefs and in the inshore areas provide livelihoods for around 24,000 fishers (Dohar and Anggraeni 2007). A total of 196 species, representing 59 genera and 19 families, are classified as target species for reef fisheries in Raja Ampat (Tanda 2002). Reef fish constitutes about 40% of the catch by the local fishers; the remainder of the catch is contributed by Spanish mackerel, sea cucumber, snails, and lobsters in almost equal proportions (Muljadi 2004). The fishing gear types used in the region include spear fishing, reef gleaning, shore gillnets, driftnets, permanent and portable traps, spear diving (for fish and invertebrates), diving specifically for live fish (with or without the aid of cyanide), blast fishing using dynamite, trolling, purse seining, pole and line, set lines, lift nets, and shrimp trawls. The shrimp trawl fishery is located in the Arafura Sea, southeast of Raja Ampat. A foreign fleet, consisting mainly of powered Philippino tuna vessels, also operates in deeper areas in the north of Raja Ampat (Muljadi, A[4]. pers. comm.).

1.2.2 Historical and political background of marine resource use in Raja Ampat

It is important to understand resource use history because the current perceptions towards management are based on the events in the history[5]. Importance of fisheries resources to the people of Raja Ampat increased after 10th century AD. The people in Raja Ampat communicated with people from Biak, Seram, Central Mollucas, and south of Papua to Fak Fak and began barter of marine snails (Trochus spp.), turtle etc. By 13th century they learnt the technology to make canoes. Marine resource use increased from the 14th to 16th centuries with the formation of a trading triangle with the Sultanate of Ternate and Tidore in the north. There were increased interactions of regional people with the seafarers and traders from Biak (Ploeg 2002), who came to anchor and fish for a couple of months annually in Raja Ampat. Over time, migrants from Biak and the Mollucas began to live in Raja Ampat islands. During the 17th-19th century, fish catch from Raja Ampat was sent as tax to the King of Mollucas who had become the King over Raja Ampat after defeating the local King.

[4] Andreas Muljadi TNC-CTC. Jl Gunung Merapi No. 38, Kampung Baru, Sorong, Papua, Indonesia 98413.

[5] This account is written based on an unpublished account that belongs to the Council of Traditional Ethnic Groups in Raja Ampat: Dewan Adat Suku Maya Kepulauan Raja Ampat (The Council of Traditional Ethnic Groups in Raja Ampat) 2006. Sejarah pemanfaatan sumberdaya alam di kepulauan Raja Ampat (Perspektif Adat) The history of the utilization of nature resources in the island of Ring Ampat (Traditional perspective)

Towards the end of 19th century, the Dutch established control over the Mollucas and Papua (then referred to as Netherlands East Indies jointly) (Ploeg 2002). After World War II, Papua became a separate administrative unit under Dutch command. Papuans were trained to hold lower and middle level administrative positions. When Indonesia secured independence in 1949, Papua was not part of the sovereign territory but remained under Dutch control. Negotiations were continued between the Dutch and the Indonesian governments regarding the fate of Papua. Finally in 1962, the Dutch ceded control over Papua[6] to Indonesia, and in May 1963, Indonesia took over the administration of the region. From then on, Raja Ampat and other regions in Papua witnessed immigration (locally referred to as "Indonesianization") from other provinces of Indonesia including Java, Sulawesi, and Sumatra. The period under Indonesian government rule is regarded as a time of discrimination in which Papuans had fewer rights than Indonesians; in fact, "the rivalries and antagonism between Papuans and Indonesians were even more apparent after Indonesia took control" (Chauvel 2005). From the perspective of marine resource use in Raja Ampat, the migrants introduced different kinds of gears and crafts based on the skills that belonged to the regions where they had come from. The immigrant fishers began to catch fish and sell them to Java and Sumatra. The immigrants did not recognize the 'adat' values (traditional resource management principles) since all resources were now supposed to be owned by the state. There was a gradual transformation from a subsistence-based lifestyle to a cash-based economy.

At the national level, interest in management of the fisheries sector grew at a gradual pace in Indonesia. Repelita IV (Rencana Pembangunan Lima Tahun - Five Year Development Plan) in 1984 and Repelita VI in 1994 placed emphasis on integrated coastal zone management including "fish production and environmental protection of marine areas" (Patlis et al. 2001). An independent Dinas Kelautan dan Perikanan (DKP – the Ministry of Marine Affairs and Fisheries) was established in 2000 (Patlis et al. 2001).

[6] From 1969 to 1973 the region was referred to as 'West Irian' and 'Irian Barat', after which the region was renamed as 'Irian Jaya'. Later in 2002, the name 'Papua' was adopted.

Political reformation in Indonesia from the centralized Suharto regime (New Order 1965) to a decentralized government (Act no. 22/1999 on regional autonomy and Act no. 25/1999 on financial relations) gave more powers to the provincial and regency governments. The provinces were allowed authority up to 12 nautical miles from the coastal shoreline including "supervision of fishery resources, licensing of permits for catching, and cultivating fish" while regencies were allowed authority within 4 nautical miles from the shoreline. These acts specially mentioned that traditional fishing rights would not be restricted by the "regional territorial sea delimitation". Except for few areas of governance, the regencies had the authority for all decision making within their jurisdiction. Regencies, because they had the political authority and were in close proximity with the resource users, had the ability to establish management programs adapted to local interests (Patlis et al. 2001).

1.2.3 Current management in Raja Ampat

A decree by the Bupati (Regent) in 2003 declared Raja Ampat a Maritime Regency 'Kabupaten Bahari' (Conservation International 2008). The goals of the Raja Ampat Regency are to improve the welfare and prosperity of the community by promoting fisheries, conservation, and tourism while respecting customary rights (Raja Ampat Regency 2007). The regency established a new network of marine reserves in 2006 covering more than 650,000 hectares of sea area and 44% of reef area in Raja Ampat. The DKP pledged that 30% of the marine area of Raja Ampat would be declared as protected zones, exceeding the national goal of 20%, and that no-take areas would be established within the protected zones (Rahawarin, B[7]. pers. comm.).

In February 2002, the Papuan Traditional Council (Dewan Adat Papua) held the Papuan congress. The goal of the council was to integrate the indigenous and immigrant population in Raja Ampat within the traditional adat and to revive the traditional marine tenure in the region in collaboration with the fisheries management department in Raja Ampat. Studies of fisher perceptions in Raja Ampat showed that the fishers believed that fish catch had declined over the past 10 to 20 years (Muljadi 2004; Ainsworth et al.

[7] Becky Rahwarin DKP, Raja Ampat. Jl. A. Yani, Kuda laut, Sorong, Papua.

2008). The main threats to management in the region, as recognized by the regency and the local population, were blast fishing, cyanide fishing, fishing by migrant fishers, and overfishing (Muljadi 2004; Raja Ampat Regency 2007)

In 2005, concerned with the issues of fisheries management and with the intention to develop environmentally sound ecosystem based policies, the regency government participated in a collaborative project—the Birds Head Seascape Ecosystem Based Management (BHS EBM) project—funded by the David and Lucille Packard Foundation. The BHS EBM project involved three environmental NGO partners (Conservation International, The Nature Conservancy's Southeast Asia Center for Marine Protected Areas, and WWF-Indonesia) in a science-based initiative in partnership with local stakeholders to explore ecosystem processes relevant to management. This author conducted an evaluation of the expected progress from the successful implementation of the project. The evaluation was based on the framework of Ward et al. (2002) framework on Ecosystem Based Management (EBM); the framework evaluates EBM using five overall principles, six criteria for success, and twelve implementation steps. The project was able to increase awareness on the threats to coral reef resources (mainly destructive fishing methods in the region). The project was also able to fill several gaps of information by conducting surveys on reef health and fishing effort. During the project conducted demographic surveys were conducted to evaluate fisheries and other economic sectors. An inventory on habitats and eco-regions was created and ecosystem models were built for analysis of policies for fisheries management. For details of the evaluation of EBM in Raja Ampat please refer to Appendix A at the end of the thesis. It is expected that successful implementation of the project will improve the management status of the region considerably.

1.3 Thesis focus: restoration in fisheries with focus on coral reef ecosystems

1.3.1 Restoration within an EBM framework

With the increase in accounts of marine declines, the emphasis on restoration has increased (Pitcher 2001; Fox et al. 2003; Russ and Alcala 2003; Lotze et al. 2006). For example, on World Oceans Day in 2010, legislators from predominant fishing nations agreed on a ‘Global Marine Recovery Strategy’ to restore the declined fish populations (GLOBE 2010). Moreover, recent assessment of the status of exploited fish stocks emphasized the need for large scale effort at rebuilding marine ecosystems, but also stated that recovery and rebuilding were poorly understood (Worm et al. 2009). Studies of historical and archaeological evidence (Jackson et al. 2001; MacKenzie et al. 2002; Lotze and Milewski 2004; Roberts 2007; Rose 2007) showed the high abundances of species in ancient ecosystems. It was suggested that the historical levels of population abundance be used as goals for rebuilding the current ecosystems (Pitcher and Pauly 1998; Pitcher 2001). Decreasing fishing capacity and establishing MPAs are two key tools to ensure rebuilding (Pauly et al. 2002). Though MPAs have been advocated atleast as early as 1997 as valuable insurance against environmental and management uncertainty (Roberts 1997), MPAs have also faced considerable scepticism about their value in restoring species abundances (Willis et al. 2003). More recent work has shown that different species respond differently to protection (Claudet et al. 2006; McClanahan et al. 2007; Molloy et al. 2009). Placement and design of MPAs has also been widely researched both from theoretical and practical perspectives (Walters et al. 1998; Ball and Possingham 2000; Halpern 2003; Lubchenco et al. 2003; Shanks et al. 2003).

In addition to the focus on restoration and rebuilding, increased need was observed to understand the inter-relationships of species and their consequences in fisheries management decisions (Link 2002a; Christensen et al. 2007). The concept of EBM to incorporate issues beyond single species questions, began becoming popular in 1990s (Szaro et al. 1998; Link 2002b). Incorporation of ecosystem approaches in fisheries management was discussed at the FAO conference on responsible fisheries (FAO 2002) conference and guidelines were published in 2003 (Garcia et al. 2003). Ward et al. (2002) proposed a framework for EBM based on three sets of attributes: overall principles (5 attributes; Table 2, page 19 in Ward et al. 2002); criteria for success (6 attributes; Table 3, pages 19-20 in Ward et al. (2002); and implementation steps (12 attributes; Table 6, pages 50-51 in Ward et al. 2002). The attributes included a broad range of concepts from data gathering, recognising ecosystem values to involving stakeholders in management. Legislative requirements in several countries demanded the inclusion of principles of EBM (Hall and Mainprize 2004); numerous international conventions also required this type of holistic view (Garcia et al. 2003). In spite of its popularity, EBM continued to remain an "elusive concept" that was interpreted differently by different users (Hilborn et al. 2004). In 2009 (McLeod and Leslie 2009), more than 200 scientists and policy experts agreed on a common goal of EBM as "conservation of the long-term potential of ecosystems to deliver of a broad suite of ecosystem services" and also agreed on a definition for EBM—"key aspects of the definition include: (1) considering the entire ecosystem, including humans; (2) taking an integrated view across species, sectors, activities, and concerns; (3) evaluating cumulative impacts across sectors; (4) emphasizing the protection of ecosystem structure, functioning, and key processes; (5) accounting for the interconnectedness within and among systems; and (6) recognizing the interdependence among ecological, social, economic, and institutional perspectives." EBM in Great Barrier Reef Marine Park is referred to as "gold standard for EBM", and the success has been attributed to "equal attention to human and natural" aspects (Ruckelshaus et al. 2008).

1.3.2 Coral reefs

Coral reefs are magnificent marine ecosystems; their incredible biodiversity supports numerous types of livelihoods. Coral reefs are characterized by three main features: (1) high species diversity (Sale 1977; Connell 1978), (2) complexity of relationships (Sale and Douglas 1984; Hixon and Beets 1993), and (3) high rates of production (Lewis 1977). The species richness and composition of "functional groups", species occupying the same niche or delivering the same function within an ecosystem, on reef ecosystems play an important role in the ability of the ecosystems to respond to fishing and other stressors (Bellwood et al. 2004). Changes in both target and non-target reef fish communities (Jennings et al. 1995; Jennings and Polunin 1996; Jennings and Polunin 1997; Sala et al. 1998), benthic and algal communities (Sala et al. 1998) have been attributed to fishing. The changes ultimately alter the competitive balance and associated trophic structure among reef communities (Roberts 1995; McClanahan 1997). Local abundances of coral-reef fish are also determined by the relative magnitudes of larvae recruitment, colonization by juveniles and adults, predation and competition for refuges—each of them varies through time and space (Swearer et al. 1999). A global review of the status of coral reefs found that several coral reef ecosystems had declined; the review suggested that management for status quo was a "weak" goal; rather efforts should be made to restore the reefs (Pandolfi et al. 2003). Similar to the changes observed with fishing, recovery is also dependent on the trophic composition of reef ecosystems (Mumby et al. 2006; Hughes et al. 2007). Building an ecosystem model of the coral reef ecosystem can offer insights into options for management and recovery of the ecosystem.

1.3.3 Ecopath with Ecosim

This thesis uses ecosystem models to represent the species interactions on coral reefs. Ecosystem models are able to integrate information from different components of the ecosystem. Ecopath with Ecosim (EwE) incorporates biological information on species with fisheries catch information to explore the effects of species and fisheries interactions. The EwE models help to understand ecosystem behaviour and to assist analysis of trade-offs in marine policy (Christensen and Walters 2004a). Since its origins, (Polovina 1984) in the span of 25 years the modeling tool EwE has advanced considerably (Christensen 1992; Walters et al. 1997; Walters et al. 1998; Pauly et al. 2000; Walters et al. 2000). The various capacities of the EwE include modeling trophic linkages, life history stanzas of species, simulation of fisheries impacts, optimal policy searches and so on. Ecosystem modeling using EwE has become very popular (Christensen and Walters 2005); a total of more than 100 EwE models have been built with at least one EwE model for almost all (excluding some polar regions) large marine ecosystems (LMEs).

EwE and its spatial component Ecospace are used to represent the food web of Raja Ampat and simulate trophic interactions of interest to fisheries and conservation. Ecopath provides a “static picture of the ecosystem trophic structure” (Walters et al. 1997). The ecosystem components are summarized into functional groups (species aggregated by trophic similarity). Ecopath describes the flux of matter and energy in and out of each group and models human influence through fishery removals. Ecosim allows modeling of species composition changes over time (Walters et al. 1997) and exploration of past and future effects of fishing (Christensen and Walters 2004b). EwE models have been used in fisheries management to a limited extent (Christensen and Walters 2005). Reviews and criticisms of the EwE approach (Fulton et al. 2003; Christensen and Walters 2004b; Plagányi and Butterworth 2004; Plagányi 2007) highlight the strengths and weaknesses of the modelling approach. Please refer to Appendix B for details on parameterization of the EwE models for Raja Ampat. Restoration scenarios are explored for the coral reef ecosystems in Raja Ampat are explored using the models.

The ideas for fisheries restoration in the thesis are developed within the framework of EBM of coral reefs. In brief my thesis explores the history of events in the region to understand management challenges, explores multiple utilities of stakeholders from the perspective of marine ecosystem restoration, explores ecosystem response in marine protected areas to protection using ecosystem models, explores carrying capacity of a species in a system, and explores the influence of life history in the response of a species to protection.

1.3.4 Thesis outline

Catch data missing from records is an ecological and a management problem. It is an ecological problem because when missing catch is not accounted for in stock assessment or in ecosystem models then over-optimistic levels of resource status may be estimated (Pauly et al. 2002; Pitcher et al. 2002). The optimistic estimates can lead to limited management controls or increased investment into fisheries development which can further deplete the resource. From a management perspective, the most basic functions of a management agency are to record or estimate the amount of catch and the number of fishing operations in a region, so illegal, unreported and unregulated (IUU) catch is the first indication of flaws in management. Therefore, in Chapter 2, I explore the unreported catch in Raja Ampat to estimate the total extractions from the ecosystem and also to understand the management challenges in the region.

In Chapter 2, the history of regulatory, technological, political, and market changes in the fishery from 1960 to present were analyzed using a method of "semi-quantitative Monte-Carlo integration of historical sources" originally developed by Pitcher and Watson (2000). The advantage of the technique was that all available data on under-reporting, no matter what regulatory regime was in place, could be combined to calculate the trends. The trends in the relative rate of misreporting of fisheries catch were estimated and converted to absolute values using anchor points. Anchor points were known rates of misreporting obtained either from the literature or from the surveys in the region or based on expert opinion. A Monte Carlo analysis was used to estimate the likely quantity of IUU catch with associated error ranges for six fisheries: reef fish, tuna, anchovy, shark, sea cucumber, and lobster. This method of estimating IUU catches has been used previously to estimate IUU for fisheries in the North Atlantic (Forrest et al. 2001), Iceland and Morocco (Pitcher et al. 2002), British Columbia, Canada (Ainsworth and Pitcher 2005), and Eritrea (Tesfamichael and Pitcher 2007). When this dissertation work was started, the coral reef ecosystem in Raja Ampat was a highly data-limited system; though, some information on fisheries landings was available. Therefore, adopting a method that could use multiple sources of data was essential. In addition, studying the history of the events led to greater understanding of challenges for fisheries management in Raja Ampat. The perspectives gathered in this study were useful in framing the questions addressed in the other chapters of the dissertation.

After being declared a maritime regency, the Raja Ampat Regency government undertook the initiative to manage Raja Ampat on guidelines of EBM. Towards this goal, they set up a network of marine protected areas in 2006 (Conservation International 2008.). In Chapter 3, I analyzed ecological restoration from effort reduction and specific gear restrictions inside MPAs using the Ecospace model for Raja Ampat. Ecospace integrates Ecopath and Ecosim across a two dimensional spatially explicit domain (Walters et al. 1998; Pauly et al. 2000). In Ecospace, functional groups which are linked by trophic relationships migrate between cells on a grid map of habitat (Walters et al. 1998; Pitcher and Buchary 2002). Ecospace models are useful tools to explore ecological changes in MPAs in response to change in fishing pressure. The research questions were identified through discussions with the Regency fisheries managers and scientific partners working in Eastern Indonesia. Ecospace has been used previously to explore changes in MPAs (Walters et al. 1998; Pitcher and Buchary 2002; Salomon et al. 2002; Jiang et al. 2008; Le Quesne and Codling 2009).

Chapter 3 also explored no-take zoning options for Raja Ampat. I used high resolution sub-area Ecospace models for Dampier Strait, Misool and Kofiau (Islands in Raja Ampat) to compare outcomes between MPAs in several size combinations. This chapter also developed an 'ideal minimum size' for an MPA—it is the minimum size of an MPA after which ecological benefits in terms of reef fish biomass density begin to asymptote. Suggestions on percentage of area to be closed as no-take in marine reserves range from 10 to 50% (Lauck et al. 1998; Dahlgren and Sobel 2000; Botsford 2001; Roberts et al. 2003; Parnell et al. 2006; Stewart et al. 2007). In a re-zoning effort, the no-take area in Great Barrier Reef Marine Park was increased six fold to 33%, and this no-take area includes at least 20% of each of the 70 bioregions (Olsson et al. 2008). Plans exist to designate 20% of North-western Hawaiian Islands Coral Reef Ecosystem Reserve as no-take area (Hoegh-Guldberg 2006). The cited studies usually represent the percentage area to be closed or the percentage of the species population to be protected. Declaring percentages of habitats to be protected is useful at the level of international policy guidelines. However, at a local management level, a guideline on the area to be protected in square kilometers (like the one developed in the Chapter) would be more valuable.

Among the 4 major islands of Raja Ampat, MPAs have been declared in Kofiau, Misool and Dampier islands. Kofiau is a smaller island in comparison and has a relatively homogenous population. Misool and Dampier are larger islands and have diverse populations. A proportion of the population in Misool are descendants of immigrants from Sulawesi who still maintain trade and personal contact with their relatives in Sulawesi. There is lack of camaraderie between the indigenous Papuan population and immigrants who have settled from elsewhere. Chapter 3 thus tried to arrive at ecological solutions for restoration which offered flexibility in their application in the real world. In the real world optimum size of an MPA would depend on many factors which are extraneous to the ecological system.

Chapter 4 used archaeological data to reconstruct the snapper population biomass on the west coast of New Zealand in ~1400 AD with the goal to understand ecosystem carrying capacity Studies based on historical and archaeological evidence have shown declines of several orders of magnitude in exploited species of marine mammals, and turtles, as well as Atlantic cod, also declines in coral cover; the past abundances have been suggested as targets for restoration (Jackson et al. 2001; Pitcher 2001; Rosenberg et al. 2005). Extensive fisheries archaeology work has been done in New Zealand; so my work was based on an archaeological fish population in New Zealand. The data consisted of the length frequency of archaeological New Zealand snapper (*Chrysophrys auratus*) population fished in early Maori times (~ 1400 AD). I decided to work on a single population since it was a more reliable approach than modelling (using scanty information) complex inter-species interactions in ancient ecosystems. The total mortality of the ancient population was estimated by fitting mixture distributions to the length frequency data. An equilibrium age structure model was applied to the growth and mortality information for calculating the ancient biomass. From the perspective of archaeological science, the methodology is highly useful since it provides a way to arrive at estimates of the ancient population based on data collected from archaeological middens. The estimates of the modern population were obtained from modern stock assessments and were compared with the results for the ancient population obtained in the chapter.

Chapter 5 evaluated a suite of restoration scenarios for coral reef ecosystems in Raja Ampat for robustness to uncertainties of ecosystem status, tourism growth, interest in conservation, and utility functions of different stakeholders. Fisheries restoration always put some restrictions on fishing activities. Sometimes fishers view the management body as an adversary (Hilborn 2007); at other times fishers themselves are interested in the protection of the environment (Crawford et al. 2004). Also, fishers are not the only stakeholders associated with using the ecological resources of the marine system. The groups of stakeholders I considered are the tourism industry and conservationists. In Chapter 5, I combined the results from ecosystem simulation model of Raja Ampat with projections of tourism and conservation benefits into an ‘Influence diagram’ (special application of Bayesian network analysis) to evaluate alternate restoration scenarios for the Raja Ampat coral reef ecosystem. The restoration scenarios were modeled using the Ecosim model for Raja Ampat. Scenarios were evaluated based on different combinations of utility functions of the different stakeholders: fishers, tourism industry, and conservationists. The chapter also explored levels of expected revenue from tourism that could offset the losses to fishers under different restoration scenarios.

Chapter 6 was devoted to understand how growth parameters of a species influenced restoration (i.e. the response to reduction in mortality). Two factors control the recovery of a population: biomass per recruit and increase in the number of recruits. The biomass per recruit (B/R) and was calculated using growth parameters, fishing mortality, and natural mortality parameters. The chapter analysed several combinations of mortality and growth parameters to determine if there was a pattern in the response of fish B/R to protection that depended on the growth parameters of the population. To determine the response in recruitment, the chapter analysed the range within which the equilibrium recruitment varied for any species at different levels of recruitment compensation. Finally the species were grouped according to their growth parameters to observe any overarching patterns in the variation in mean recruitment.

Finally, Chapter 7 summarized and synthesized the findings in the thesis. It discusses broad generalizations resulting from the thesis, the overall significance of the findings to research in marine restoration, and some directions for future work.

1.4 References

Ainsworth C. H., T. J. Pitcher. 2005. Estimating illegal, unreported and unregulated catch in British Columbia's marine fisheries. Fisheries Research **75**:40-55.

Ainsworth C. H., T. J. Pitcher, and C. Rotinsulu. 2008. Evidence of fishery depletions and shifting cognitive baselines in Eastern Indonesia. Biological Conservation **141**(3):848-859.

Allen G. R. 2002. Reef fishes of Raja Ampat Islands, Papua province, Indonesia. In: McKenna S.A., G.R. Allen, and S. Suryadi, editors. A marine rapid assessment of the Raja Ampat Islands, Papua Province, Indonesia. Conservation International, Washington, DC. RAP Bulletin of Biological Assessment **22**: 46-57.

Ball I. R., H. P. Possingham. 2000. MARXAN (V1. 8.2). Marine reserve design using spatially explicit annealing, A Manual.

Baranov T.I. 1918. On the question of the biological basis of fisheries. *Nauch.issledov. iktiol. Inst. Izv.* (Report of Division of fisheries management and scientific study of the fishing industry) Moscow. **I**(1):81-128.

Bellwood D. R., T. P. Hughes, C. Folke, and M. Nyström. 2004. Confronting the coral reef crisis. Nature **429**(6994):827-833.

Bertalanffy L. von 1938. A quatitative theory of organic growth. Inquiries of growth laws II. Human Biology **10**(2):181-213.

Beverton R. J. H., S. J. Holt. 1957. On the dynamics of exploited fish populations. Chapman and Hall, London, 1957; Facsimile reprint 1993.

Botsford H. 2001. Dependence of sustainability on the configuration of marine reserves and larval dispersal distance. Ecology Letters **4**(2):144-150.

Chauvel R. 2005. Constructing Papuan nationalism: history, ethnicity, and adaptation. Washington DC: East-West Center Washington. Policy Studies **14**. 140p.

Christensen V. 1992. ECOPATH II--A software for balancing steady-state ecosystem models and calculating network characteristics. Ecological Modelling **61**(3):169-185.

Christensen V., C. J. Walters. 2004a. Trade-offs in ecosystem-scale optimization of fisheries management policies. Bulletin of Marine Science **74**(3):549-562.

Christensen V., C. J. Walters. 2004b. Ecopath with Ecosim: methods, capabilities and limitations. Ecological Modelling **172**(2-4):109-139.

Christensen V., C. J. Walters. 2005. Using ecosystem modeling for fisheries management: Where are we? ICES CM **1000**:19.

Christensen V., K. A. Aiken, and M. C. Villanueva. 2007. Threats to the ocean: on the role of ecosystem approaches to fisheries. Social Science Information **46**(1):67.

Clark C. W. 2006. The worldwide crisis in fisheries: economic models and human behavior. Cambridge University Press. 270p.

Claudet J., D. Pelletier, J. Y. Jouvenel, F. Bachet, and R. Galzin. 2006. Assessing the effects of marine protected area (MPA) on a reef fish assemblage in a northwestern Mediterranean marine reserve: Identifying community-based indicators. Biological Conservation **130**(3):349-369.

Connell J. H. 1978. Diversity in tropical rain forests and coral reefs. Science **199**(4335):1302.

Conservation International. 2008. Environmental friendly development in Raja Ampat Available:
<www.conservation.or.id/home.php?catid=30&tcatid=178&page=g_peluang.detail >
[Accessed on July 30, 2008].

Cooke J. 1999. Improvement of fishery-management advice through simulation testing of harvest algorithms. ICES Journal of Marine Science **56**(6):797-810.

Costanza R., F. Andrade, P. Antunes, M. Van Den Belt, D. Boesch, D. Boersma, F. Catarino, S. Hanna, K. Limburg, and B. Low. 1999. Ecological economics and sustainable governance of the oceans. Ecological Economics **31**(2):171-187.

Crawford B. R., A. Siahainenia, C. Rotinsulu, and A. Sukmara. 2004. Compliance and enforcement of community-based coastal resource management regulations in North Sulawesi, Indonesia. Coastal Management **32**(1):39-50.

Dahlgren C. P., J. Sobel. 2000. Designing a Dry Tortugas Ecological Reserve: How big is big enough? to do what? Bulletin of Marine Science **66**(3):707-719.

Daw T., T. Gray. 2005. Fisheries science and sustainability in international policy: a study of failure in the European Union's Common Fisheries Policy. Marine Policy **29**(3):189-197.

Derzhavin A. N. 1922. The stellate sturgeon (Acipenser *stellatus Pallas*), a biological sketch. (in Russian). Byulleten Bakinskoi Ikhtiologicheskoi Stantsii **1**:1-393.

Dohar A. G., D. Anggraeni. 2007. Valuasi Ekonomi Sumberdaya Alam di Kepulauan Raja Ampat. Conservation International, Bekerjasama dengan, Universitas Negeri Papua. (in Indonesian) Economic valuation report for Raja Ampat, Indonesia; Conservation

International Indonesia. Available: Rotinsulu C. Gunung Arfak CI. Jl. 45. Sorong, Papua, Indonesia. email: chris@conservation.or.id.

Donnelly R., D. Neville, and P. J. Mous. 2003. Report on a rapid ecological assessment of the Raja Ampat Islands, Papua, Eastern Indonesia. The Nature Conservancy Coral Triangle Center. Indonesia. Final Draft. 250p.

Dulvy N. K., Y. Sadovy, and J. D. Reynolds. 2003. Extinction vulnerability in marine populations. Fish and Fisheries **4**(1):25-64.

Erdmann M. V., J. S. Pet. 2002. A rapid marine survey of the northern Raja Ampat islands. Henry Foundation/The Nature Conservancy/NRM/EPIQ. Indonesia: The Nature Conservancy Coral Triangle Center. Available: <www.coraltriangle.org/downloads/Raja4%20trip%20report.pdf> [Accessed on July 21, 2006]

FAO. 1995. Code of conduct for responsible fisheries. FAO, Rome. 41p.

FAO. 2002. Report of the Reykjavik Conference on Responsible Fisheries in the Marine Ecosystem. Reykjavik, Iceland, 1-4 October 2001. FAO. Rome. FAO Fisheries Report No. **658**. 182p.

Forrest R., T. J. Pitcher, R. Watson, H. Valtysson, and S. Guenette. 2001. Estimating illegal and unreported catches from marine ecosystems: two case studies. In: Pitcher T.J., U.R. Sumaila, and D. Pauly, editors. Fisheries impacts on North Atlantic ecosystems: evaluations and policy explorations. Fisheries Centre Research Reports **9**(5):81-93.

Fox H. E., J. S. Pet, R. Dahuri, and R. L. Caldwell. 2003. Recovery in rubble fields: long-term impacts of blast fishing. Marine Pollution Bulletin **46**(8):1024-1031.

Fulton E. A., D. M. Smith, and C. R. Johnson. 2003. Effect of complexity on marine ecosystem models. Marine Ecology Progress Series **253**:1-16.

Garcia S. M. 1994. The precautionary principle: its implications in capture fisheries management. Ocean and Coastal Management **22**(2):99-125.

Garcia S. M., A. Zerbi, C. Aliaume, T. Dochi, and G. Lasserre. 2003. The ecosystem approach to fisheries. Issues, terminology, principles, institutional foundations, implementation and outlook. FAO Fisheries Technical Paper. No. **443**. FAO, Rome. 71p.

GLOBE. 2010. Legislators agree global marine recovery strategy to address collapse of world fisheries. World Oceans Day press release. Global Legislators Organization for a Balanced Environment fund. Available: <http://copenhagen.globeinternational.org/news/globe_briefing_uk_low_carbon_transition_plan.aspx> [Accessed on July 15, 2010]

Graham M. 1935. Modern theory of exploiting a fishery, and application to North Sea trawling. ICES Journal of Marine Science **10**(3)264-274.

Halim A., P.J. Mous. 2006. Community perceptions of marine protected area management in Indonesia. Bali, Indonesia: The Nature Conservancy - Coral Triangle Center. Available: < www.coraltrianglecenter.org/downloads/MPA_Perc_Monitoring.pdf> [Accessed on February 12, 2007]

Hall S. J., B. Mainprize. 2004. Towards ecosystem-based fisheries management. Fish and Fisheries **5**(1):1-20.

Halpern B. S. 2003. The impact of marine reserves: do reserves work and does reserve size matter? Ecological Applications **13**(sp1):117-137.

Hannesson R. 1993. Fishing capacity and harvest rules. Marine Resource Economics **8**(2):133-143.

Hilborn R. 2007. Managing fisheries is managing people: what has been learned? Fish and Fisheries **8**(4):285-296.

Hilborn R., C. J. Walters. 1992. Quantitative fisheries stock assessment: choice, dynamics and uncertainty. Reviews in Fish Biology and Fisheries **2**(2):177-178.

Hilborn R., A. E. Punt, and J. Orensanz. 2004. Beyond band-aids in fisheries management: fixing world fisheries. Bulletin of Marine Science **74**(3):493-507.

Hixon M. A., J. P. Beets. 1993. Predation, prey refuges, and the structure of coral-reef fish assemblages. Ecological Monographs **63**(1):77-101.

Hoegh-Guldberg O. 2006. Ecology: complexities of coral reef recovery. Science **311**(5757):42-43.

Hughes T. P. 1994. Catastrophes, phase-shifts, and large-scale degradation of a Caribbean coral reef. Science **265**:1547-1551.

Hughes T. P., M. J. Rodrigues, D. R. Bellwood, D. Ceccarelli, O. Hoegh-Guldberg, L. McCook, N. Moltschaniwskyj, M. S. Pratchett, R. S. Steneck, and B. Willis. 2007. Phase shifts, herbivory, and the resilience of coral reefs to climate change. Current Biology **17**(4):360-365.

Hutchings J. A., J. D. Reynolds. 2004. Marine fish population collapses: consequences for recovery and extinction risk. Bioscience **54**(4):297-309.

Jackson J. B. C., M. X. Kirby, W. H. Berger, K. A. Bjorndal, L. W. Botsford, B. J. Bourque, R. H. Bradbury, R. Cooke, J. Erlandson, J. A. Estes, T. P. Hughes, S. Kidwell,

C. B. Lange, H. S. Lenihan, J. M. Pandolfi, C. H. Peterson, R. S. Steneck, M. J. Tegner, and R. R. Warner. 2001. Historical overfishing and the recent collapse of coastal ecosystems. Science **293**(5530):629-638.

Jennings S., N. V. C. Polunin. 1996. Effects of fishing effort and catch rate upon the structure and biomass of Fijian reef fish communities. Journal of Applied Ecology **33**(2):400-412.

Jennings S., N. V. C. Polunin. 1997. Impacts of predator depletion by fishing on the biomass and diversity of non-target reef fish communities. Coral Reefs **16**(2):71-82.

Jennings S., E. Grandcourt, and N. Polunin. 1995. The effects of fishing on the diversity, biomass and trophic structure of Seychelles' reef fish communities. Coral Reefs **14**(4):225-235.

Jiang H., H. Q. Cheng, W. J. F. Le Quesne, H. G. Xu, J. Wu, H. Ding, and F. Arreguin-Sanchez. 2008. Ecosystem model predictions of fishery and conservation trade-offs resulting from marine protected areas in the East China Sea. Environmental Conservation **35**(2):137-146.

Larkin P. 1977. An epitaph for the concept of maximum sustained yield. Transactions of the American Fisheries Society **106**(1):1-11.

Lauck T., C. W. Clark, M. Mangel, and G. R. Munro. 1998. Implementing the precautionary principle in fisheries management through marine reserves. Ecological Applications **8**(1) Supplement:S72-S78.

Le Quesne W. J. F., E. A. Codling. 2009. Managing mobile species with MPAs: the effects of mobility, larval dispersal, and fishing mortality on closure size. ICES Journal of Marine Science: **66**(1):122-131.

Lewis J. B. 1977. Processes of organic production on coral reefs. Biological Reviews **52**(3):305-347.

Link J. S. 2002a. Ecological considerations in fisheries management: When does it matter? Fisheries **27**(4):10-17.

Link J.S. 2002b. What does ecosystem-based fisheries management mean? Fisheries **27**(4):18-21.

Lotze H. K., I. Milewski. 2004. Two centuries of multiple human impacts and successive changes in a North Atlantic food web. Ecological Applications **14**(5):1428-1447.

Lotze H. K., H. S. Lenihan, B. J. Bourque, R. H. Bradbury, R. G. Cooke, M. C. Kay, S. M. Kidwell, M. X. Kirby, C. H. Peterson, and J. B. C. Jackson. 2006. Depletion, degradation, and recovery potential of estuaries and coastal seas. Science **312**(5781):1806.

Lubchenco J., S. R. Palumbi, S. D. Gaines, and S. Andelman. 2003. Plugging a hole in the ocean: the emerging science of marine reserves 1. Ecological Applications **13**(1) Supplement:S3-S7.

MacKenzie B. R., J. Alheit, D. J. Conley, P. Holm, and C. C. Kinze. 2002. Ecological hypotheses for a historical reconstruction of upper trophic level biomass in the Baltic Sea and Skagerrak. Canadian Journal of Fisheries and Aquatic Sciences **59**(1):173-190.

McAllister M., G. Kirkwood. 1998. Bayesian stock assessment: a review and example application using the logistic model. ICES Journal of Marine Science **55**(6):1031.

McClanahan T. R. 1997. Primary succession of coral-reef algae: differing patterns on fished versus unfished reefs. Journal of Experimental Marine Biology and Ecology **218**(1):77-102.

McClanahan T. R., N. A. J. Graham, J. M. Calnan, and M. A. MacNeil. 2007. Toward pristine biomass: reef fish recovery in coral reef marine protected areas in Kenya. Ecological Applications **17**(4):1055-1067.

McKenna S. A., G. R. Allen, and S. Suryadi (editors). 2002a. A marine rapid assessment of the Raja Ampat Islands, Papua Province, Indonesia. Conservation International, Washington, DC. RAP Bulletin of Biological Assessment **22**. 193p.

McKenna S. A., P. Bolis, and G. R. Allen. 2002b. Condition of coral reefs at the Raja Ampat Islands, Papua Province, Indonesia. In: McKenna, S.A., G.R. Allen, and S. Suryadi, editors. A marine rapid assessment of the Raja Ampat Islands, Papua Province, Indonesia. Conservation International, Washington, DC. RAP Bulletin of Biological Assessment **22**: 66-78.

McLeod K., H. Leslie. 2009. Ecosystem-based management for the oceans. Island Press, Washington DC. 368p.

McNicoll G. 1982. Recent demographic trends in Indonesia. Population and Development Review **8**(4):811-819.

Megrey B. A. 1989. Review and comparison of age-structured stock assessment models from theoretical and applied points of view. American Fisheries Society Symposium. **6**:8-48.

Molloy P. P., I. B. McLean, and I. M. Côté. 2009. Effects of marine reserve age on fish populations: a global meta-analysis. Journal of Applied Ecology **46**(4):743-751.

Morris A. V., C. M. Roberts, and J. P. Hawkins. 2000. The threatened status of groupers (Epinephelinae). Biodiversity and Conservation **9**(7):919-942.

Muljadi A. 2004. Coastal Rural Appraisal report in Kofiau and Misool area of Raja Ampat Islands. Sorong, Indonesia: The Nature Conservancy, South East Asia Centre for Marine Protected Areas. Available: amuljadi@tnc.org.

Mumby P. J., C. P. Dahlgren, A. R. Harborne, C. V. Kappel, F. Micheli, D. R. Brumbaugh, K. E. Holmes, J. M. Mendes, K. Broad, J. N. Sanchirico, K. Buch, S. Box, R. W. Stoffle, and A. B. Gill. 2006. Fishing, trophic cascades, and the process of grazing on coral reefs. Science **311**(5757):98-101.

Myers R. A., J. A. Hutchings, and N. J. Barrowman. 1996. Hypotheses for the decline of cod in the North Atlantic. Marine Ecology Progress Series. **138**(1):293-308.

Nature News. 1883. The great international fisheries exhibition**.** Nature **27**:536-538. Available:
< www.nature.com/nature/journal/v27/n701/abs/027536a0.html> [Accessed on June 18, 2010].

Olsson P., C. Folke, and T. P. Hughes. 2008. Navigating the transition to ecosystem-based management of the Great Barrier Reef, Australia. Proceedings of the National Academy of Sciences **105**(28):9489-9494.

Pandolfi J. M., R. H. Bradbury, E. Sala, T. P. Hughes, K. A. Bjorndal, R. G. Cooke, D. McArdle, L. McClenachan, M. J. H. Newman, G. Paredes, R. R. Warner, and J. B. C. Jackson. 2003. Global trajectories of the long-term decline of coral reef ecosystems. Science **301**:955-958.

Parnell P. E., P. K. Dayton, C. E. Lennert-Cody, L. L. Rasmussen, and J. J. Leichter. 2006. Marine reserve design: optimal size, habitats, species affinities, diversity, and ocean microclimate. Ecological Applications **16**(3):945-962.

Patlis J. M., R. Dahuri, M. Knight, and J. Tulungen. 2001. Integrated coastal management in a decentralized Indonesia: how can it work. Pesisir and Lautan (Indonesian Journal of Coastal and Marine Resources) **4**(1):24-39.

Patterson K., R. Cook, C. Darby, S. Gavaris, L. Kell, P. Lewy, B. Mesnil, A. Punt, V. Restrepo, and D. W. Skagen. 2001. Estimating uncertainty in fish stock assessment and forecasting. Fish and Fisheries **2**(2):125-157.

Pauly D., G. Morgan. 1987. Length-based methods in fisheries research. ICLARM conference proceedings 13, International Center for Living Aquatic Resources Management, Manila, Philippines, and Kuwait Institute for Scienctific Research, Safat, Kuwait. 468p.

Pauly D., V. Christensen, S. Guénette, T. J. Pitcher, U. R. Sumaila, C. J. Walters, R. Watson, and D. Zeller. 2002. Towards sustainability in world fisheries. Nature **418**(6898):689-695.

Pauly D., V. Christensen, and C. J. Walters. 2000. Ecopath, Ecosim and Ecospace as tools for evaluating ecosystem impact of fisheries. ICES Journal of Marine Science **57**:697-706.

Pauly D., V. Christensen, J. Dalsgaard, R. Froese, and F. Torres Jr. 1998. Fishing down marine food webs. Science **279**(5352):860-863.

Pitcher T., D. Kalikoski, G. Pramod, and K. Short. 2009. Not honouring the code. Nature **457**(7230):658-659.

Pitcher T. J., E. A. Buchary. 2002. Ecospace simulations for Hong Kong Waters. In: Pitcher T., E. Buchary, and P. Trujillo, editors. Spatial simulations of Hong Kong's marine ecosystem: forecasting with MPAs and human-made reefs. Fisheries Centre Research Reports **10**(3):27-35.

Pitcher T. J., R. Watson. 2000. The basis for change 2: estimating total fishery extractions from marine ecosystems of the North Atlantic. In: Pauly D., T.J. Pitcher, editors. Methods for assessing the impact of fisheries on marine ecosystem of the North Atlantic. Fisheries Centre Research Reports **8**(2):40-53.

Pitcher T. J., D. Pauly. 1998. Rebuilding ecosystems, not sustainability, as the proper goal of fishery management. Fish and Fisheries **23**:311-330.

Pitcher T. J., R. Watson, R. Forrest, H. P. Valtýsson, and S. Guénette. 2002. Estimating illegal and unreported catches from marine ecosystems: a basis for change. Fish and Fisheries **3**(4):317-339.

Pitcher T. J. 2001. Fisheries managed to rebuild ecosystems? Reconstructing the past to salvage the future. Ecological Applications **11**(2):601-617.

Plagányi E. E., D. S. Butterworth. 2004. A critical look at what Ecopath with Ecosim can and cannot achieve in practical fisheries management. African Journal of Marine Science **26**:261-287.

Plagányi É. E. 2007. Models for an ecosystem approach to fisheries. FAO Fisheries Technical Paper. No. **477**. FAO. Rome. 108p.

Ploeg A. 2002. De Papoea. The Asia Pacific Journal of Anthropology **3**(1) 75 – 101.

Polovina J. J. 1984. Model of a coral reef ecosystem I. The Ecopath Model and its Application to French Frigate Shoals. Coral Reefs **3**(1):1-11.

Punt A. E., A. D. M. Smith. 2001. The gospel of maximum sustainable yield in fisheries management: birth, crucifixion and reincarnation. In: Reynolds J.D., G. M. Mace, K. H. Redford, and J.G. Robinson (editors) Conservation of exploited species. Conservation Biology **6**:41-66.

Raja Ampat Regency. 2007. Bersama membangun Raja Ampat (in Indonesia). Strategy, vision and mission statement of maritime Regency Raja Ampat. Available: Rotinsulu C. Gunung Arfak CI. Jl. 45. Sorong, Papua, Indonesia. email: chris@conservation.or.id.

Roberts C. M. 1995. Effects of fishing on the ecosystem structure of coral reefs. Conservation Biology **9**(5):988-995.

Roberts C. M. 1997. Ecological advice for the global fisher crisis. Trends in Ecology & Evolution **12**(1):35-38.

Roberts C. M. 2007. The unnatural history of the sea. Island Press, Washington DC. 435p.

Roberts C. M., G. Branch, R. H. Bustamante, J. C. Castilla, J. Dugan, B. S. Halpern, K. D. Lafferty, H. Leslie, J. Lubchenco, and D. McArdle. 2003. Application of ecological criteria in selecting marine reserves and developing reserve networks. Ecological Applications **13(**sp1):215-228.

Rose G. A. 2007. Cod: the ecological history of the North Atlantic fisheries. Breakwater Books, Newfoundland, Canada. 591p.

Rose G. A., D. W. Kulka. 1999. Hyperaggregation of fish and fisheries: how catch-per-unit-effort increased as the northern cod (*Gadus morhua*) declined. Canadian Journal of Fisheries and Aquatic Sciences **56**(S1):118-127.

Rosenberg A. A., W. J. Bolster, K. E. Alexander, W. B. Leavenworth, A. B. Cooper, and M. G. McKenzie. 2005. The history of ocean resources: modeling cod biomass using historical records. Frontiers in Ecology and the Environment **3**(2):78-84.

Ruckelshaus M., T. Klinger, N. Knowlton, and D. P. DeMaster. 2008. Marine ecosystem-based management in practice: scientific and governance challenges. Bioscience **58**(1):53-63.

Russ G. R., A. C. Alcala. 2003. Marine reserves: rates and patterns of recovery and decline of predatory fish, 1983-2000. Ecological Applications **13**(6):1553-1565.

Sala E., C. Boudouresque, and M. Harmelin-Vivien. 1998. Fishing, trophic cascades, and the structure of algal assemblages: evaluation of an old but untested paradigm. Oikos **82**(3):425-439.

Sale P. F. 1977. Maintenance of high diversity in coral reef fish communities. American Naturalist **111**:337-359.

Sale P. F., W. A. Douglas. 1984. Temporal variability in the community structure of fish on coral patch reefs and the relation of community structure to reef structure. Ecology **65**(2):409-422.

Salomon A. K., N. P. Waller, C. McIlhagga, R. L. Yung, and C. Walters. 2002. Modeling the trophic effects of marine protected area zoning policies: A case study. Aquatic Ecology **36**(1):85-95.

Schaefer M. 1954. Some aspects of the dynamics of populations important to the management of the commercial marine fisheries. Bulletin of Inter American Tropical Tuna Commission **1**(2):27-56.

Shanks A. L., B. A. Grantham, and M. H. Carr. 2003. Propagule dispersal distance and the size and spacing of marine reserves. Ecological Applications **13**(1) Supplement:S159-S169.

Sissenwine M. P. 1978. Is MSY an adequate foundation for optimum yield? Fisheries **3**(6):22-42.

Smith T. D. 1994. Scaling fisheries: the science of measuring the effects of fishing, 1855-1955. Cambridge University Press, Cambridge, UK. 392p.

Sparre P., S. C. Venema. 1998. Introduction to tropical fish stock assessment-Part I: Manual. FAO Fisheries Technical Paper. No. **306**/1. FAO, Rome. 375p.

Stewart R. R., I. Ball, and H. P. Possingham. 2007. The effect of incremental reserve design and changing reservation goals on the long-term efficiency of reserve systems. Conservation Biology **21**(2):346-354.

Sumaila U. R., J. Alder, and H. Keith. 2006. Global scope and economics of illegal fishing. Marine Policy **30**(6):696-703.

Surbakti S., R. L. Praptoprijoko, and S. Darmesto. 2000. Indonesia's 2000 Population Census: A Recent National Statistics Activity. BPS Statistics Indonesia. United Nations Economic and social commission for Asia and the Pacific UNESCAP. Bangkok.

Swearer S. E., J. E. Caselle, D. W. Lea, and R. R. Warner. 1999. Larval retention and recruitment in an island population of a coral-reef fish. Nature **402**(6763):799-802.

Szaro R. C., W. T. Sexton, and C. R. Malone. 1998. The emergence of ecosystem management as a tool for meeting people's needs and sustaining ecosystems. Landscape and Urban Planning **40**(1-3):1-7.

Tanda L. 2002. A basic stock assessment of economically important coral reef fishes of Raja Ampat Islands, Papua Province, Indonesia. In: McKenna, S.A., G.R. Allen, and S. Suryadi, editors. A marine rapid assessment of the Raja Ampat Islands, Papua Province, Indonesia. Conservation International, Washington, DC. RAP Bulletin of Biological Assessment **22**: 58-65.

Tesfamichael D., T. J. Pitcher. 2007. Estimating the unreported catch of Eritrean Red Sea fisheries. African Journal of Marine Science **29**(1):55-63.

Walters C., P. H. Pearse. 1996. Stock information requirements for quota management systems in commercial fisheries. Reviews in Fish Biology and Fisheries **6**(1):21-42.

Walters C. J., S. J. D. Martell. 2004. Fisheries ecology and management. Princeton University Press, New Jersey. 448p.

Walters C. J., D. Pauly, and V. Christensen. 1997. Structuring dynamic models of exploited ecosystems from trophic mass-balance assessments. Reviews in Fish Biology and Fisheries **7**:139-172.

Walters C. J., D. Pauly, and V. Christensen. 1998. Ecospace: prediction of mesoscale spatial patterns in trophic relationships of exploited ecosystems, with emphasis on the impacts of marine protected areas. Ecosystems **2**(6):539-554.

Walters C. J., D. Pauly, V. Christensen, and J. F. Kitchell. 2000. Representing density dependent consequences of life history strategies in aquatic ecosystems: Ecosim II. Ecosystems **3**:70-83.

Ward T., D. Tarte, E. Hegerl, and K. Short. 2002. Policy proposals and operational guidance for ecosystem-based management of marine capture fisheries. World Wide Fund for Nature, Sydney, Australia. 80p.

Willis T. J., R. B. Millar, R. C. Babcock, and N. Tolimieri. 2003. Burdens of evidence and the benefits of marine reserves: putting Descartes before des horse? Environmental Conservation **30**(02):97-103.

Worm B., R. Hilborn, J. K. Baum, T. A. Branch, J. S. Collie, C. Costello, M. J. Fogarty, E. A. Fulton, J. A. Hutchings, and S. Jennings. 2009. Rebuilding global fisheries. Science **325**(5940):578.

2 Illegal, Unreported and Unregulated Fisheries Catch in Raja Ampat Regency, Eastern Indonesia[8]

2.1 Introduction

2.1.1 The IUU problem in Indonesia

World-wide, illegal, unreported and unregulated (IUU) fishing prevents governments and resource managers from capturing the full economic rent from fisheries, and hampers the sustainable and ecologically responsible management of marine ecosystems (Pitcher et al. 2002; Agnew et al. 2008; Pitcher et al. 2009). Indonesia, with a reported catch of 4.7 million tonnes (average from 2002 to 2005) of fish and shellfish, is currently ranked as the world's sixth most important fishing nation (FAO 2008), but has a substantial problem with IUU catches in excess of those reported to government agencies and to FAO. As such, were the true estimates of catch to be considered, including large unreported extractions by both foreign and national vessels, and by both small scale fisheries and commercial fleets (Butcher 2002), Indonesia would probably rank higher in the list of top fishing nations (Pitcher et al. 2007).

As highlighted in a synthesis of fisheries management issues in Indonesia, the high prevalence of IUU fishing in Indonesia can in part be explained by Indonesia's inefficient fisheries' data collection systems (Willoughby et al. 1999; Mous et al. 2005). For example, in western Bali, fishers land only about 45% of the catch at official landing sites, despite the close proximity of government landing sites (Buchary E.[9] , pers. comm.). In the Arafura Sea, in Eastern Indonesia, Nurhakim et al. (2008) and Pitcher et

[8] A version of this chapter has been published. Varkey, D.A., Ainsworth, C.H., Pitcher, T.J., Goram, J. and Sumaila, R. Illegal, unreported and unregulated fisheries catch in Raja Ampat Regency, Eastern Indonesia. Marine Policy **34**: 228-236.

[9] Eny Buchary, Ph.D. University of British Columbia.

al. (2007) estimated IUU catches in excess of one million tonnes per year; chiefly due to a lack of the financial and human capacity necessary to monitor and maintain accurate records. To compound this, the Fishery Act no. 9/1985 and the Fishery Act no.31/2004 do not require subsistence or traditional fishing vessels (i.e., fishing fleets ≤ 5 gross tonnage (GT) or boats without engines or with engine size ≤ 15HP) to have fishing permits (BRKP 2005). As a result, small scale fishing, which accounts for a large proportion of all fishing activities in Indonesia, remains largely unreported (Buchary, pers. comm.).

2.1.2 Study area—Raja Ampat Archipelago

The Raja Ampat Archipelago extends over 45000 km^2 and includes 4 large islands (Waigeo, Batanta, Salawati and Misool) and around 600 small islands, located adjacent to the northwest tip of the province Papua in eastern Indonesia (Donnelly et al. 2003). The bulk of the catch is caught by small-scale fisheries operating in the reefs and inshore waters (Pitcher et al. 2007) using hook and line, traps, gillnets, lift nets and other methods. A total of 196 species, representing 59 genera and 19 families are classified as target species for reef fisheries in Raja Ampat (Tanda 2002).

In 2002, Law no.26 established the new Regency of Raja Ampat (Sumule and Donnelly 2003). A decree by the Bupati (Regent) in 2003 declared Raja Ampat a ‘Kabupaten Bahari’ (maritime regency) (Conservation International 2008) and consequently aroused interest in fisheries management issues. The aims of this chapter are:

i. to estimate the likely range of IUU catch using a semi-quantitative methodology,
ii. to reconstruct fisheries catches from the year 1960 to 1994, and
iii. to quantify the value of IUU catch from the year 2003 to 2006.

2.2 Methods

A ‘semi-quantitative method for Monte Carlo integration of historical sources’ (MRAG 2005) was applied; the method was originally developed by Pitcher and Watson (2000) to estimate IUU catches for Atlantic Canada. The history of regulatory, technological, political and market changes from 1960 to present were used to estimate trends in the relative rate of misreporting of fisheries catch. The trends were converted to absolute values using anchor points: known rates of misreporting from the literature from the region, and based on expert opinion. A Monte Carlo analysis was used to estimate the likely quantity of IUU catch with associated error ranges. Since its original publication, the methodology has since been further refined and widely used, for example, for fisheries in the North Atlantic (Forrest et al. 2001), Iceland and Morocco (Pitcher et al. 2002), British Columbia, Canada (Ainsworth and Pitcher 2005) and Eritrea (Tesfamichael and Pitcher 2007). The technique has two major advantages: (1) all available data on under-reporting, no matter what regulatory regime is in place, can be combined; and (2) uncertainty of estimates and trends can be addressed by applying a Monte Carlo simulation that uses likely error ranges (ICES 2005).

The level of misreporting was analyzed for fisheries targeting reef fish, important pelagics (tuna, anchovy, and shark) and commercial invertebrates (sea cucumber and lobster). After a survey, Erdmann and Pet (2002) reported that the reefs in Raja Ampat were widely impacted by blast fishing, and cyanide fishing and seemed to be a 'patchwork quilt of damaged and healthy areas'. The fishery for reef fish was hence divided into illegal catch using destructive methods and unreported catch of fish caught by other gears. Due to the difficulty in dividing up catches of the remaining groups into illegal, unregulated and unreported, a single 'unreported' catch category was used to combine the influence of unreported artisanal fisheries and unreported commercial fisheries (the latter including both catches by local fishers and catches by fishers from outside Raja Ampat).

2.2.1 Catch reconstruction

Fisheries catch records for the years from 1994 to 2005 for Raja Ampat were available from the Indonesian Department of Fisheries (Dinas Kelautan dan Perikanan—DKP) (DKP 2007). In 1960, a few hook and line and gleaning fishers operated from canoes in Raja Ampat. As anyone could catch fish by themselves for their own consumption, no local markets existed (Goram 2007). In 1962, the Dutch ceded control of West New Guinea to Indonesia (Cookson 2002), and in May 1963, Indonesia took over the administration of the region (Cookson 2002). From then on, Papua witnessed immigration (locally referred to as 'Indonesianisation') from other provinces of Indonesia (Goram 2007). The immigrant fishers began to catch and sell fish to Java and Sumatra. Papuans were also introduced to fishing with nets (Goram 2007). The increase in population by the influx of immigrants changed the exploitation pattern in Raja Ampat. Catch reconstruction from 1960 to 1993 is based on human population growth rates for 1960-1980 (2.7) from McNicoll (1982) and for 1980 to 1990 (3.41) from Surbakti et al. (2000). For the year 2006, a simple forecast of the catch, equal to the average of the years 2003-2005, was assumed.

2.2.2 Compilation of the influence table

An influence table is a chronological documentation of events in the regulatory, technological, political, and economic history of Raja Ampat considered to have influence on the IUU catch for each of the fisheries (i.e., caused an increase or decrease). A significant shift in the exploitation pattern in Raja Ampat occurred after 1960, after the Indonesian government took control over the region. Therefore 1960 was chosen as the starting point of the analysis. Individual events were referred to as 'influences' and assigned numerical 'IUU influence' ratings: (+1) when the influence led to an increase in IUU and (-1) when the influence caused a decline in IUU catches. Each of the influence ratings was weighted by a factor according to the strength of the change it caused to the resource use patterns in Raja Ampat. Six weighting factors were used. Major events at the 'National' (Indonesia), 'Provincial' (Papua) and 'Local' (Raja Ampat) levels were weighted by 1, 3 and 5, respectively. Minor events at the 'National', 'Provincial' and 'Local' level received ratings equal to 0.5, 1 and 2, respectively. Weightings were assigned based on: (i) the poor level of enforcement of existing regulations due to remoteness of Papua from Jakarta, the Indonesian capital city (Hill 1998), and (ii) the assumption that Papuans respond better to decentralized government (Wanandi 2002; Timmer 2005; Bailey 2007) . Most events in the timeline were obtained through interviews conducted by Yohanis Goram (The Nature Conservancy, Sorong, Papua) with local community members (Goram 2007). A list of 150 influences (presented in Appendix C) was considered for the timeline of Raja Ampat's fisheries. For each fishery, an influence trend was created by calculating a numerical running total of the weighted IUU influence ratings from 1960 to 2006, where influences likely to have increased IUU fishing were added to the cumulative score, while influences likely to have reduced IUU fishing were subtracted from the same. Figure 2.1 summarizes the influence trends for all fisheries considered.

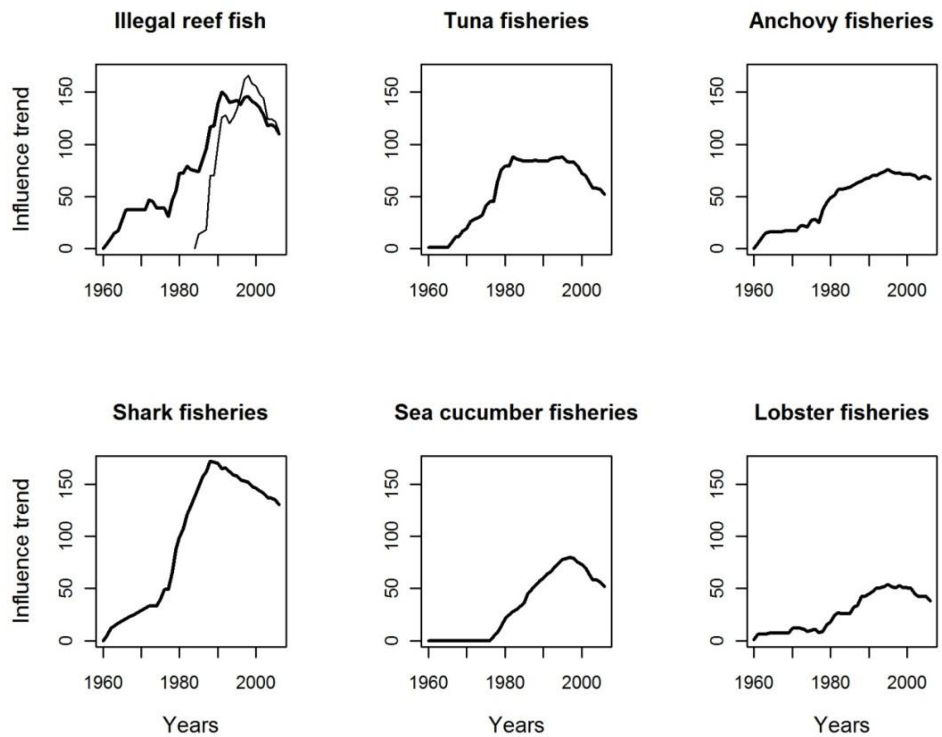

Figure 2.1 Influence trend.
The baseline year for the analysis is 1960, hence the influence trend for IUU starts at 'zero' for all fisheries except illegal reef fishery, for which the baseline year is 1984. The figure shows the cumulative numerical trend representing relative change in the rate of misreporting versus the time period 1960-2006.

2.2.3 Quantifying incentive

For each fishery, the numerical influence total was divided into five incentive categories: low, low/medium, medium, medium/high, and high. Figure 2.2 illustrates this for the unreported reef fish fishery. The period from 1980 to 2000 showed a sharp increase in unreported reef fish catch. After 2000, the trend reversed slightly.

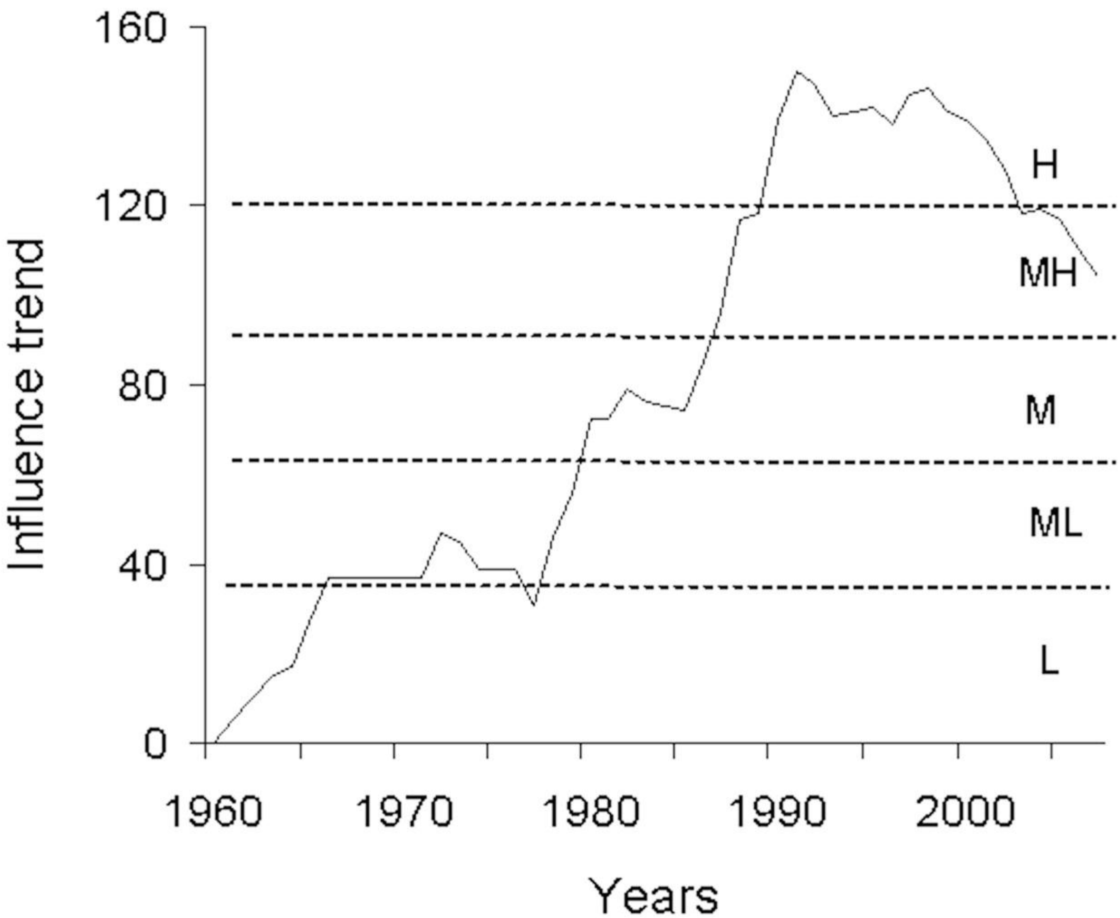

Figure 2.2 Quantifying incentive for unreported reef fish fishery.
The influence trend is divided into 5 categories: high (H), medium high (MH), medium (M), medium low (ML) and low (L)

2.2.4 Anchor points

The incentive categories were converted into actual catch estimates using anchor points. Anchor points as defined here were absolute estimates of fish catch derived from the literature or from survey information. The details of the anchor points are provided in Appendix C. For incentive categories where anchor points were not available, the absolute catch was obtained using a scaling factor. The scaling factor is based on a rule that the category 'medium–high' represents 80% of the upper cumulative influence total, ‘medium’ 60%, ‘medium–low’ 40%, and ‘low’ 20%. Table 2.1 indicates the absolute range of IUU catch rates for incentive categories of this study’s selected fisheries.

Table 2.1 Absolute estimates for IUU catch ranges.
The values in bold are anchor points from literature. The other estimates were calculated using the scaling factor

Influence level	Range	Illegal reef fish	Unreported reef fish	Tuna	Anchovy	Shark	Sea cucumber	Lobster
H	high	**49.06**	**75.00**	61.37	**90.35**	48.33	**59.87**	**48.25**
	low	61.32	**90.00**	**76.71**	**93.78**	64.20	**97.38**	**68.79**
MH	high	**36.62**	54.00	**54.95**	72.28	**38.67**	47.90	19.35
	low	**49.06**	**75.00**	61.37	**90.35**	**51.36**	**59.87**	55.03
M	high	24.53	36.00	30.68	54.21	29.00	35.92	14.51
	low	36.62	54.00	54.95	72.28	38.67	47.90	41.27
ML	high	12.26	18.00	15.34	36.14	19.33	23.95	9.67
	low	24.53	36.00	30.68	54.21	29.00	35.92	27.52
L	high	9.81	0.00	0.00	18.07	9.67	11.97	4.84
	low	12.26	18.00	15.34	36.14	19.33	23.95	13.76

2.2.5 *Addressing uncertainty*

The anchor points provided the range of IUU catch level for each incentive category: low, low/medium, medium, medium/high and high. A Monte Carlo technique was employed to estimate the mean of missing catch with error for each year. The true amount of missing catch (*X*) would fall somewhere in the estimated range between the lower bound (*A*) and the upper bound (*C*) so that,

$$P[A \le X \le C] = \int_a^c f(X)dX = 1$$

For values of *X* between *A* and *C*, the probability density function *f* (*X*) of the triangular distribution is then given by:

$$f(X) = \begin{cases} \dfrac{2(X-A)}{(C-A)(B-A)} & \text{if } A \le X \le B \\ \\ \dfrac{2(C-X)}{(C-A)(C-B)} & \text{if } B \le X \le C \end{cases}$$

B is the 'best guess'—the mode of the distribution. Figure 2.3 shows the empirical probability distribution. Sampling 5000 times, the Monte Carlo routine empirically determines the mean and 95% confidence intervals. For most of the fisheries, a symmetrical error distribution was assumed, with the most likely missing catch value (the mode) equidistant between maximum and minimum estimates. However, an asymmetric distribution was used for unreported reef fish, in which the mode was represented using a 'best guess' estimate that was shifted to the left of the median value. The asymmetric distribution assumes that the unreported catch might be overestimated by a small amount, but the catch could be potentially underestimated by a large amount.

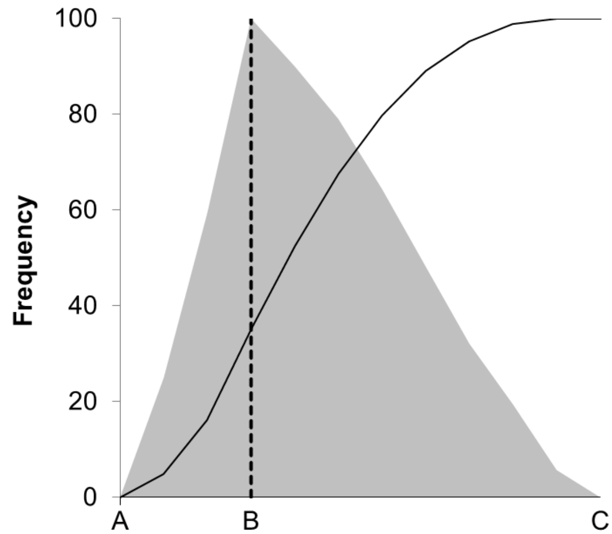

Figure 2.3 Example distribution of the error assumption.
Triangular distribution provided for example. (A) lower bound, (B) 'best guess', and (C) upper bound. Cumulative probability distribution of missing catch (line).

2.2.6 Quantifying IUU catch revenues in Raja Ampat Regency 2003-2006

The revenue generated from IUU fishing was split into 2 components: (1) revenue from the illegal fishery of reef fish; and (2) revenue from unreported fisheries. 2003 was chosen as the base year for the economic analysis as Raja Ampat started operating as a new regency in 2003 with semi-autonomous government. Fish and shellfish prices were obtained from survey data for the years 2003 (Farid and Anggraeni 2003) and 2006 (Dohar and Anggraeni 2007). Prices were not available for years 2004 and 2005. The consumer price index (CPI) (OECD 2007) for Indonesia, an index used to measure the general rate of inflation (Diewert 2001), was used to convert the nominal 2006 price to real price in 2003 (measured in 2003 US dollars). Prices for years 2004 and 2005 were calculated by interpolating the real price difference between 2003 and 2006 (Diewert 2001).

2.3 Results

2.3.1 Catch reconstruction and IUU catch estimation

The absolute unreported catch for fisheries operating in Raja Ampat was calculated using the reconstructed catch for the years 1960-2006. Aggregating the results for year 2006 for the reef fish fishery showed that only about 26% of the catch was reported, 20% was caught illegally. Of pelagic species' catches, about 43%, 93% and 44% of the catch for tuna, anchovy and shark catches were unreported, respectively. For invertebrates, 42% of the sea cucumber catches and 37% of the lobster catches were unreported. The amount of unreported catch in tonnes in 2006 and the errors associated with individual estimates are shown in Table 2.2. Figure 2.4 shows the reconstructed catches and the trend of reported and unreported catches over the time period 1960-2006.

Table 2.2 IUU catch in thousand tonnes in 2006 and the error on the estimates.

Catch and error estimate	Illegal reef fish	Other reef fish	Tuna	Anchovy	Shark	Sea cucumber	Lobster
Reported catch ('000 tonnes)	4.054	4.054	17.626	1.321	0.598	0.017	0.630
Unreported catch ('000 tonnes)	3.043	8.205	13.233	15.339	0.460	0.012	0.371
Error% (-ve)	18.6	34.3	33.2	15.5	14.4	17.5	48.4
Error% (+ve)	21.7	38.2	45.7	21.2	16.3	21.2	75.4

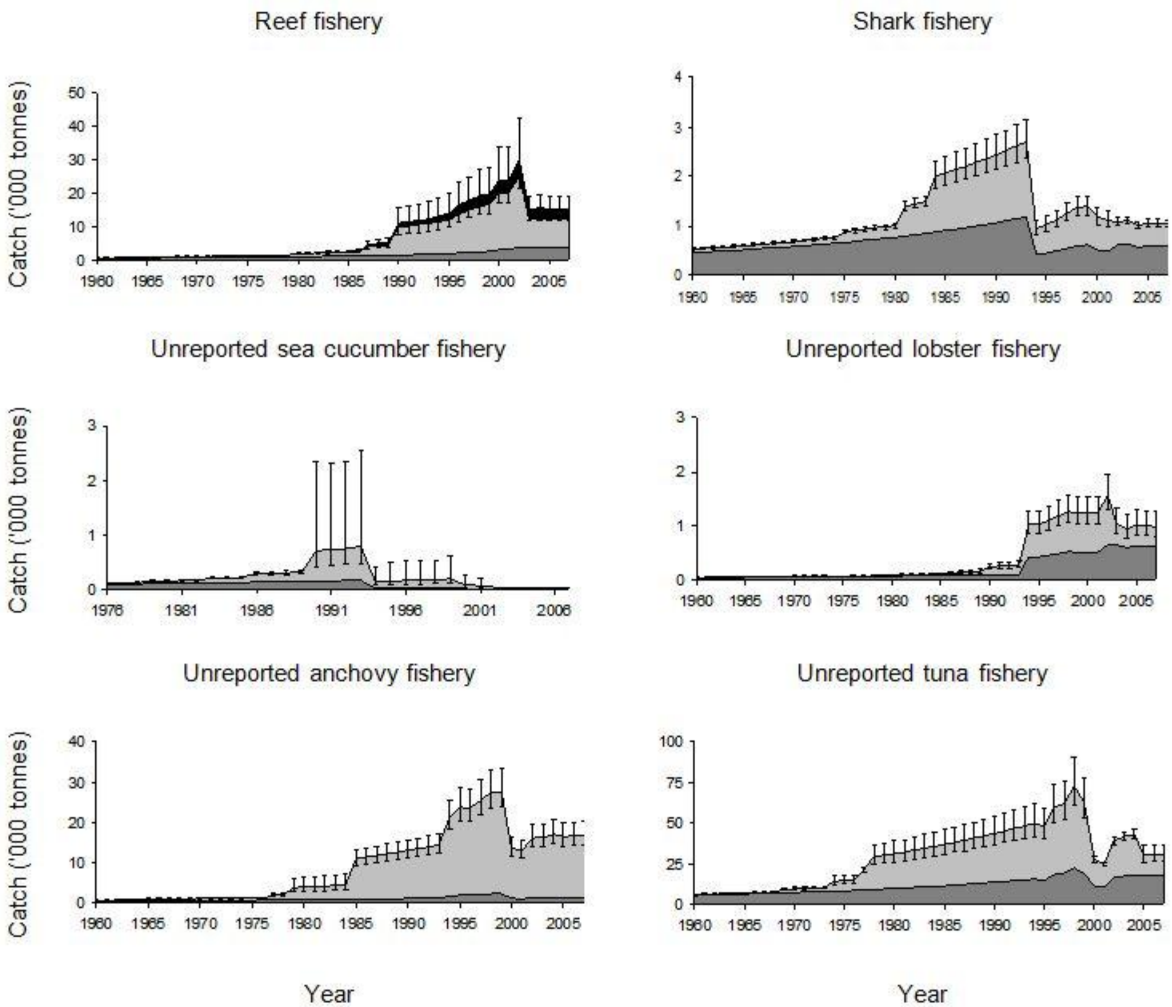

Figure 2.4 Reported and unreported catches in Raja Ampat.
The dark grey is the reported catch; the catches prior to 1990 are the results of reconstruction. The light grey is the unreported catch. In the first graph the black area represents illegal fishery for reef fishes.

2.3.2 Quantifying the IUU catch revenues in Raja Ampat Regency 2003-2006

Results from a comparison of total revenue generated in the period 2003-2006 from reported versus illegal and unreported catch are shown in Figure 2.5. The error associated with individual estimates is not shown in the graph but is included in Table 2.3. The results show that over the four year period, revenue from the IUU catch in Raja Ampat totalled 160 million US dollars (in 2003 USD), or an average of 40 million USD a year.

Table 2.3 Total revenue from IUU fishing for 2003-2006 and error on the estimates

Value of catch and error	Illegal Reef fish	Other Reef fish	Tuna	Anchovy	Shark	Sea Cucumber	Lobster
Catch value (million USD)	42.4	16.1	50.4	18.5	7.6	0.1	25.2
Error% (-ve)	18.3	34.5	19.0	15.5	14.3	17.2	48.5
Error% (+ve)	21.7	38.1	24.6	21.8	16.3	20.9	76.1

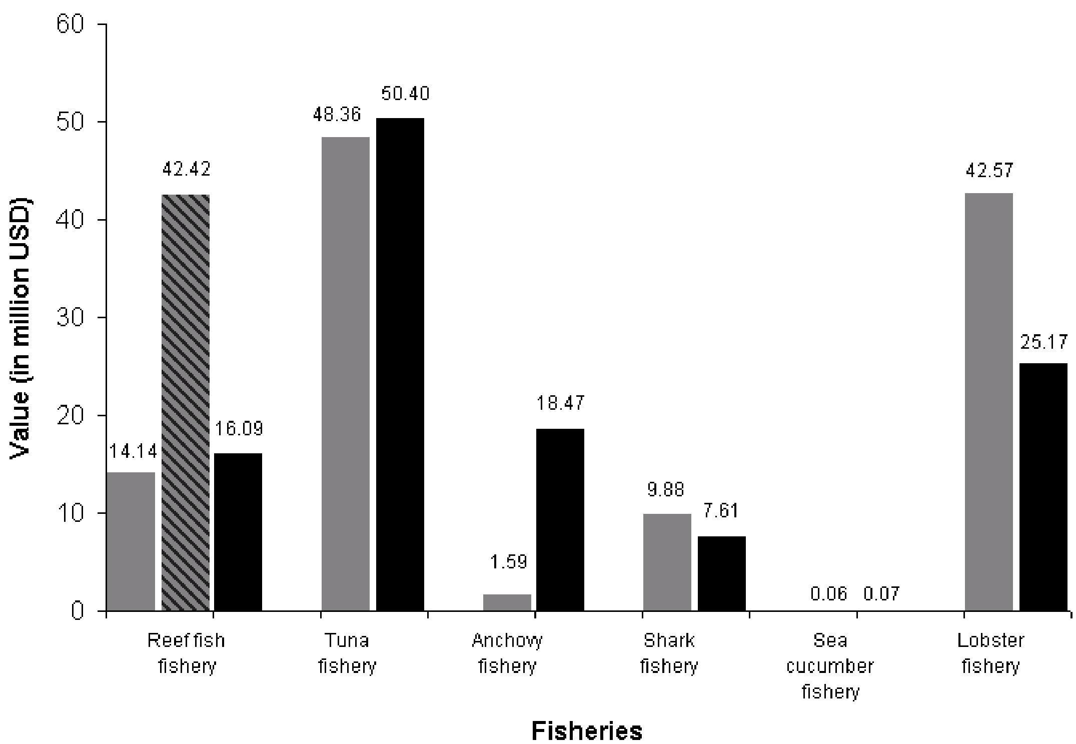

Figure 2.5 Total revenue from IUU fishing in Raja Ampat (2003-2006).
The light grey bars are the revenue generated from reported catches, the black bars are the revenue from unreported catches. The shaded bar in category reef fish is the revenue from illegal fishing.

2.4 Discussion

2.4.1 Catch reconstruction

Since 1963, Raja Ampat and other areas in Papua have experienced an increasing influence from the central government: new schools have been established and new development projects undertaken and there has been an influx of administrators, businessmen, and security forces from the other provinces of Indonesia (Goram 2007). The population influx and technological advances led to a shift away from a predominantly subsistence based lifestyle to one based on cash crops and extractive industries such as mining, logging (WWF/IUCN 1996) and fishing for commercial purposes (Palomares and Heymans 2006). Important factors that contributed to the change in marine exploitation patterns were the significant surge in population size and demand for sea cucumber, pearls and sea turtles from the Raja Ampat Archipelago (Palomares and Heymans 2006).

2.4.2 Estimation of IUU fishing in Raja Ampat

2.4.2.1 Reef-fish fisheries

For the purpose of clarity, the results for the reef-fish fisheries are described under two categories: the illegal catch and the unreported catch of reef fish.

<u>Illegal fishery</u>

Cyanide fishing began in a limited capacity by fishers from outside Raja Ampat in the early 1980s and became very popular by the mid-1980s (Goram 2007). Indonesia began exporting live reef fish to Hong Kong in 1988 (Chan 2000a). The expanding market for live reef fish fuelled over-exploitation. By the mid-1990s, Indonesia accounted for about half of the live fish supply in the markets of Hong Kong and Singapore (Johannes and Riepen 1995). By the late 1990s, fishers in several parts of Indonesia were experiencing a decline in target fish in shallow waters; cyanide fishermen reported declines in catch per unit effort of up to 90% in the latter half of the 1990s (Chan 2000b). Similar to the trend experienced in the other parts of Indonesia, fishers in Raja Ampat also experienced declines in large groupers and Napoleon wrasse (*Cheilinus undulates*) (Goram 2007). However, hope of better catches in Eastern Indonesia caused further influx of more fishers into Raja Ampat and in the early 2000s, mouse grouper (*Cromileptes altivelis*) and Napoleon wrasse had also become scarce in Raja Ampat.

Blast fishing in Raja Ampat was introduced by fishers from Buton, Sulawesi and Biak (an island located in Cenderawasih Bay close to the northern coast of Papua). It started on Crocodile Island (a small island close to Sorong) in the late 1980s (Kadarusman unpublished document). Fishers in Raja Ampat, especially the younger generation were encouraged to adopt destructive fishing methods when they observed the high profits made by fishers from outside Raja Ampat (Goram 2007). This shift that happened in late 1980s was locally recognized as a viable option because of increased competition from Sulawesi fishers fishing for marine invertebrates (Goram 2007). The associated 'macho' status and favorable response from the opposite sex was a bonus. It was not difficult to adapt to this fishing method because fishers from Sulawesi were supplying bombing material in Sorong (Goram 2007; Kadarusman unpublished document). Villagers who chose to participate in the cyanide fishery were supplied with boats and all necessary equipment (Sumule and Donnelly 2003). Live fish buyers from Sorong offered fishers a large down payment in return for sole purchasing rights to the fishermen's catch. The 'exclusive buyer' often forced fishers to fish heavily every day in order to clear his debt (Sumule and Donnelly 2003).

The rise in the number of fishers engaged in blast fishing (Goram 2007), and the supply of bombing material from Sulawesi (Kadarusman unpublished document) and East Java (Goram 2007) caused widespread destruction of reefs around Raja Ampat. Villagers in Waigeo stated that they heard blasts almost daily (Bailey, M.[10] pers. comm.)—a situation corroborated by villagers in Kofiau. The actual level of blast fishing remains difficult to quantify as recent aerial surveys failed to observe any active operations (Ainsworth et al. 2008).

[10] Megan Bailey, University of British Columbia

Illegal fish catches in Raja Ampat peaked in the late 1990s and early 2000s (Goram 2007) (as can be observed in Figure 2.4). Live fish transport vessels from Hong Kong periodically collected fish from major *karambas* (floating net cages for holding live fish); Indonesian military personnel were usually on board the transport vessels hinting at the 'collusion between the outside syndicates and military officials'(Sumule and Donnelly 2003). Willoughby et al. (1999) recognized the difficulty in controlling the trans-shipment of large numbers of illegally caught and unreported fish. Fishers from outside Raja Ampat trans-shipped their catch to the ships from Hong Kong or landed at unofficial landing sites outside Raja Ampat (Suebo, A.[11] pers. comm). Hence, chances for illegal catch being recorded in the official Indonesian catch statistics were minimal.

In the early 2000s, as large reef fish were getting scarce, fishers began targeting small reef fish to supply the ornamental fish trade (Goram 2007). The relative scarcity of large reef predators, particularly the absence of males, was also recorded during resource evaluation assessment of coral reefs in Raja Ampat (Donnelly et al. 2003) in 2002. Scarcity of breeding males is recognized to undermine the viability of spawning aggregations (Donnelly et al. 2003). Overfishing has been previously implicated in the disappearance of spawning aggregations (Colin 1992; Aguilar-Perera and Aguilar-Dávila 1996; Domeier and Colin 1997; Johannes et al. 1999).

Several conservation minded non-governmental organizations (NGOs) arrived in Raja Ampat in the period 2000-2005; since then, they have considerably increased public awareness of the destructive effects of cyanide and blast fishing. In fact, many Raja Ampat fishers have stopped blast and cyanide fishing and shifted to longlines and gillnets as a result of awareness campaigns launched by the Nature Conservancy (Pastor Mambrasar and Pastor Katutun[12] pers. comm.). Fishers reported higher catches in pelagic species following the self-imposed ban on use of destructive fishing methods (Pastor Katutun pers. comm.). Today, local fishers who engage in blast fishing are despised by village chiefs and local elders (Pastor Mambrasar pers. comm.). Homilies at local churches in rural communities such as Kofiau are pro-conservation in their message and

[11] Anton Suebo, The Nature Conservancy, Bali, Indonesia
[12] Pastor Membrasar and Pastor Katutun, Religious priests, Raja Ampat, Indonesia

this has inspired villagers to support the implementation of large (>4700 km²) marine protected areas (MPAs) where local fishermen and the Raja Ampat Regency government might be able to prevent access by destructive fishers from other parts of Indonesia. E.g., the villagers celebrated with enthusiasm the setting up of an MPA in Kofiau Island (Ainsworth and Varkey 2007).

Unreported reef fish fishery

The hook and line fishery is the most important fishery for reef fish in Raja Ampat (Ainsworth et al. 2007). The high quality live fish are sold to fishers owning *karambas*, who sell it to local live reef fish traders or to ships from Hong Kong (Donnelly et al. 2003). The remainder of reef fish catch is sold in the local market. The landing center in Sorong is more than 100 miles away from Kofiau and Misool Islands in Raja Ampat. The price of fish is higher in Sorong market (Rotinsulu[13], pers. comm.), but the incentive is not worth the cost in terms of fuel, time and travel. Misool island has larger number of villages (20), compared to 3 villages in Kofiau (Djuang and Imbir 2007). The catch has good demand from employees of pearl farms in Misool. A large number of residents of Misool Island are descendants of immigrants from Sulawesi, and still maintain strong ties. The catch landed in Misool is often dried and traded in Sulawesi markets (Suebo, A. pers. comm.). To the east of Misool Island, on the west coast of mainland Papua, is the city Seram (not part of Raja Ampat Regency). Fishers often obtain fuel for their boats in Seram and trade their fisheries catch there (Suebo A. pers. comm.). As these smaller landing sites do not keep records, a large proportion of the overall catch does not enter the official Indonesian statistics. The Raja Ampat Regency needs better methods to quantify the catches from the islands which are far from Sorong.

2.4.2.2 Tuna fisheries

In 1967-1969, two boats (Injeros and Cakalang) operated by local government company (PD. Irian Bakti) began fishing for tuna under the control of Fisheries Department in Sorong (Goram 2007). In 1973-1975, two companies—Usaha Mina and PT. Alfa

[13] Chris Rotinsulu, Conservation International, Raja Ampat, Indonesia

Kurnia—started fishing for tuna in Raja Ampat waters (Goram 2007). Up to 80 boats of 50 tonnage capacity operated in this time. Another tuna company PT. Ramoi started its operation in 1982-1984; the peak period of tuna fishing was in 1994-1996 (Goram 2007)

The Fisheries Department (DKP) reported that in 2005 the commercial catch of tuna from Raja Ampat was approximately 369 tonnes. However, information collected during the Conservation International valuation study (Dohar and Anggraeni 2007) showed that the catches from two companies (PT. Radios Apirja Sorong and KUD Tuna Cakalang Tunas Jaya) fishing in Raja Ampat and adjacent waters alone exceeded the reported catch from Raja Ampat (819.16 tonnes). There was anonymous information that the tuna industries based in Sorong severely under-reported their tuna catch. In spite of considerable effort (Rotinsulu pers. comm.) no data could be collected on the true tuna catches. Given the financial incentive to under-report (reduced taxes), the secrecy raises concerns over the true levels of under-reporting. Willoughby et al. (1999) reported a similar situation with tuna catch reports; he stated that total tuna declarations were probably little more than half the actual catches. This chapter explores only the level of unreported fishing for tuna in Raja Ampat, estimating the illegal fishing for tuna by foreign vessels will raise the estimate of tuna catches higher.

2.4.2.3 Anchovy fisheries

An anchovy fishery in Kaboei bay fishery on Waigeo Island has been recognized as being feasible since 1954 because catches up to 1 tonne per hour in shallow coastal areas could be achieved (Palomares and Heymans 2006). An artisanal lift net raft provided with a small shelter '*bagan*', is most commonly used for anchovy fisheries in Raja Ampat. The Waigeo fishery began in 1973-1975 by migrant fishers from South Sulawesi (Goram 2007). Bailey et al. (2008) estimated of 49-76 tonnes annual catch of anchovy per *bagan.* The total number of *bagans* fishing in Raja Ampat was based on sightings during an aerial survey of Raja Ampat (Barmawi 2006). The migrant anchovy fishers are not required to report their catch, and it is trans-shipped at sea to Java, Western Indonesia. The migrant fishers earn almost twice as much as an average fisherman from Raja Ampat (Bailey et al. 2008). Monitoring the migrant anchovy fishery is an important consideration for the fisheries management program in Raja Ampat Regency (Bailey et al. 2008).

2.4.2.4 Shark fisheries

Fishing for shark fin became very popular from 1976 to 1981. Local fishers often found bodies of shark with fins cut off in the coastal areas. Nets more than 2 km long were used by fishers from Madura, East Java; Selayar and Buton, South Sulawesi. But by 1990-1993 fishers had started experiencing difficulty in locating shark (Goram 2007). Farid and Anggraeni (2003) reported that in 2000-2002 shark fin collectors in Sorong gave 8 to 10 million Rp. per trip to the fishers from outside Raja Ampat to catch sharks, equivalent to approximately $900 to $1000 USD (1USD~9000 Indonesian Rupiah). Allen (2003) attributed the ‘paucity’ of reef sharks in Raja Ampat to the shark fin trade. In 2002, there were reports that villages in Kapadiri, Waigeo Island cooperated with fishing companies from the Philippines. The companies paid an access fee of Rp 500,000 (=~55$USD) to the village and provided fishers with generators and outboard motors. Depending on the quality of the shark fin, the company paid the fishers Rp 1,800,000 to Rp 3,000,000 (200-300 $USD) per kilogram of dried shark fin. The fishers landed carcasses of small and medium-sized sharks for consumption, but the larger sharks were discarded (Donnelly et al. 2003). More than 100 boats (about 7m long) from Halmahera (islands west of Raja Ampat) currently fish for sharks in Raja Ampat. The shark fin catch is trans-shipped to Halmahera or Makassar and then to Japan. All the fishers on the vessels are Indonesian; however, the investment for the vessels comes from outside Indonesia (Suebo, A. pers. comm.).

The local fishers and live aboard operators suggest there has been a great decline in the shark population in Raja Ampat (Djuang[14] pers. comm.). In recent years, the number of shark fishers have decreased; however, the fishers who remained in business have started targeting manta ray aggregations near Wayag and Sayang Islands (two islands Southwest of Waigeo Island) in Raja Ampat (Suebo, A. pers. comm.). The price data (Farid and

[14] Jacinta Djuang, Conservation International, Raja Ampat, Indonesia

Anggraeni 2003; Dohar and Anggraeni 2007) showed a dip in the prices of shark fin from 2003 to 2006. The probable reason for the dip is that the prices reported in 2003 were for only 2 species (black shark *Charcarhinus melanopterus* and lontar sharks *Isurus glaucus*) and those species have declined considerably over the years. The price reported in 2006 is for an assorted group of shark fins.

2.4.2.5 Sea cucumber fisheries

Though the total revenue from sea cucumber fisheries is low, a large number of fishers in Raja Ampat engage in gleaning to catch them. Sea cucumber fetches a very high price in the market, but the catch is small compared to the other fisheries analyzed in this chapter. Between 1928 and 1933 both *Trochus* shells and sea cucumber were exported from Sorong and Misool (Palomares and Heymans 2006). By 1934-1935 the export of sea cucumber from the territory of Papua was 40 tonnes per year. Sea cucumber remained an important export item in 1954 (Palomares and Heymans 2006). Commercial extraction of invertebrates, e.g., mollusc shells and sea cucumber, continued in spite of signs of overexploitation (Palomares and Heymans 2006; Palomares et al. 2007). Fishers from South Sulawesi began fishing for sea cucumber in 'large scale' in Raja Ampat in 1978 and fishers began observing a decline in the sea cucumber population in late 1990s (Goram 2007).

2.4.2.6 Lobster fisheries

A resource use survey in 2007 (Muljadi unpublished data) observed lobster catch on 5 out of 10 vessels from Sulawesi. All these vessels operated with inboard engines and fished using compressors. Lobsters are an important catch for local fishers; crustacean catches account for about 13% of total catch and is mainly composed of lobsters and shrimp (Muljadi 2004). In 2003, the price for lobster ranged from Rp 35000 (4 $USD)/kg (for baby size 0.2–0.5 kg) to Rp 115000 (13$USD)/kg (for super-size 0.8–1.2 kg) (Farid and Anggraeni 2003). The super-size was caught by fishers who used compressors for diving. In 2006, the average price for lobster in Sorong market declined to Rp 32,500 (3.5 $USD)/kg (Dohar and Anggraeni 2007). The decline is probably due to the fact that the average size of the lobster has declined over the years due to overexploitation. Since the majority of the fishers in Raja Ampat do not use compressors, they are not able to catch big lobsters and fetch higher prices. Fishers from outside Raja Ampat who have motorized vessels and use compressors would fetch higher price than fishers in Raja Ampat, but their catch is not landed in Sorong.

2.4.3 Quantifying the economics of IUU catch in Raja Ampat 2003-2006

In Raja Ampat Regency, sustainability of marine resources is important to the economy and food security. Throughout the year, most Regency inhabitants are involved in subsistence fishing, even though they may be employed in other industries as their main revenue sources (farming, construction, pearl farming, etc.) (Bailey et al. 2008). The estimated revenue generated by illegal fishing of reef fish is almost equal to the revenue from all reef fish catch in Raja Ampat (reported and unreported combined). Until the late 1990s, almost 90% of the grouper and Napoleon wrasse are caught by fishers from outside Raja Ampat while the local fishers caught about 90% of the other reef fishes (Erdmann, M.[15] pers. comm.). The grouper and Napolean wrasse fetch very high price in the market (~50 000 Rp/kg = USD 5.5/kg) compared to other reef fish (~7000 Rp/kg = <1 USD/kg). The resource is hence being exploited but with little gain to the local fishers. There was anecdotal information of serious under-reporting by tuna companies: a conservative estimate of $50 million USD over the four year period 2003-2006 was arrived at. At a tax rate of 2.5%, the government likely lost revenues of over $1 million USD in 4 years. Fishers and fishing companies fishing for anchovy and shark paid the villages a one-time small access fee to fish in their waters (Donnelly et al. 2003; Bailey et al. 2008). The revenue generated from unreported anchovy and shark fishing in the 4 year period 2003-2006 was over $18 million and $7 million USD, respectively. Centralization of the access system to Raja Ampat waters could turn these fisheries into a profitable enterprise for the regency. The hook and line fishery is the most important fishery for reef fish in Raja Ampat (Ainsworth et al. 2007) and the unreported reef fish fishery accounts for about $16 million USD. Most of the fisheries are small scale and do not contribute to government revenue in the form of taxes. However, the amount of catch and revenues is

[15] Dr. Mark Erdmann, Conservation International, Bali, Indonesia

indicative of the economic status of the average Raja Ampat fisher and serves as a guide for deciding the trade-off between the monitoring expenditure and expected revenue.

2.5 Conclusion

The marine species diversity in Raja Ampat is one of the highest in the 'coral triangle' (McKenna et al. 2002). However, Diamond (1986) noted in that marine resources in Raja Ampat were probably overfished. Subsistence or traditional fishing vessels are not required to have fishing permits (BRKP 2005); this is the reason why overexploitation and under-reporting by small scale vessels has received little attention compared to illegal fishing by foreign vessels in Indonesian waters. The indigenous people of Raja Ampat are rapidly being integrated into the cash economy and moving away from subsistence to commercial exploitation (Sumule and Donnelly 2003). After being declared as a 'Maritime Regency', it is the mandate of the regency to improve the marine management system. To this end, the fisheries department in Raja Ampat is conducting an inventory of the fishing vessels operating in Raja Ampat (Rahwarin, B. pers. comm.). Seven protected areas of total size of 4700 km^{-2} were declared in Raja Ampat in 2006 (Rabu 2006).

For better reporting of the fisheries in Raja Ampat, it is necessary to setup catch recording booths in the major fishing villages that would report to the Raja Ampat Regency fisheries department office. The true extraction of fish and shellfish from the coral reefs is essential to plan for future management policies, for example—control of access in Raja Ampat waters, improvement of data collection mechanisms, control of illegal fisheries. The fishers in Raja Ampat had traditional marine tenure which declined in importance after integration into Indonesia. It was disillusionment over unregulated access and the belief that their paradise was being plundered that led young fishers to engage in destructive fishing (Halim et al. 2007). Estimates of the true catch and its effect on local ecosystems and economies should encourage a restructuring of marine management. This may lead Raja Ampat on a path towards sustainable resource exploitation.

2.6 References

Agnew D., J. Pearce, T. Peatman, T. J. Pitcher, and G. Pramod. 2008. The global extent of illegal fishing. Marine Resources Assessment Group. London, UK. Available: < www.mrag.co.uk/Documents/ExtentGlobalIllegalFishing.pdf > [Accessed on February 12, 2009]

Aguilar-Perera A., W. Aguilar-Dávila. 1996. A spawning aggregation of Nassau grouper *Epinephelus striatus* (Pisces: Serranidae) in the Mexican Caribbean. Environmental Biology of Fishes **45**(4):351-361.

Ainsworth C. H., D. A. Varkey. 2007. Collaborative ecosystem-based management project bears fruit in Indonesia. Fishbytes **13**(6): 1-2.

Ainsworth C. H., T. J. Pitcher. 2005. Estimating illegal, unreported and unregulated catch in British Columbia's marine fisheries. Fisheries Research 75:40– 55.

Ainsworth C. H., D. A. Varkey, and T. J. Pitcher. 2007. Ecosystem simulation models for the Bird's Head Seascape, Papua. In: Pitcher, T.J., C.H. Ainsworth, and M. Bailey, editors, Ecological and economic analyses of marine ecosystems in the Birds Head Seascape, Papua, Indonesia. Fisheries Centre Research Reports **15**(5):6–172.

Ainsworth C. H., D. A. Varkey, and T. J. Pitcher. 2008. Ecosystem simulations supporting ecosystem-based fisheries management in the Coral Triangle, Indonesia. Ecological Modelling **214**(2-4):361-374.

Allen G. R. 2003. Coral reef fishes of Raja Ampat. In: Donnelly R., D. Neville, and P.J. Mous, editors, Final draft Report on a rapid ecological assessment of the Raja Ampat Islands, Papua, Eastern Indonesia. The Nature Conservancy Coral Triangle Center, Indonesia. Final Draft: 42-58.

Bailey M. 2007. Economic analysis of unregulated and illegal fishing in Raja Ampat, Indonesia. Master thesis submitted to University of British Columbia, Canada.

Bailey M., C. Rotinsulu, and U. R. Sumaila. 2008. The migrant anchovy fishery in Kabui Bay, Raja Ampat, Indonesia: Catch, profitability, and income distribution. Marine Policy **32**(3):483-488.

Barmawi M. 2006. Aerial survey in Raja Ampat: An observation of fisheries activity, large marine fauna, and coastal habitat (January 9-13, 2006). Data Report. The Nature Conservancy Coral Triangle Center. Jl. Pengembak no. 2, Sanur, Denpasar. Available: joanne_wilson@tnc.org.

BRKP. 2005. Fishing vessel registration and licensing system in Indonesia. Badan Riset Kelautan dan Perikanan (Research Body of Marine Affairs and Fisheries) Directorate of Fisheries Services and Enterprises. Directorate General of Capture Fisheries. Paper presented at the National Workshop on Illegal Unreported and Unregulated (IUU) Fishing in Indonesia. Jakarta, Indonesia. Funded by the Centre for Maritime Policy. University of Wollongong. Research Centre for capture fisheries. Agency for Marine and Fisheries Research Department of Marine Affairs and Fisheries. Organized by Australian Centre for International Agricultural Research. 28 April 2005, 12p.

Butcher J. G. 2002. Getting into trouble: the diaspora of Thai trawlers, 1965-2002. International journal of maritime history **14**(2):85-122.

Chan P.S.W 2000a. The industry perspective: wholesale and retail marketing aspects of the Hong Kong Live Reef Food Fish Trade. SPC Live Reef Fish Information Bulletin 7:3-7.

Chan P. S. W. 2000b. Current status of the live reef fish trade based in Hong Kong. SPC Live Reef Fish Information Bulletin **7**:8-9.

Colin P. L. 1992. Reproduction of the Nassau grouper, *Epinephelus striatus* (Pisces: Serranidae) and its relationship to environmental conditions. Environmental Biology of Fishes **34**(4):357-377.

Conservation International. 2008. Environmental Friendly Development in Raja Ampat. Available: <http://www.conservation.or.id/home.php?catid=30&tcatid=178&page=g_peluang.detail> [Accessed on July 30, 2008].

Cookson M. 2002. History of Papua – chronology of Papua from 1945 to 2002. Papuaweb Research Project. Indonesia: Papuaweb. Available: <www.papuaweb.org/chrono/files/kp.html> [Accessed on June 22, 2008].

Diamond J. 1986. The design of nature reserve system for Indonesian New Guinea. In: Soule ME. Conservation Biology: the science of scarcity and diversity. Sinauer and Associates, Massachussetts. p. 485-503

Diewert W. E. 2001. The consumer price index and index number purpose. Journal of Economic and Social Measurement **27**(3):167-248.

DKP. 2007. Laporan tahun 1994, 1998, 1999, 2002. Annual Reports – 1994.; 1998.; 1999. Departemen Kelautan dan Perikanan. Department of Marine Affairs and Fisheries. Sorong, Indonesia [in Indonesian].

Dohar A. G., D. Anggraeni. 2007. Valuasi Ekonomi Sumberdaya Alam di Kepulauan Raja Ampat. Conservation International, Bekerjasama dengan, Universitas Negeri Papua. (in Indonesian) Economic valuation report for Raja Ampat, Indonesia; Conservation International Indonesia.Available: Rotinsulu C. Gunung Arfak CI. Jl. 45. Sorong, Papua, Indonesia. chris@conservation.or.id.

Domeier M. L., P. L. Colin. 1997. Tropical reef fish spawning aggregations: Defined and reviewed. Bulletin of Marine Science **60**(3):698-726.

Donnelly R., D. Neville, and P. J. Mous. 2003. Report on a rapid ecological assessment of the Raja Ampat Islands, Papua, Eastern Indonesia. The Nature Conservancy Coral Triangle Center. Indonesia. Final Draft. 250p.

Erdmann M. V., J. S. Pet. 2002. A rapid marine survey of the northern Raja Ampat islands. Henry Foundation/The Nature Conservancy/NRM/EPIQ. Indonesia: The Nature Conservancy Coral Triangle Center. Available: <www.coraltriangle.org/downloads/Raja4%20trip%20report.pdf> [Accessed on June 14, 2007].

FAO. 2008. FAO Statistical Yearbook. FAO, Rome. Report number **2**.

Farid M., D. Anggraeni. 2003. Preliminary assessment of community-based natural resource management and conservation options in South Waigeo. Indonesia: Conservation International Indonesia.

Forrest R., T. J. Pitcher, R. Watson, H. Valtysson, and S. Guenette. 2001. Estimating illegal and unreported catches from marine ecosystems: two case studies. In: Pitcher, T.J., U.R. Sumaila, and D. Pauly, editors. Fisheries impacts on North Atlantic ecosystems: evaluations and policy explorations. Fisheries Centre Research Reports **9**(5):81-93.

Goram Y. 2007. Interview (Sawen, Sawaki, J., Raubaba, S. and L. Metansan, 10-11 February, 2007).

Halim A., T. Soekirman, and W. Ramono. 2007. Involving resource users in the regulation of access to resources for the protection of ecosystem services provided by protected areas in Indonesia. In: Sodhi, N., G. Acciaioli, M. Erb, A.K. Tan. Biodiversity and human livelihoods in protected areas – Case studies from the Malay Archipelago. UK: Cambridge University Press, p.122-138.

Hill H. 1998. The challenge of regional development in Indonesia. Australian Journal of International Affairs **52**(1):19-34.

ICES. 2005. Joint Report of the Study Group on Unaccounted Fishing Mortality (SGUFM) and the Workshop on Unaccounted Fishing Mortality (WKUFM). 25–27 September 2005. International Council for the Exploration of the Seas. Aberdeen. UK: ICES; CM 2005/B:08.

Djuang J., C. Imbir. 2007. Analisis Tingkat Kesejehteraan Masyarakat Raja Ampat. Participatory Rural Appraisal of Raja Ampat community (in Indonesian). Draft Report. Indonesia: Conservation International Indonesia. Available: Rotinsulu C. CI. Jl.Gunung Arfak.45.Sorong, Papua, Indonesia. Available: chris@conservation.or.id.

Johannes R. E., M. Riepen. 1995. Environmental economic and social implications of the live reef fish trade in Asia and the Western Pacific. Indonesia: The Nature Conservancy, Jakarta Selatan, Indonesia. 81p.

Johannes R. E., L. Squire, T. Graham, Y. Sadovy, and H. Renguul. 1999. Spawning aggregations of groupers (Serranidae) in Palau. The Nature Conservancy Marine Research Series Publication **1**:1-144.

Kadarusman K. unpublished document. Studi Awal: Kegiatan Pengeboman Ikan di Pulau Buaya. Akademi Perikanan Sorong. Departemen Kelautan dan Perikanan. The study of early fish bombing activities in Crocodile Island Sorong Fisheries Academy. Department of Marine affairs and Fisheries. Indonesia: Dewan Adat Suku Maya, Kepulauan Raja Ampat (The Maya Council, Raja Ampat islands). [in Indonesian].

McKenna S. A., G. R. Allen, and S. Suryadi. 2002. A marine rapid assessment of the Raja Ampat Islands, Papua Province, Indonesia. Conservation International, Washington, DC. RAP Bulletin of Biological Assessment **22**. 193p.

McNicoll G. 1982. Recent demographic trends in Indonesia. Population and Development Review **8**(4):811-819.

Mous P. J., J. S. Pet, Z. Arifin, R. Djohani, M. V. Erdmann, A. Halim, M. Knight, L. Pet-Soede, and G. Waidnya. 2005. Policy needs to improve marine capture fisheries management and to define a role for marine protected areas in Indonesia. Fisheries Management and Ecology **12**:259-268.

MRAG. 2005 IUU fishing on the high seas: Impacts on ecosystems and future science needs. Marine Resources Assessment Group, London, UK. Available: < www.high-seas.org/docs/Ecosystem_Impacts_IUU_Final_Report_MRAG_2005.pdf > [Accessed on February 12, 2007]

Muljadi A. unpublished data. The Nature Conservancy Coral Triangle Center. Jl Gunung Merapi no. 38, Kampung Baru, Sorong, Papua, Indonesia 98413. Available: amuljadi@tnc.org.

Muljadi A. Coastal rural appraisal report in Kofiau and Misool area of Raja Ampat Islands. Sorong, Indonesia: The Nature Conservancy, South East Asia Centre for Marine Protected Areas; 2004. 12 p.

Nurhakim S, V.P.H Nikijuluw, M. Badrudin, T. J. Pitcher, and G.A Wagey. 2008. A study of illegal, unreported and unregulated (IUU) fishing in the Arafura Sea, Indonesia. FAO, Rome 41p.

OECD. 2007. StatExtracts DataSet: Price Indices (MEI). Main Economic Indicators. Organisation for Economic cooperation and development. Available:

<www.stats.oecd.org/OECDStat_Metadata/ShowMetadata.ashx?Dataset=MEI_PRICES&ShowOnWeb=true&Lang=en> [Accessed on August 11, 2007].

Palomares M. L. D., J. J. Heymans. 2006. Historical ecology of the Raja Ampat Archipelago, Papua Province, Indonesia. Fisheries Centre Research Reports **14**(6):1-114.

Palomares M. L., J. J. Heymans, and D. Pauly. 2007. Historical ecology of the Raja Ampat Archipelago, Papua Province, Indonesia. History and Philosophy of the Life Sciences **29**(1):33-56.

Pitcher T., D. Kalikoski, G. Pramod, and K. Short. 2009. Not honouring the code. Nature **457**(7230):658-659.

Pitcher T. J., E. A. Buchary, and G. Pramod. 2007. An estimation of compliance of the fisheries of Indonesia with Article 7 (Fisheries Management) of the UN Code of Conduct for Responsible Fishing. Fisheries Centre Research Reports **14**(2):34.

Pitcher T. J., R. Watson. 2000. The basis for change 2: estimating total fishery extractions from marine ecosystems of the North Atlantic. In: Pauly, D., and T.J. Pitcher, editors. Methods for assessing the impact of fisheries on marine ecosystem of the North Atlantic. Fisheries Centre Research Reports **8**(2):40-53.

Pitcher T. J., R. Watson, R. Forrest, H. P. Valtýsson, and S. Guénette. 2002. Estimating illegal and unreported catches from marine ecosystems: a basis for change. Fish & Fisheries **3**(4):317-339.

Rabu P. 2006. Tabloid Raja Ampat. Conservation International, Indonesia. **6**:1-4.

Sumule A., R. Donnelly. 2003. Socio-economic conditions in the Raja Ampat islands. In: Donnelly R., D. Neville, and P.J. Mous, editors. Report on a rapid ecological assessment of the Raja Ampat Islands, Papua, Eastern Indonesia. The Nature Conservancy Coral Triangle Center, Indonesia. Final Draft: 21-41.

Surbakti S., R. L. Praptoprijoko, and S. Darmesto. 2000. Indonesia's 2000 population census: a recent national statistics activity. BPS Statistics Indonesia. United Nations Economic and Social Commission for Asia and the Pacific. Bangkok: UNESCAP.

Tanda L. 2002. A basic stock assessment of economically important coral reef fishes of Raja Ampat Islands, Papua Province, Indonesia. In: McKenna S.A., G.R. Allen, and S. Suryadi, editors. A marine rapid assessment of the Raja Ampat Islands, Papua Province, Indonesia. Conservation International, Washington, DC. RAP Bulletin of Biological Assessment **22**: 58-65.

Tesfamichael D., T. J. Pitcher. 2007. Estimating the unreported catch of Eritrean Red Sea fisheries. African Journal of Marine Science **29**(1):55-63.

Timmer J. 2005. Decentralisation and elite politics in Papua. State society and governance in Melanesia. Research School of Pacific and Asian Studies. The Australian National University. SSGM Discussion paper. **6**:1-18.

Wanandi J. 2002. Indonesia: A Failed State? The Washington Quarterly **25**(3)135-146.

Willoughby N., D. Monintja, and M. Badrudin. 1999. Do fisheries statistics give the full picture? Indonesia's non-recorded fish problems. In: Report of the regional workshop on the precautionary approach to fishery management. 25-28 February, 1997. Medan Indonesia. Bay of Bengal Program, Chennai, India. BOBP 1999, BOBP/REP/**82**: 163-172.

WWF/IUCN. 1996. Indonesian country study on integrated coastal and marine biodiversity management. Ministry of State for environment, Republic of Indonesia in cooperation with Directorate for Nature Management Kingdom of Norway. Appendix I Marine Conservation.

3 Ecological Restoration and Ideal Minimum Size of No-Take Zones in Marine Protected Areas of Raja Ampat, Indonesia[16]

3.1 Introduction

3.1.1 MPA for ecosystem based management

In the view of global declines of target and non-target marine fish and invertebrates (Alverson 1994; Pauly et al. 1998; Hutchings and Reynolds 2004), management emphasis has shifted towards integrated ecosystem approaches and a variety of nomenclature has evolved around this shift in management focus. Ecosystem approaches have been adopted under several names: Ecosystem approach to fisheries (EAF) by the UN Food and Agriculture Organization, Ecosystem based fisheries management (EBFM) by the US National Marine Fisheries Service and Ecosystem based management (EBM). Though based on the same concept, the difference in nomenclature represents some differences in operation. EAF is an overarching concept in that "it is not limited to management but could also include development, planning etc" (NMFS 1999; Garcia et al. 2003). EBM is management based on ecosystem approaches and can be interpreted correctly to mean several aspects of management, for example, impact of pollution on coral reefs to studying fisher poverty and its influence on resource use. EBFM is a more precise approach where the focus is on making decisions for the management of a resource (species or groups of species) based on understanding their roles and interrelationships in the ecosystem (NMFS 1999). Marine protected areas (MPAs) may

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offer an important tool to reduce fishing mortality, mediate habitat damage, increase stock biomass, and preserve ecosystem biodiversity (Gell and Roberts 2003; Hooker and Gerber 2004). Establishing MPAs may provide managers the opportunity to achieve EBM (Halpern et al. 2010) by addressing biological concerns and socio-economic needs (Sumaila et al. 2000), both of which are integral components of EBM. Review of 89 empirical results of marine reserves shows that on average the density, biomass, diversity and size of organisms are higher inside the reserves (Halpern 2003). Reserves appeared to promote an increased density of exploitable fishes in reef ecosystems in Philippines (Russ and Alcala 2003; Alcala and Russ 2006), and in the Caribbean (Bartholomew et al. 2008; Schrope 2008).

3.1.2 Raja Ampat

The Raja Ampat archipelago, consisting of approximately 610 islands, is located in the Southeast Asian Coral Triangle. The area extends over 45,000 km^2 and encompasses a variety of marine habitats including some of the most biodiverse coral reef areas on Earth (McKenna et al. 2002; Donnelly et al. 2003). The name Raja Ampat (four kings) refers to the four major islands (Figure 3.1)— Batanta, Misool, Salawati, and Waigeo (Donnelly et al. 2003). Small-scale fisheries operations on the reefs and in the inshore areas provide livelihoods for around 24,000 fishers (Dohar and Anggraeni 2007). Modeling work (Ainsworth et al. 2008a) and analysis of fisher perceptions (Ainsworth et al. 2008b) show that fishing pressure on the resources has caused the decline of several exploited species.

In 2002, Law no. 26 established the new Regency of Raja Ampat, and in 2003 a decree by the Bupati (Regent) declared Raja Ampat a 'Kabupaten Bahari' (maritime regency) (Conservation International 2008.). These political changes helped to establish a new network of marine reserves in 2006. The network covers a total of 4793 km^2 of sea area and 44% of reef area in Raja Ampat. It includes seven MPAs in the Islands: Ayau (28 km^2), Southwest Waigeo (162 km^2), Sayang-Wayag (178 km^2), South Waigeo or Dampier Strait (202 km^2), Mayalibit (277 km^2), Kofiau (328 km^2) and Southeast Misool (943 km^2). Ecological changes in three (Kofiau, Southeast Misool and Dampier Strait) MPAs were investigated.

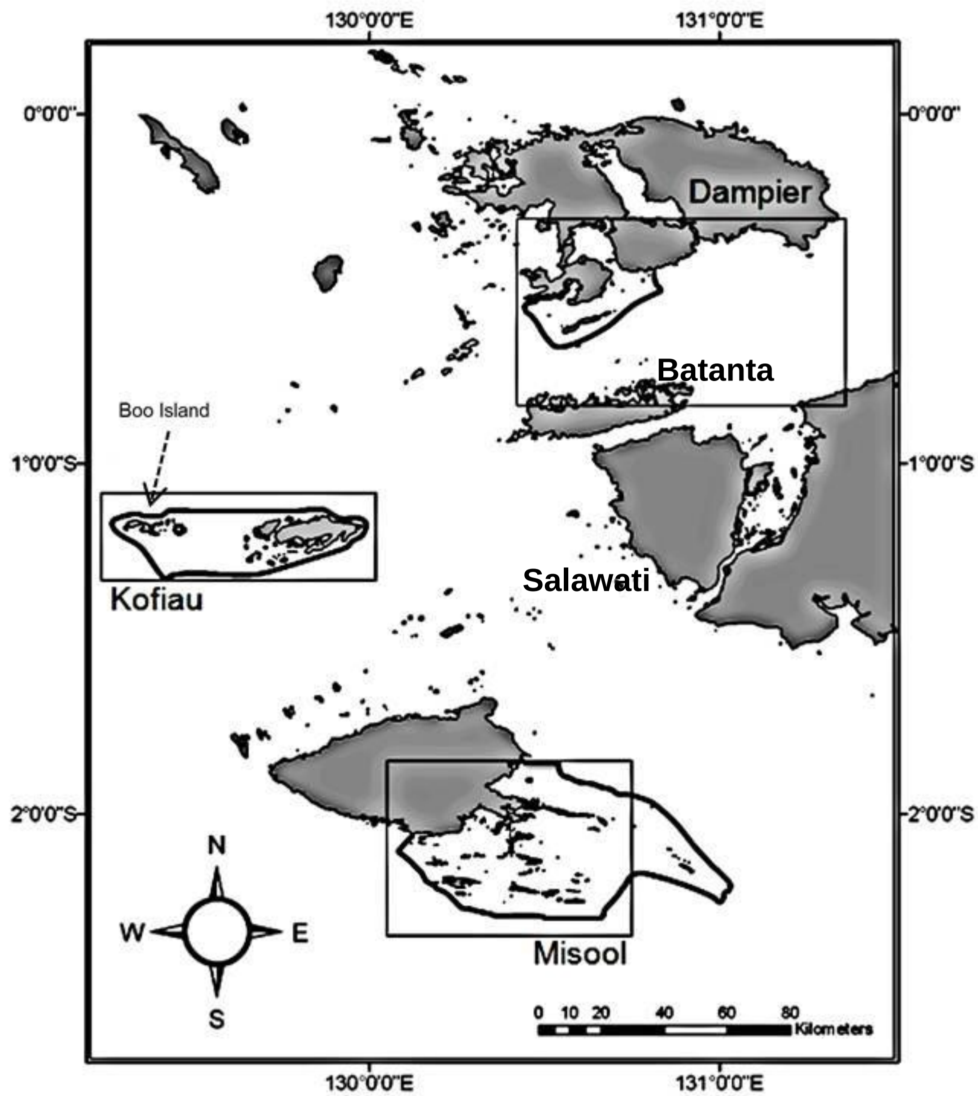

Figure 3.1 Map of Raja Ampat
The map shows the location of the Raja Ampat model (full map area) and the sub-area models within the Raja Ampat map (Dampier, Kofiau and Misool). The areas drawn in bold are the official MPA areas. (The figure is reproduced from Ainsworth et al. 2007).

3.1.3 Birds Head Seascape ecosystem based management project and spatial ecosystem based management research interests

Concerned with the issues of fisheries management and with the intention to develop environmentally sound ecosystem based policies, the Regency government participated in a science-based initiative—the Birds Head Seascape[17] Ecosystem Based Management (BHS EBM) project—funded by the David and Lucille Packard Foundation. The project involved field study and ecological modeling with The Nature Conservancy (TNC), Conservation International (CI), World Wildlife Fund (WWF) and the University of British Columbia (UBC). The following research questions focused on increased species biomass and MPA zoning options were identified during discussions with the Raja Ampat Fisheries Office and the partner institutions in the project.

i. Determine conservation benefits of restricting fishing effort inside MPA
ii. Determine conservation benefits of a single large versus several small MPAs

3.2 Methods

3.2.1 Ecopath with Ecosim and Ecospace

Ecopath with Ecosim (EwE) modeling approach was used to build the coral reef ecosystem model, and Ecospace for spatial analysis of MPAs. The details of EwE model parameterization can be found in Appendix B. EwE is a mass balance trophic simulation model that acts as a thermodynamic accounting system for marine ecosystems. Ecopath is a static snapshot of the system (Christensen 1992) that maps the thermodynamic flows in the system. Ecosim allows modeling of species composition changes over time (Walters et al. 1997); finally Ecospace integrates Ecopath and Ecosim across a two dimensional spatially explicit domain (Walters et al. 1998; Pauly et al. 2000). In Ecospace, a regular grid, which represents the study area, is divided into a number of habitat types. Each functional group is allocated to its appropriate habitat(s). Each cell hosts its own Ecosim simulation and is linked through symmetrical biomass flux in four directions. The exchange rate of biomass between the cells is determined mainly by dispersal rates in combination with the habitat type in adjacent cell, and group foraging and predator avoidance behaviour (Walters et al. 1998). Optimal and sub-optimal habitat in adjacent

[17] Bird Head Seascape is located in northwest Papua, Indonesia, it extends from Raja Ampat archipelago in the west to Cenderawasih Bay in the east and FakFak-Kaimana coastline in the south (http://www.conservation.org/sites/marine/initiatives/seascapes/birds_head/Pages/birdshead.aspx)

cell can be distinguished using parameters such as the availability of food, vulnerability to predation, and immigration/emigration rate. Dispersal rates represent net residual movement of functional groups on an annual basis and are not related to swimming speeds (see Walters et al. (1998) for more details). Details of the dispersal rates used in the model are provided in Appendix D. The effects of MPAs can be explored, and hypotheses regarding ecological function and effects of fisheries can be tested by delimiting an area as a protected zone in the Ecospace model. Previous authors have used Ecospace in this capacity (Walters et al. 1998; Pitcher and Buchary 2002; Salomon et al. 2002; Jiang et al. 2008; Le Quesne and Codling 2009).

3.2.2 Ecospace models used in the analysis

EwE models of Raja Ampat were built by integrating data from extensive field studies. Fisheries catch data for the same model were assembled from records of the the Sorong Regency Fisheries Office (Departemen Kelautan dan Perikanan, DKP), the Raja Ampat Regency Fisheries Office, and the Trade and Industry Office (Departemen Perinustrian dan Perdagangan). For greater detail on Ecopath model parameters and Ecosim fitting to time series, interested readers are referred to online technical reports (Ainsworth et al. 2007) and (Ainsworth et al. 2008c) (see Appendix F). This chapter explores marine protected areas using Ecospace models for Raja Ampat.

Raja Ampat Ecospace model was used for the analysis of the first research question. The Ecospace model inherited the standard EwE parameters from the 2005 Raja Ampat model. The model was used to compare the effects of restricting fisheries in three of the seven MPAs declared in Raja Ampat (Kofiau Island, Southeast Misool Island and Dampier Strait). The habitat maps were created by utilizing GIS information assembled by the BHS EBM project (Barmawi, M[18]. pers. comm.).

For the analysis of the second research question, the 2005 Raja Ampat model was adapted to build higher resolution Ecospace models for the same MPA areas analyzed in

[18] M. Barmawi TNC-CTC. Jl Pengembak 2, Sanur, Bali, Indonesia, 80228. unpublished data. Contact: joanne_wilson@tnc.org.)

the first research question. The higher resolution models (Figure 3.1); hereafter referred to as the sub-area models, improved the spatiotemporal representation and allowed us to simulate natural predator-prey segregation. The details for creation of sub-area models can be found in Appendix E. In our previous publication (Ainsworth et al. 2008a presented in Appendix F), the results of dynamic Ecosim simulations for Raja Ampat were synthesized.

3.2.3 Ecosystem effects of restricting fisheries inside the MPAs

The following paragraphs describe the three types of fishing restrictions employed in the Raja Ampat Ecospace model. At the end of 20-year simulations, the changes in biomass and catch for reef fish inside the MPAs and catch in the spillover regions (cells adjacent to the MPAs) were examined.

3.2.3.1 No fishing allowed (no-take)

In the Raja Ampat Ecospace model, all fisheries from inside the MPAs were eliminated to examine ecosystem recovery.

3.2.3.2 Commercial fisheries restricted (artisanal fisheries allowed)

The following fisheries were assumed to be commercial: driftnet, diving for live fish, diving with cyanide, blast fishing, trolling, purse seine, and pole and line. The other gear types were assumed to be primarily artisanal: spear and harpoon, reef gleaning, shore gillnets, permanent trap, portable trap, diving with spear, and set line. The distinction between artisanal and commercial catch is difficult to draw due to the unreported and unregulated nature of Raja Ampat reef fish fisheries and widespread casual local trade. The gear types were chosen to highlight the distinction between fishing sectors that require low capital investment, and/or whose products are destined for a small-scale local market; versus fishing sectors that require high capital investment and/or whose products are destined for regional or international market. Capital-intensive fishing methods such as compressor diving and fisheries that produce a high value product suitable for export, such as cyanide fishing, were assumed to be commercial. Blast fishing provides a high yield of low-value product that is likely to be absorbed by a large regional market, and so this fishery was assumed to be commercial. In this scenario, the above stated commercial fisheries were eliminated from inside the MPA in the Raja Ampat Ecospace model.

3.2.3.3 Destructive (blast fishing and cyanide) fisheries restricted

Destructive fishing practices, cyanide fishing and blast fishing, are widely prevalent in Eastern Indonesia (Erdmann and Pet-Soede 1996; Edinger et al. 1998) and are recognized as serious threats to coral reef ecosystems (Erdmann 2000; Fox et al. 2003). One of the major goals of the declared MPAs in Raja Ampat was to restrict the entry of fishers, especially fishers from outside Raja Ampat, engaged in destructive fishing. In this scenario, destructive fishing methods (cyanide fishing and blast fishing) were eliminated from inside the MPAs to examine recovery.

3.2.4 Ecological benefits of single large versus several small MPAs

In each of the sub-area models, 8 combinations of MPA sizes were analyzed. The total area protected was set equal to approximately 100 km^2 in all the scenarios. The total protected area was divided into combinations of 1, 2, 4 6, 8, 10, 20 and 30 MPAs (100km^2*1, 50*2, 25*4, 16.67*6, 12.5*8, 10*10, 5*20 and 3.3*30). To control the uncertainty from non-random siting of MPAs of different sizes, it was ensured that the multiple MPAs had similar amounts of reef habitat and the MPAs were located roughly evenly along the coast. It is also expected that as the number of MPAs increased, concern from non-random placement of MPAs became less as more values were averaged. The same pattern of closure was followed in all the sub area models to see if similar results would be obtained in the replications (see Figure 3.2 for an example of the closure patterns). At the end of the 20 year simulation run, the relative differences in reef fish biomass density between the various MPA sizes were analyzed.

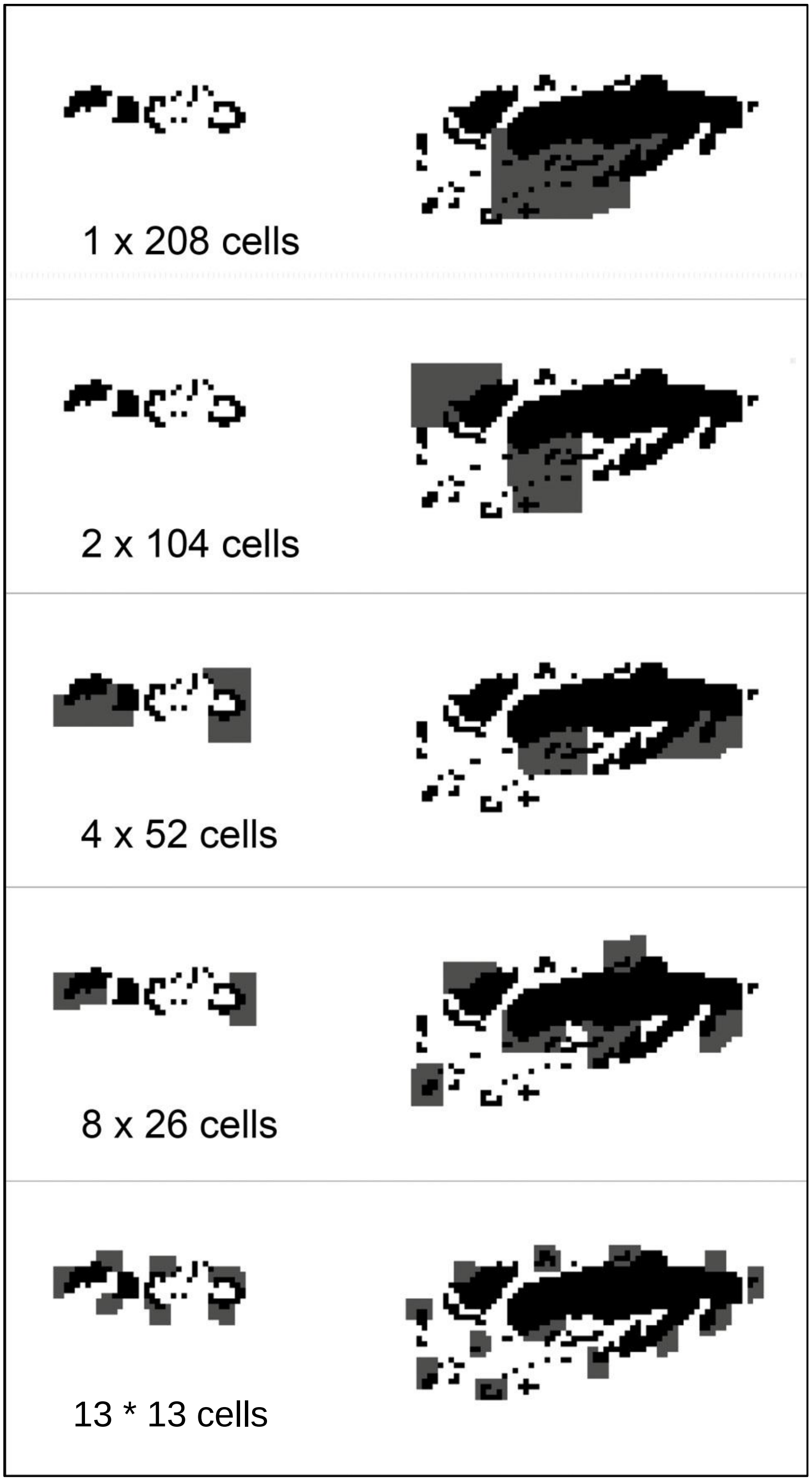

Figure 3.2 Example of the closure patterns in Kofiau Ecospace model.
The MPAs are indicated by the grey cells in the map. In all the closure scenarios the total area closed remains the same.

3.3 Results

3.3.1 Ecosystem effects of restricting fisheries inside the MPAs (Research question-1)

3.3.1.1 Reef fish biomass inside MPA

For the purpose of summarizing the results, the reef fish species in the Raja Ampat Ecospace model were aggregated into 3 categories: large reef fish, medium reef fish and small reef fish. In all three MPAs, the biomass of large reef fish was at least two times higher (Kofiau 2.3, Dampier 2.6 and Misool 3.1) when no fishing was allowed in the MPAs (Figure 3.3). Under restriction of commercial fisheries, rebuilding of the large reef fish populations was modest; a definite increase was observed only when no fishing was allowed (Kofiau 67% increase, Misool 92% and Dampier 112%).

When status quo fishing was continued, the biomass density of large and medium fish decreased relative to model initialization conditions in all the MPAs suggesting that current levels of fishing will lead to further declines in the biomass of target species. A trophic cascade was evident in all the MPAs. In response to increased biomass of large reef fish, the biomass of medium reef fish decreased; thereby, releasing the small reef fish from predation; this is consistent with known ecology (Carpenter and Kitchell 1996). Medium reef fish increased above their base levels only in the Dampier MPA in the ‘no fishing’ scenario. Compared to the ‘status quo’ scenario, the decline in medium reef fish was lower under fishing restriction scenarios, but the benefits of the MPAs were dampened by increased predation pressure from large reef fish. In the Misool Island and Dampier Strait MPAs, the highest increase in small reef fish (~140%) occurred when all commercial fishing was restricted. When all fishing was closed, the increased predation pressure caused a decrease in the biomass of small reef fish in Dampier (5%) and Misool (24%).

Disallowing destructive fishing alone was not sufficient to ensure rebuilding of the large and medium reef fish from their baseline biomass levels. The response of small reef fish in the Misool and Dampier Strait MPAs was strongest. In all the MPAs, reef fish benefited relative to the status quo scenario. Compared with restricting commercial fisheries, restricting destructive fisheries had similar pattern but smaller magnitude.

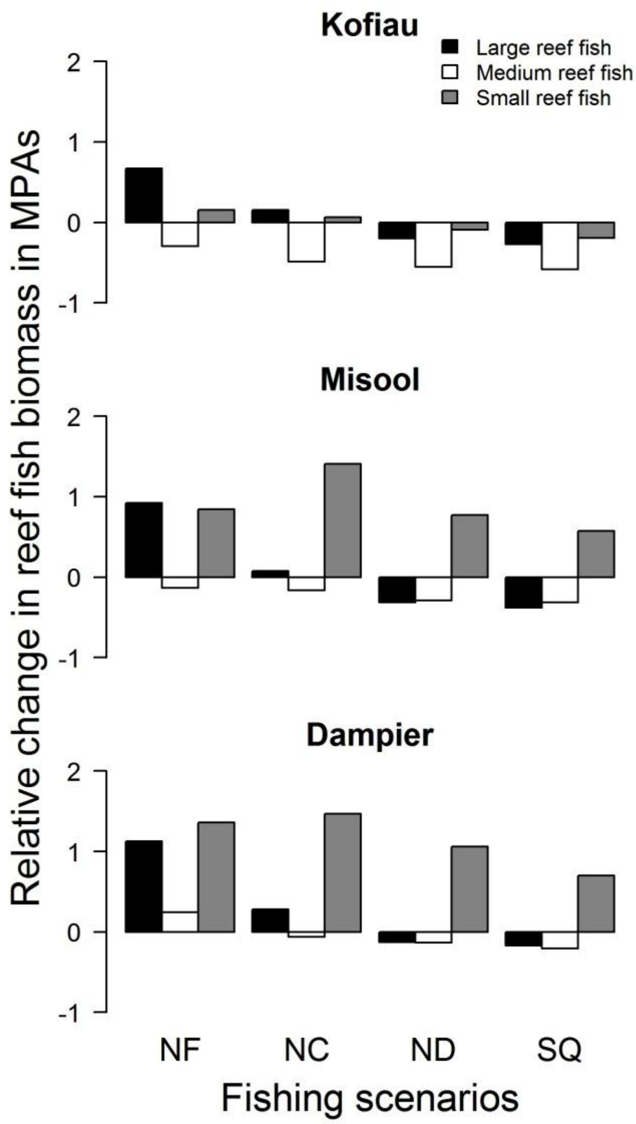

Figure 3.3 Relative biomass changes inside MPAs.
The graphs show the change in biomass relative to the base (2005) biomass of large, medium and small reef fish. The fishing restriction scenarios are shown on the horizontal axis: NF-No fishing, NC-No commercial, ND-No destructive and SQ-Status Quo. Black bars represent large reef fish, white bars represent medium reef fish, and grey bars represent small reef fish.

Ecospace model results were sensitive to dispersal rates (Figure 3.4). This is probably because species with higher dispersal rates, such as highly mobile pelagic fish, suffered fishing mortality from outside of the reserve. Prominent rebuilding effects were seen in species with low dispersal rates (<30 km.yr^{-1}), especially in the lower trophic level functional groups. At higher trophic levels, dispersal rates did not seem to influence the amount of rebuilding significantly. In the trophic level range from 2.5 to 3, highest level of rebuilding was shown by some functional groups with the lowest dispersal rates; however the response varied widely in this category especially because juveniles of several functional groups belonged in this category.

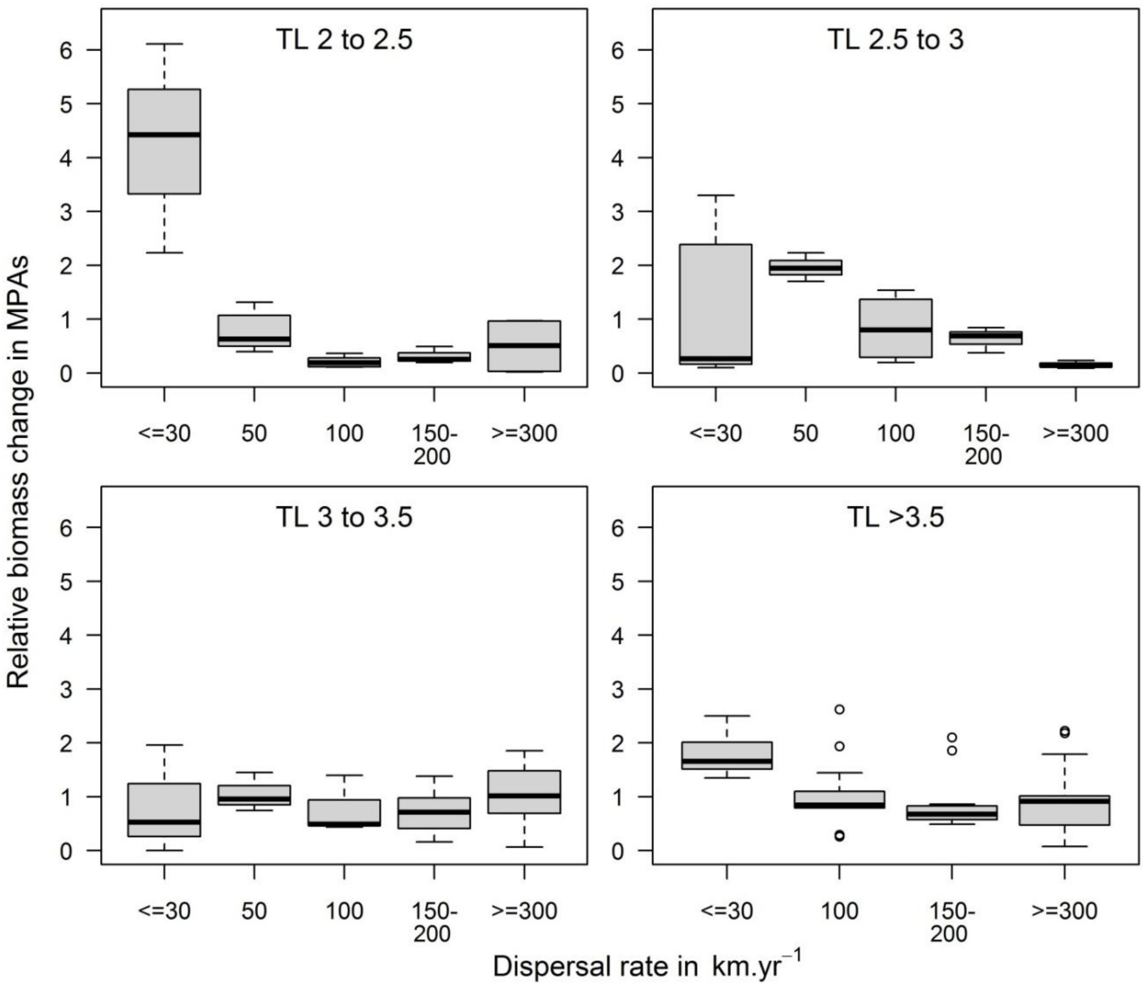

Figure 3.4 Influence of dispersal rate on biomass density change.
The relative change in biomass of functional groups obtained in the MPAs under no fishing scenario were combined and grouped into 5 classes according to the dispersal rates. Box plots are drawn to show the range in biomass change. The 5 classes of dispersal rates are shown on the horizontal axis.

3.3.1.2 Reef fish catch inside MPA and in spillover regions

Under status quo, the decrease in predator biomass caused a subsequent increase in biomass and catch of small reef fish in Misool and Dampier MPAs (Figure 3.5). Yield of large reef fish (Misool 49%, Dampier 24%), and medium reef fish (Misool 41%, Dampier 43%) decreased. The response in the Kofiau MPA was different; the catch of large, medium and small reef fish declined from the base levels in all the scenarios.

Catch of large, medium and small reef fish increased in the spillover region around Kofiau MPA. However, the catch from the spillover regions in Misool and Dampier did not increase between the status quo and fishing restriction scenarios. Catch inside the MPAs explained the difference in spillover catch in the Kofiau MPA versus Misool and Dampier MPAs. Increase in catch was observed only in Dampier Strait and Misool MPAs and not in Kofiau MPA. Alternatively, when the fishing was high inside the Kofiau spillover region, catches inside the MPA were not high. The results indicated a tradeoff between allowing some fisheries to operate inside the MPA versus expecting spillover effects from the MPA.

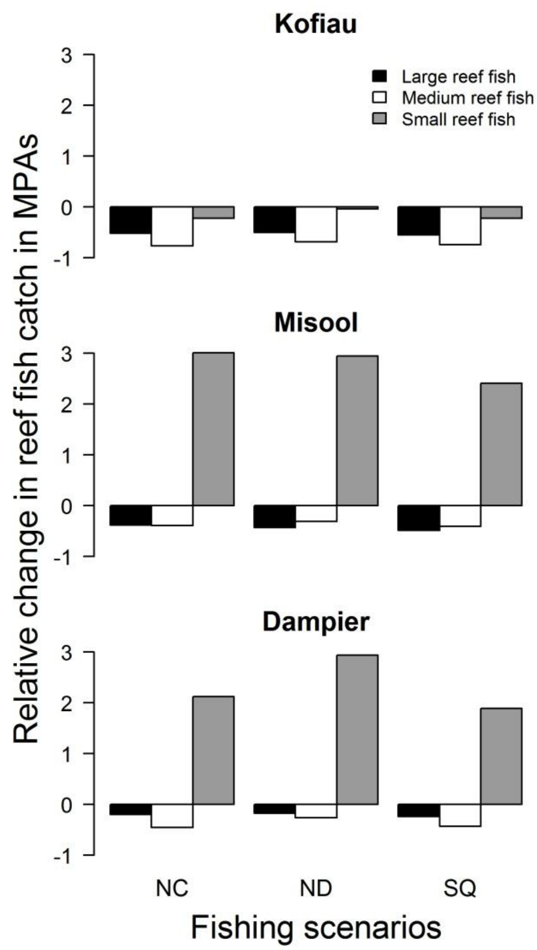

Figure 3.5 Relative catch changes inside MPA
The bars show the change in catch relative to the base (model initialization for 2005) catch of large, medium and small reef fish inside the MPAs under fishing restriction scenarios shown on horizontal axis: NC-No commercial, ND-No destructive and SQ-Status Quo. Black bars represent large reef fish, white bars represent medium reef fish, and grey bars represent small reef fish.

3.3.2 Ecological benefits of single large versus several small MPAs

The biomass density of large, medium and small reef fish increased within the protected areas as the size of MPA increased in Kofiau and Misool (Figure 3.6). However, benefits from the MPAs reached an asymptote as the size of the no-take area increased. Beyond about 16 km^2 (Kofiau) and 25 km^2 (Misool), there was no additional benefit in biomass density as the size of the MPA increased. The results from the Dampier Strait Ecospace model were opposite to the response seen in Kofiau and Misool, with the largest MPA showing the smallest values of biomass density for reef fish. The magnitude of response in the Misool model was higher than the response observed in the Kofiau model, but the absolute values at which the biomass density responses levelled off were very similar. The scenarios were repeated with default values for dispersal rate in Ecospace (300 km.yr^{-1} for all the functional groups). Overall, the performance of all the MPAs decreased at higher dispersal values: smaller MPAs performed worse than large MPAs.

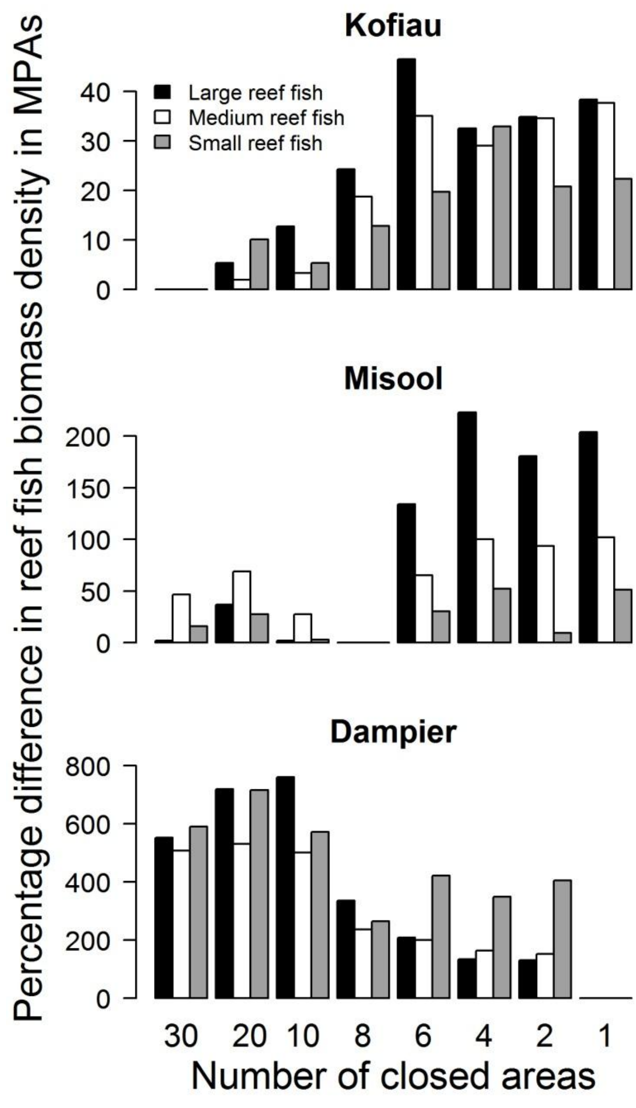

Figure 3.6 Figure 6 Biomass change in different MPA configurations.
The bars show the biomass density of large, medium and small reef fish relative to the smallest biomass density value among the 8 MPA size configuration scenarios. The various MPA configurations are shown on the horizontal axis. The MPA sizes associated with the MPA number are as follows: 3.33*30, 5*20, 10*10, 12.5*8, 16.67*6, 25*4, 50*2 and 100 km2*1. Black bars represent large reef fish, white bars represent medium reef fish, and grey bars represent small reef fish.

3.4 Discussion

3.4.1 Ecosystem effects of restricting fisheries inside the MPAs

The first research question is discussed under the following three sub-headings: dispersal rate, no-take areas and trophic cascade.

3.4.1.1 Dispersal rate

The functional groups in the model with low dispersal rates responded most to protection from MPAs. Dispersal rate is the parameter to which biomass distribution in Ecospace model is highly sensitive. Others have made similar observations (Watson et al. 2000; Beattie et al. 2002; Piroddi 2008; Christensen et al. 2009). The exchange rate across MPA boundaries is recognized as an important characteristic according to both empirical (McClanahan and Mangi 2000) and other modeling studies (Le Quesne and Codling 2009; Little et al. 2009) in determining the success of the MPA. Species specific or functional group specific dispersal rates are not very well known. The uncertainty in the dispersal rates used in the Ecospace model therefore has huge implications on the application of model results to the real world. Incorporating a sensitivity analysis on the dispersal rates in Ecospace will lead to a better understanding of the implications of the uncertainty on the results and monitoring existing and experimental closures will increase understanding of actual dispersal rates (Christensen et al. 2009). However, as suggested by Pitcher et al. (2002) "precise results, but not overall patterns are sensitive to uncertainties". It is clear that for more mobile organisms, the optimum size for closed area increases. There is thus no 'one' optimum size for an MPA; the decision on size depends on the major species for which the protection is aimed at. New approaches designed to protect far ranging pelagic species include protecting "demographically critical areas" where the populations have higher vulnerability (Game et al. 2009) or "temporary spatial closures" with the location of the closed areas changing during the course of the year (Grantham et al. 2008).

3.4.1.2 No-take areas and spillover

Another clear result from the Ecospace analysis was that a no-take area of 'some' size was needed for rebuilding the population. Compared to partial fishing restrictions, the increases in biomass density observed when the MPAs were set as no-take were much higher. A similar result was obtained in an analysis of dolphin populations in Ionian Sea––when no fishing was allowed—rebuilding of dolphin populations occurred, but the dolphin populations showed only a small increase when the artisanal fisheries were allowed (Piroddi 2008).

Among the three spillover regions compared, the relative increase in fishing effort in the spillover region was highest in Kofiau. It has been suggested that higher fishing effort in the spillover region encourages spillover (Walters et al. 2009). Studies also show that high spillover across a long perimeter of an MPA can drain the MPA of the rebuilding fish biomass (Watson et al. 2000). The results also indicated a trade-off between catch in the spillover region and catch inside MPA under restricted fishing effort scenarios.

The results have implications on MPA design—whether a buffer zone should be placed between the closed (no-take) and open areas. Spillover would depend on the type of fisheries allowed in the buffer zone and the trophic cascade effects. If the buffer zone fisheries are for example artisanal hook and line fisheries, then they might target only the top predator species in the buffer zone and not dilute the spillover of other reef fish and pelagic fish for the drift-net fishers outside the buffer zone. More modeling effort is needed to understand if buffer zones would enhance or dilute the spillover effects. It might be possible to design MPA zoning in concordance with the dispersal rate of species with very selective gears allowed in the respective buffer zones. Spillover from a reserve would also depend on distance from the reserve (Russ et al. 2003), on non-fisheries aspects like tidal flow and reef morphology (McClanahan and Mangi 2000), and on whether fishers enter the spillover habitat area and find it suitable to fish (Forcada et al. 2009).

3.4.1.3 Trophic cascade

The increase in biomass of large reef fish inside the MPA depressed the population of medium reef fish leading to an increase in biomass of small reef fish. The trophic cascade in MPAs has also been reported in other studies using Ecospace (Piroddi 2008) with high predator densities and low prey densities inside the MPA. A comparison of unfished reef versus fished reefs has shown that a larger population of higher trophic level species “overwhelmed the fish assemblages so that the biomass pyramid was inverted” (Sandin et al. 2008). Trophic cascade could be a reason why population increase of mid-trophic level species in an MPA may moderate. Mid-trophic level species will respond to protection when the release from fishing pressure is greater than the increase in predation pressure under MPA protection. However, changes in size structure of mid-trophic level species due to reduced fishing pressure in an MPA could offset the “negative impacts of enhanced predation” (Mumby et al. 2006).

3.4.2 Ecological benefits of single large versus several small MPAs

Research using Ecospace models have favored larger MPAs (Martell et al. 2005). Large MPAs would be needed to offset high exchange rates of the fish especially in situations of food limitation, excessive predation pressure and shifting of productive areas due to changes in ocean circulation patterns (Martell et al. 2005); this may be an argument for large MPAs in areas with ephemeral upwelling regimes. Larger MPAs enhanced spillover owing to “spatial cascade effects”—high predator and low prey biomass inside MPA and vice-versa outside—and fishing effort concentration outside the MPAs (Piroddi 2008). In ecosystem models, larger protected areas were able to restore fisheries while smaller protected areas were unable to avert collapses in a highly exploited ecosystem in the South China Sea (Pitcher and Buchary 2002). Small sizes of MPAs and movement of fish into the spillover regions could render the MPA ineffective (Walters 2000). Spatially explicit population dynamic modeling arrived at similar conclusions: Stefansson and Rosenberg (2005) concluded that large percentages of fish biomasses needed to be protected for rebuilding a stock and small area closures were “unlikely to give substantial protection”.

Following the work on the Raja Ampat Ecospace model, which showed that some amount of no-take area is essential, the next step was to determine the "ideal minimum size" of no-take zones inside a reserve. The results for Kofiau and Misool indicate that after an increase in the size of no-take areas beyond 16 to 25 km^2, the benefits calculated in terms of biomass density of reef fish asymptote. This result was opposite in the Dampier Strait model, but some dynamic instability was present in that model casting doubt on the finding. Based on the Kofiau and Misool Ecospace model results, the ideal minimum size of a no-take area in Raja Ampat is 16 to 25 km^2; however, the precise values cannot be relied upon as management advice. Since the analysis is done for reef fish that are (in general) less mobile, the estimate of minimum size of "no-take" area is conservative; the value would only be higher for more vagile species. Other uncertainties associated with model parameterization also influence the results. Factors not addressed in the Ecospace model such as habitat quality improvements (Jiang et al. 2008), hydrodynamics, spawning aggregations and source and sink populations will also influence the ideal minimum size of no-take area.

However, biomass density benefits from MPAs asymptote as reserve size increases. The research on the minimum size of no-take areas offers options to integrate the ecological result with social considerations that might favor smaller or larger no-take areas. For Waigeo Island, some plans exist for small 0.2 km^2 no-take zones, no zoning plans have been made for Kofiau and Misool (Rahwarin B. pers. comm.).[19] In an area like Misool, that has a diverse human population; it might be difficult to implement a single large MPA owing to a wide array of customary tenure agreements and/or difficulty in arriving at a consensus because of large number of players. In terms of population, Kofiau is relatively homogenous, and Boo island (part of Kofiau see Figure 3.1) is relatively uninhabited; thus, it might be feasible to declare a large no-take area around Boo Island. These opinions are based on cursory understanding of the social dynamics. Clever zoning that will result in successful protection will need community and administrative collaboration, probably encouraged by the presence of NGOs.

[19] Becky Rahawarin. DKP, Raja Ampat. Jl. A. Yani, Kuda laut, Sorong, Papua.

3.5 Conclusion

Though there is scepticism (Willis et al. 2003) about the utility of MPAs as fisheries management or conservation tools, several empirical and modeling studies have demonstrated the biomass and spillover enhancing potential of reserves (Russ et al. 2003; Jiang et al. 2008; Little et al. 2009; Stelzenmuller et al. 2009). To improve management potential, Sale et al. (2005) identified gaps in the research (distance and direction of larval dispersal, movement patterns in juveniles and adults, changes to community structure due to trophic cascades, hydrodynamic patterns) that complicate the decision regarding size and placement of MPAs. This chapter based on Ecospace modeling has tried to address issues about dispersal, trophic cascades, and the size of MPAs. Buffer zones are an interesting research direction for future work, with possibility for phased (in concordance with dispersal rates of species) deployment of very selective gears in successive buffer areas.

Near Apo Island, Philippines, the purpose of an established reserve was to ban non-residents from the fishing ground and prohibit destructive fishing gears; their management goals were similar to those in Raja Ampat. It is interesting that success of the MPA near Apo Island later led to increased revenues from tourism, thus the "islanders had to fish less to support their families" (Russ et al. 2004). Success of MPAs will depend on "understanding of the spatial structure of impacted fisheries, ecosystems and human communities" and "careful planning, evaluation and appropriate monitoring programs" as stated in Hilborn et al. (2004) and echoed 5 years later by Le Quesne (2009).

3.6 References

Ainsworth C. H., D. A. Varkey, and T. J. Pitcher. 2008a. Ecosystem simulations supporting ecosystem-based fisheries management in the Coral Triangle, Indonesia. Ecological Modelling **214**(2-4):361-374.

Ainsworth C. H., T. J. Pitcher, and C. Rotinsulu. 2008b. Evidence of fishery depletions and shifting cognitive baselines in Eastern Indonesia. Biological Conservation **141**(3):848-859.

Ainsworth C. H., D. A. Varkey, and T. J. Pitcher. 2008c. Ecosystem simulation models of Raja Ampat, Indonesia, in support of ecosystem based fisheries management. In: Bailey M., T. J. Pitcher, editors. Ecological and Economic Analyses of Marine Ecosystems in the Birds Head Seascape, Papua, Indonesia: II. Fisheries Centre Research Reports **16**(1):3-122.

Ainsworth C. H., D. A. Varkey, and T. J. Pitcher. 2007. Ecosystem simulation models for the Bird's Head Seascape, Papua In: Pitcher T.J., C.H. Ainsworth, and M. Bailey, editors, Ecological and economic analyses of marine ecosystems in the Birds Head Seascape, Papua, Indonesia. Fisheries Centre Research Reports **15**(5):6–172.

Alcala A. C., G. R. Russ. 2006. No-take marine reserves and reef fisheries management in the Philippines: a new people power revolution. Ambio: A Journal of the Human Environment **35**(5):245-254.

Alverson D. L. 1994. A global assessment of fisheries bycatch and discards. FAO Fisheries Technical Paper **339**:1-233.

Bartholomew A., J. A. Bohnsack, S. G. Smith, J. S. Ault, D. E. Harper, and D. B. McClellan. 2008. Influence of marine reserve size and boundary length on the initial response of exploited reef fishes in the Florida Keys National Marine Sanctuary, USA. Landscape Ecology **23**:55-65.

Beattie A., U. R. Sumaila, V. Christensen, and D. Pauly. 2002. A model for the bioeconomic evaluation of marine protected area size and placement in the North Sea. Natural Resource Modeling **15**(4):413-437.

Carpenter S. R., J. F. Kitchell. 1996. The trophic cascade in lakes. Cambridge University Press, New York, USA. 399p.

Christensen V. 1992. ECOPATH II--A software for balancing steady-state ecosystem models and calculating network characteristics. Ecological Modelling **61**(3):169-185.

Christensen V., Z. Ferdaña, and J. Steenbeek. 2009. Spatial optimization of protected area placement incorporating ecological, social and economical criteria. Ecological Modelling **220**(19):2583-2593.

Conservation International. 2008. Environmental Friendly Development in Raja Ampat. Available: <www.conservation.or.id/home.php?catid=30&tcatid=178&page=g_peluang.detail> [Accessed on July 30, 2008].

Dohar A. G., D. Anggraeni. 2007. Valuasi Ekonomi Sumberdaya Alam di Kepulauan Raja Ampat. Conservation International, Bekerjasama dengan, Universitas Negeri Papua. (in Indonesian) Economic valuation report for Raja Ampat, Indonesia; Conservation International Indonesia.Available: Rotinsulu C. Gunung Arfak CI. Jl. 45. Sorong, Papua, Indonesia. Available: chris@conservation.or.id.

Donnelly R., D. Neville, and P. J. Mous. 2003. Report on a rapid ecological assessment of the Raja Ampat Islands, Papua, Eastern Indonesia. The Nature Conservancy Coral Triangle Center. Indonesia. Final Draft. 250p.

Edinger E. N., J. Jompa, G. V. Limmon, W. Widjatmoko, and M. J. Risk. 1998. Reef degradation and coral biodiversity in Indonesia: Effects of land-based pollution, destructive fishing practices and changes over time. Marine Pollution Bulletin, **36**(8):617-630.

Erdmann M. V. 2000. Destructive fishing practices in Indonesian seas. Pages 392-393. In: Sheppard C. editor. Seas at the millenium: an environmental evaluation, Elsevier, Pergamon, Amsterdam.

Erdmann M. V., L. Pet-Soede. 1996. How fresh is too fresh? The live reef food fish trade in Eastern Indonesia. Naga: The ICLARM Quarterly **19**:4-8.

Forcada A., C. Valle, J. L. Sanchez-Lizaso, J. T. Bayle-Sempere, and F. Corsi. 2009. Structure and spatio-temporal dynamics of artisanal fisheries around a Mediterranean marine protected area. ICES Journal of Marine Science **67**(2):191-203.

Fox H. E., J. S. Pet, R. Dahuri, and R. L. Caldwell. 2003. Recovery in rubble fields: long-term impacts of blast fishing. Marine Pollution Bulletin **46**(8):1024-1031.

Game E. T., H. S. Grantham, A. J. Hobday, R. L. Pressey, A. T. Lombard, L. E. Beckley, K. Gjerde, R. Bustamante, H. P. Possingham, and A. J. Richardson. 2009. Pelagic protected areas: the missing dimension in ocean conservation. Trends in Ecology and Evolution **24**(7):360-369.

Garcia S. M., A. Zerbi, C. Aliaume, T. DoChi, and G. Lasserre. 2003. The ecosystem approach to fisheries. Issues, terminology, principles, institutional foundations, implementation and outlook. FAO Fisheries Technical Paper. No. **443**. FAO, Rome. 71p.

Gell F. R., C. M. Roberts. 2003. Benefits beyond boundaries: the fishery effects of marine reserves. Trends in Ecology and Evolution **18**(9):448-455.

Grantham H. S., S. L. Petersen, and H. P. Possingham. 2008. Reducing bycatch in the South African pelagic longline fishery: the utility of different approaches to fisheries closures. Endangered Species Research **5**:291-299.

Halpern B. S. 2003. The impact of marine reserves: do reserves work and does reserve size matter? Ecological Applications **13**(1) Supplement:S117-S137.

Halpern B. S., S. E. Lester, and K. L. McLeod. 2010. Placing marine protected areas onto the ecosystem-based management seascape. Proceedings of the National Academy of Sciences **107**(43):18312-18317.

Hilborn R., K. Stokes, J. Maguire, T. Smith, L. W. Botsford, M. Mangel, J. Orensanz, A. Parma, J. Rice, J. Bell, K. L. Cochrane, S. Garcia, S. J. Hall, G. P. Kirkwood, K. Sainsbury, G. Stefansson, and C. Walters. 2004. When can marine reserves improve fisheries management? Ocean and Coastal Management **47**(3-4):197-205.

Hooker S. K., L. R. Gerber. 2004. Marine reserves as a tool for ecosystem-based management: the potential importance of megafauna. Bioscience **54**(1):27-39.

Hutchings J. A., J. D. Reynolds. 2004. Marine fish population collapses: consequences for recovery and extinction risk. Bioscience **54**(4):297-309.

Jiang H., H. Q. Cheng, W. J. F. Le Quesne, H. G. Xu, J. Wu, H. Ding, and F. Arreguin-Sanchez. 2008. Ecosystem model predictions of fishery and conservation trade-offs resulting from marine protected areas in the East China Sea. Environmental Conservation **35**(02):137-146.

Le Quesne W. J. F., E. A. Codling. 2009. Managing mobile species with MPAs: the effects of mobility, larval dispersal, and fishing mortality on closure size. ICES Journal of Marine Science **66**(1):122-131.

Le Quesne W. J. F. 2009. Are flawed MPAs any good or just a new way of making old mistakes? ICES Journal of Marine Science **66**(1):132-136.

Little L. R., A. E. Punt, B. D. Mapstone, G. A. Begg, B. Goldman, and N. Ellis. 2009. Different responses to area closures and effort controls for sedentary and migratory harvested species in a multispecies coral reef line fishery. ICES Journal of Marine Science **66**(9):1931-1941.

Martell S. J. D., T. E. Essington, B. Lessard, J. F. Kitchell, C. J. Walters, and C. H. Boggs. 2005. Interactions of productivity, predation risk, and fishing effort in the efficacy of marine protected areas for the Central Pacific. Canadian Journal of Fisheries and Aquatic Sciences **62**(6):1320-1336.

McClanahan T. R., S. Mangi. 2000. Spillover of exploitable fishes from a marine park and its effect on the adjacent fishery. Ecological Applications **10**(6):1792-1805.

McKenna S. A., G. R. Allen, and S. Suryadi. 2002. A marine rapid assessment of the Raja Ampat Islands, Papua Province, Indonesia. Conservation International, Washington, DC. RAP Bulletin of Biological Assessment 22. 193p.

Mumby P. J., C. P. Dahlgren, A. R. Harborne, C. V. Kappel, F. Micheli, D. R. Brumbaugh, K. E. Holmes, J. M. Mendes, K. Broad, J. N. Sanchirico, K. Buch, S. Box, R. W. Stoffle, and A. B. Gill. 2006. Fishing, trophic cascades, and the process of grazing on coral reefs. Science **311**(5757):98-101.

NMFS, 1999. Ecosystem based fishery management. A report to Congress by the ecosystem principles advisory panel. National Marine Fisheries Service. National Oceanic and Atmospheric Administration. Available online: http://www.nmfs.noaa.gov/sfa/EPAPrpt.pdf [Accessed 29 October, 2010].

Pauly D., V. Christensen, and C. J. Walters. 2000. Ecopath, Ecosim and Ecospace as tools for evaluating ecosystem impact of fisheries. ICES Journal of Marine Science **57**:697-706.

Pauly D., V. Christensen, J. Dalsgaard, R. Froese, and F. Torres Jr. 1998. Fishing down marine food webs. Science **279**(5352):860-863.

Piroddi C. 2008. An ecosystem-based approach to study two dolphin populations around the Island of Kalamos, Ionian Sea, Greece. Master Thesis, University of British Columbia. Vancouver, Canada. 122p.

Pitcher T. J., E. A. Buchary. 2002. Ecospace simulations for Hong Kong Waters. In: Pitcher, T., E. Buchary, E. and P. Trujillo, editors. Spatial Simulations of Hong Kong's Marine Ecosystem: Forecasting with MPAs and Human-Made Reefs. Fisheries Centre Research Reports **10**(3):27-35.

Pitcher T. J., E. A. Buchary, and T. Hutton. 2002. Forecasting the benefits of no-take human-made reefs using spatial ecosystem simulation. ICES Journal of Marine Science **59**:17-26.

Russ G. R., A. C. Alcala. 2003. Marine reserves: rates and patterns of recovery and decline of predatory fish, 1983-2000. Ecological Applications **13**(6):1553-1565.

Russ G. R., A. C. Alcala, and A. P. Maypa. 2003. Spillover from marine reserves: the case of Naso vlamingii at Apo Island, the Philippines. Marine Ecology Progress Series **264**:15-20.

Russ G. R., A. C. Alcala, A. P. Maypa, H. P. Calumpong, and A. T. White. 2004. Marine reserve benefits local fisheries. Ecological Applications **14**(2):597-606.

Sale P. F., R. K. Cowen, B. S. Danilowicz, G. P. Jones, J. P. Kritzer, K. C. Lindeman, S. Planes, N. V. C. Polunin, G. R. Russ, and Y. J. Sadovy. 2005. Critical science gaps impede use of no-take fishery reserves. Trends in Ecology and Evolution **20**(2):74-80.

Salomon A. K., N. P. Waller, C. McIlhagga, R. L. Yung, and C. Walters. 2002. Modeling the trophic effects of marine protected area zoning policies: A case study. Aquatic Ecology **36**(1):85-95.

Sandin S. A., J. E. Smith, E. E. DeMartini, E. A. Dinsdale, S. D. Donner, A. M. Friedlander, T. Konotchick, M. Malay, J. E. Maragos, and D. Obura. 2008. Baselines and degradation of coral reefs in the northern Line Islands. PLoS One **3**(2):e1548.

Schrope M. 2008. Conservation: providential outcome. Nature **451**(7175):122-123.

Stefansson, G. and A. A. Rosenberg. 2005. Combining control measures for more effective management of fisheries under uncertainty: quotas, effort limitation and protected areas. Philosophical Transactions of the Royal Society Biological Sciences **360**: 133-146.

Stelzenmuller V., F. Maynou, and P. Martín. 2009. Patterns of species and functional diversity around a coastal marine reserve: a fisheries perspective. Aquatic Conservation: Marine and Freshwater Ecosystems **19**(5):554-565.

Sumaila U. R., S. Guenette, J. Alder, and R. Chuenpagdee. 2000. Addressing ecosystem effects of fishing using marine protected areas. ICES Journal of Marine Science **57**(3):752-760.

Walters C. 2000. Impacts of dispersal, ecological interactions, and fishing effort dynamics on efficacy of marine protected areas: how large should protected areas be? Bulletin of Marine Science **66**(3):745-757.

Walters C., R. Hilborn, and C. Costello. 2009. Comparison of marine protected area policies using a multispecies, multigear equilibrium optimization model (EDOM). Fisheries Centre Working Paper **4**:1-44.

Walters C. J., D. Pauly, and V. Christensen. 1998. Ecospace: prediction of mesoscale spatial patterns in trophic relationships of exploited ecosystems, with emphasis on the impacts of marine protected areas. Ecosystems **2**(6):539-554.

Walters C. J., D. Pauly, and V. Christensen. 1997. Structuring dynamic models of exploited ecosystems from trophic mass-balance assessments. Reviews in Fish Biology and Fisheries **7**:139-172.

Watson R., J. Alder, and C. Walters. 2000. A dynamic mass-balance model for marine protected areas. Fish and Fisheries **1**(1):94-98.

Willis T. J., R. B. Millar, R. C. Babcock, and N. Tolimieri. 2003. Burdens of evidence and the benefits of marine reserves: putting Descartes before des horse? Environmental Conservation **30**(02):97-103.

4 Reconstructing Ancient New Zealand Snapper Biomass from Archaeological Data[20]

4.1 Introduction

4.1.1 Fisheries management and restoration

Overfishing and the consequent collapse of marine ecosystems have been veiled by the shifting baselines syndrome (Pauly et al. 1998; Jackson et al. 2001; Pitcher 2001) wherein the cognitive baseline of each generation for pristine nature shifts towards a more exploited system[21]. The Food and Agriculture Organization of the United Nations maintains a global repository of fisheries' statistics which includes records from 1950, but this short time period (1950-present) that does not cover the long period of exploitation faced by many species for centuries/millenia prior (Roberts 2007). When fisheries catch data from 1950 (or later) is used as baseline in fisheries assessments, the models erroneously assume that the species were at their unexploited biomass levels at that time; the pre-1950 declines in biomass are ignored. For example, current stock assessments for Gulf of Maine cod are based on a fraction of the ancient biomass (Rosenberg et al. 2005). Other impacts of human fishing activity—decline in the size of rockfish in British Columbia, Canada (McKechnie 2005) and in California (Love et al. 2002; Braje 2009)—have been reported by archaeological studies. When research is based on under-estimates of unexploited population biomass or growth, the calculations could lead to erroneous reference points for fisheries management strategies.

[20] A version of this chapter will be submitted for publication. Varkey, D. A., Pitcher, T. J., Leach, F., MacDiarmid, A. Exploring ecosystem carrying capacity – Reconstruction of New Zealand snapper population using archaeological data.

[21] Each generation thinks that the state of the ecosystems during its time represents the pristine state of nature, but in reality the ecosystem changed in the period when it was exploited by the previous generations

When working towards marine ecosystem-based management, it is valuable to understand the carrying capacity of the ecosystem, and how species assemblages have changed through history. Apart from satisfying our curiosity about ancient fishing, "understanding the influence of human predation on marine resources" (Leach 2006) and the underlying drivers (Campbell et al. 2009) is an important component of rational management (Pitcher and Lam 2010). Potential sources of information on the carrying capacity of a system are estimates of ancient abundance; information on carrying capacity will help guide rebuilding and restoration.

4.1.2 Modern snapper fishing in New Zealand

New Zealand Snapper (*Chrysophrys auratus*), a member of the family Sparidae (sea breams), are present mostly between 10 to 60 m depth; therefore, they are commonly found within a few kilometers of the shoreline. Snapper fishery is an important contributor to the coastal fisheries in New Zealand. Total annual snapper catches from the late 1980s to 2007 have ranged between 6,000 and 8,000 tonnes (MFish New Zealand 2007). When signs of overfishing were observed in mid 1980s, a quota management system (QMS) was introduced (Davies and McKenzie 2001), and QMS continues to be the management regime in the present. The analysis is based on snapper population in SNA 8 (Figure 4.1), one of the five snapper management areas in NZ. Snapper catches from SNA 8 contribute about one-fifth of the total snapper landings. The snapper stock in SNA 8 approximates a biological stock, including mainly the stock which recruits from Kaipara Harbour and some other smaller stocks (Paul[22] pers. comm.). For stock assessment purposes, SNA 8 is considered to be "separate from other snapper stocks and to be defined by the SNA 8 management area" (Davies et al. 2006).

[22] Larry Paul, National Institute of Water & Atmospheric Research, New Zealand

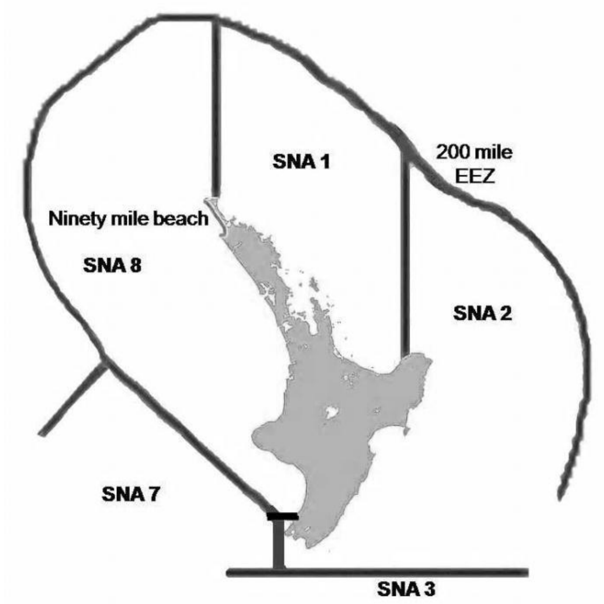

Figure 4.1 Map of North Island of New Zealand.
The modern data are from snapper management area SNA 8 and the ancient sample is from Twilight beach which is located at the northern end of the ninety mile beach which is shown as a dark strip in the figure.

4.1.3 Prehistoric fishing in New Zealand

Ancestors of today's Maori were Polynesian immigrants who arrived in New Zealand about 800 years ago (Wilmshurst et al. 2008). They possessed a long tradition of fishing and maritime skills. Maori fishing was typically confined to coastal waters less than 100 m deep, and their fishing methods included seine nets, small hand nets, set nets, hoop nets, basket like traps, netting walls, and hook and line fishing (Leach 2006). Leach (2006) describes the ease of capturing snapper: "Snapper have strong spines which become entangled in almost any mesh and are seldom caught by the gills. If there are plenty of snapper to be caught, you would only need a net with very large mesh size." The description of seine nets was catalogued by explorer Joseph Banks as "being so big (80 to 100 fathoms long and 5 to 6 feet wide) that it takes all the inhabitants of the village working together to pull one (Doubtless Bay, North Island)" (Leach 2006). The most abundant fish in Maori catches in New Zealand were: barracouta (*Thyrsites atum*), blue cod (*Parapercis colias*), snapper, and spotty (*Pseudolabrus celidotus*).

Snapper remains were found in 54 archaeological sites in New Zealand, most of them on the North Island. Length data were obtained from fish remains (n = 1914) (referred to as 'ancient snapper') at Twilight Beach, the site with the highest (92.6%) relative abundance (compared to other fish) of snapper bones (Leach 2006). The Twilight Beach archaeological site is located in SNA 8 at the northern end of Ninety Mile Beach (Figure 4.1). Middens at this site have been dated to the period 1400 -1500 AD (Leach 2006). It is expected that the middens from Twilight beach represent the ancient fish population in SNA 8. A study of spatial distribution of modern snapper showed that the strength of different year classes for fish 5 years and older were quite consistent throughout SNA 8; and inferred "little spatial variation in average growth rate of snapper" (Walsh et al. 2006). In particular reference to Ninety Mile Beach, they found that the "spread" of length at age was greater for fish older than 9 years (Walsh et al. 2006).

The objective was to estimate the ancient biomass (c. 1400AD) of New Zealand snapper combining archaeological data with tools in fisheries science. Candidate growth curves for the ancient snapper were proposed; total mortality estimates for the corresponding growth curves were calculated. The growth and mortality estimates were combined in an equilibrium age structure model to estimate the ancient population biomass. Finally, the estimates of ancient snapper biomass were contrasted with published estimates of modern snapper population biomass from stock assessments and surveys in New Zealand and a benchmark for ancient population size was provided.

4.2 Methods

4.2.1 Growth parameters of the modern population

The growth of the modern snapper population can be described by the von Bertalanffy growth function (VBGF):

(1) $L_{age} = L_{\infty}\left(1 - e^{-k(t-t_0)}\right)$

where, L_∞ is the length at which the growth of fish asymptotes, 'k' is the metabolic growth coefficient, and 't_0' is the initial condition parameter (point in time when the fish has zero length). The modern published estimates of L_∞ of snapper population in SNA 8 range from 528 mm to 709 mm (see Table 4.1 for details and sources). Modern age-length data for SNA 8 snapper were available for years 1973 to 2007 from the Ministry of Fisheries, New Zealand. A VBGF was fitted (using 'vonb' function from the UBCFC package in R (Martell 2005) to the modern age length data to obtain another estimate of the growth parameters L_∞ and k *and* t_o. The 'vonb' function uses a non-linear least square (nls) fitting function (R Development Core Team 2009) to estimate the VBGF parameters.

Table 4.1 VBGF parameters for modern snapper populations.

Serial number	L_∞ (in cm)	k	t0	Reference
1	63.2	0.138	-0.72	(Walsh et al. 2006)
2	70.9	0.113	-0.87	(Walsh et al. 2006)
3	66.3	0.125	-0.8	(Walsh et al. 2006)
4	66.2	0.143	-0.52	(Walsh et al. 2006)
5	65.8	0.13	-0.75	(Walsh et al. 2006)
6	57.4	0.17	-0.56	(Davies et al. 2003)
7	63.5	0.13	-1.07	(Davies et al. 2003)
8	56.2	0.18	-0.3	(Davies and McKenzie 2001)
9	52.8	0.21	-0.28	(Davies and McKenzie 2001)

Serial number	L_∞ (in cm)	k	t0	Reference
10	66.9	0.16	-0.11	(McKenzie et al. 1992; cited in Davies and McKenzie 2001)
11	58.6	0.1557	-1.016	VBGF fit estimate

4.2.2 Midden descriptions/archaeological data

Calculation of body length from midden data is a laborious process (Leach 2006). A series of allometric relationships between morphometric measurements (mainly of various snapper bones relative to snapper fork length) were estimated for a wide size range of modern specimens. More details about calculations and bones used for the allometric relationships can be found in Leach and Boocock (1995). The allometric relationships were used to estimate the corresponding fork lengths for the fish bones found in the middens. Age information corresponding to the length data was not available. In general, it is difficult to obtain age data from archaeological specimens. Otoliths are only rarely extracted from archaeological sites, and when they are, archaeological otoliths are very difficult to read annuli from under thin section (Foss Leach pers. comm.). However, in a study (Helen Neil pers. comm.) few (~10) snapper otoliths were obtained from a c.1400 AD midden at Hot water Beach in Hauraki Gulf and were used for a comparison of ancient and modern growth patterns. Their comparison based on standardized annual increment widths from modern and ancient otoliths did not indicate a difference in growth pattern between the ancient and modern otoliths (Helen Neil pers. comm.). Dr Larry Paul also echoed similar subjective impression based on his experience in otolith interpretation. Lengths corresponding to the ancient otoliths were not available, so it was not possible to gauge any age-length information from the ancient otoliths.

4.2.3 Growth parameters of the ancient population

All the growth parameter combinations of the modern population were tried as potential growth curves of the ancient population to see if the growth curves of modern population could explain the length frequency distribution in the ancient data.

Studies that have explored the issue of prehistoric impact on the New Zealand snapper fishery found no conclusive evidence for a "decline in mean fish size during the pre-European period" (Leach et al. 1997; Leach and Davidson 2001). They have, however, concluded on definite differences in mean-size of snapper caught in modern versus ancient times. The length frequency distributions based on archaeological samples were very different from those based on modern fish catch (Leach and Davidson 2000). The proportion of large (> 600 mm) fish was much higher in the ancient than in the modern samples (Figure 4.2). Hence, it was necessary to explore if growth curves other than modern published growth curves explained the higher proportion of large fish in the ancient population.

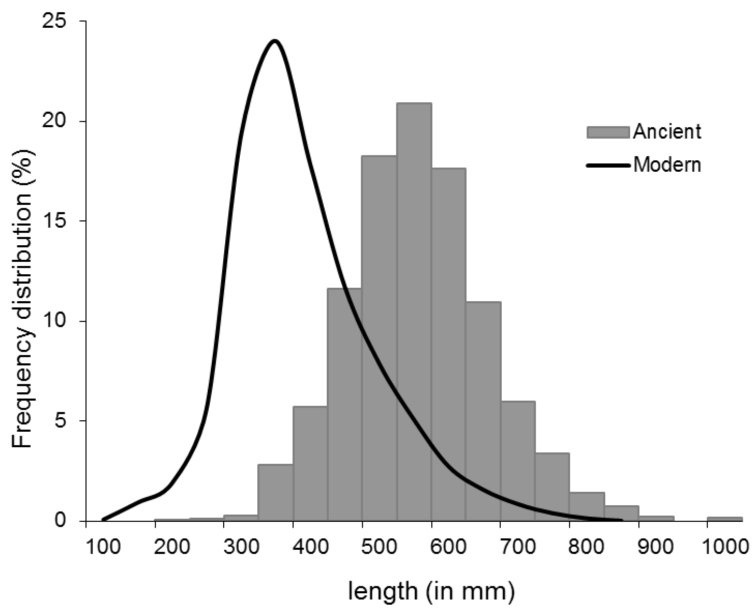

Figure 4.2 Comparison of length frequency data for ancient and modern snapper populations.

4.2.4 Candidate growth parameters of the ancient population

A range of L_∞ values from 660 mm to 1046 mm (at increments of 1mm) were chosen as potential candidate L_∞ values. The lower end of this range represented the upper quartile of the modern growth parameters while the upper end constituted the highest probable value of L_∞ calculated using the empirical formula:

(2) $L_\infty = L_{max} / 0.95$

Where, L_{max} is the maximum length of fish in the ancient data. Thus L_∞ for the ancient population smaller than the modern published values of L_∞ were not explored; this was because larger fish were observed in the ancient population. For each individual L_∞ value within this range, a corresponding '*k*' parameter was estimated by fitting a VBGF to the modern age length data to optimise for '*k*' alone ('t_0', was assumed to be -1yr). The L_∞ and *k* parameters thus calculated lie on an isocline (Figure 3a). By fitting to the modern age length data, it was assumed that the growth pattern of the snapper population had not changed between the ancient and modern times. This assumption was in agreement with the observations made by archaeologists who compared ancient and modern snapper otoliths. Thus the possibility of 'no change in growth pattern' was explored. It is possible that the *k* parameter for the ancient snapper was slightly overestimated because of size selective mortality in the modern snapper population. The implications of such a bias are discussed later in the chapter. Thus 386 candidate growth curves for the ancient population (Figure 4.3) were evaluated in the analysis.

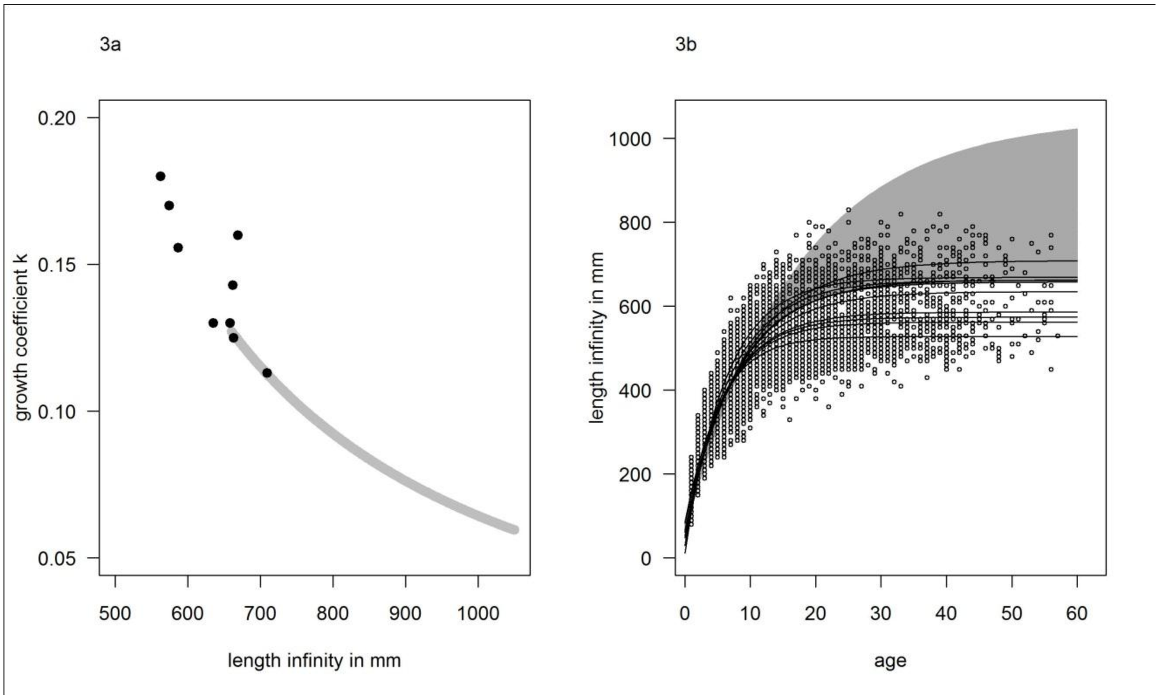

Figure 4.3 Candidate growth curves for ancient population.
3a. Modern* $\mathbf{L_\infty}$-k *combination in filled black dots and candidate* $\mathbf{L_\infty}$-k *combinations for ancient population in grey dots. 3b. Open black dots show modern age-length data. Black lines show modern published growth curves and the grey curves show the growth curves based on candidate growth parameters for the ancient population.

4.2.5 Estimation of mortality for ancient population

The length-frequency data of the fish remains found at the archaeological middens represent a mixture of several (unknown) age groups in the population. In order to fit these mixture distributions to length frequency data, a method developed by Macdonald and Pitcher (1979), updated by Macdonald and Green (1988), and rewritten for R (‘mixdist’ package) by Macdonald and Du (2004) was applied. The package applies a standard maximum likelihood estimation (MLE) method to calculate the mixing proportions (the proportion of each component age class in the total population) and the mean and standard deviations of each component distribution (Du 2002), thereby minimising the difference between the sum of the component distributions and the data length frequency. The R program requires a complete set of initial parameter values (i.e., the mean and standard deviation for each length at age, and the proportion of each age cohort in the population).

The initial parameters were set up as follows:

i. For each L_∞-k combination, the mean length at age was calculated using the VBGF equation. A constant coefficient of variation of 10% was applied to the mean length at age to calculate the standard deviations for mean length at age.
ii. The proportions of the age-bins were set to be equal (i.e. similar number of individuals in each age-bin). Initializing the proportions using any (other) survivorship schedule could amount to providing prior information about the total mortality in the ancient population, so the proportions were initialized at same value for all age-bins. A set of survivorship schedules were tested as alternate values to initialize the proportions to test if the results were sensitive to the starting values. The influence of different initial values for the proportions is discussed later in the chapter.
iii. During the fitting process, the mean lengths at age were fixed. Thus, the fit was achieved by changing the proportions of each age group in the population. An example of a fit is shown in Figure 4.4a.

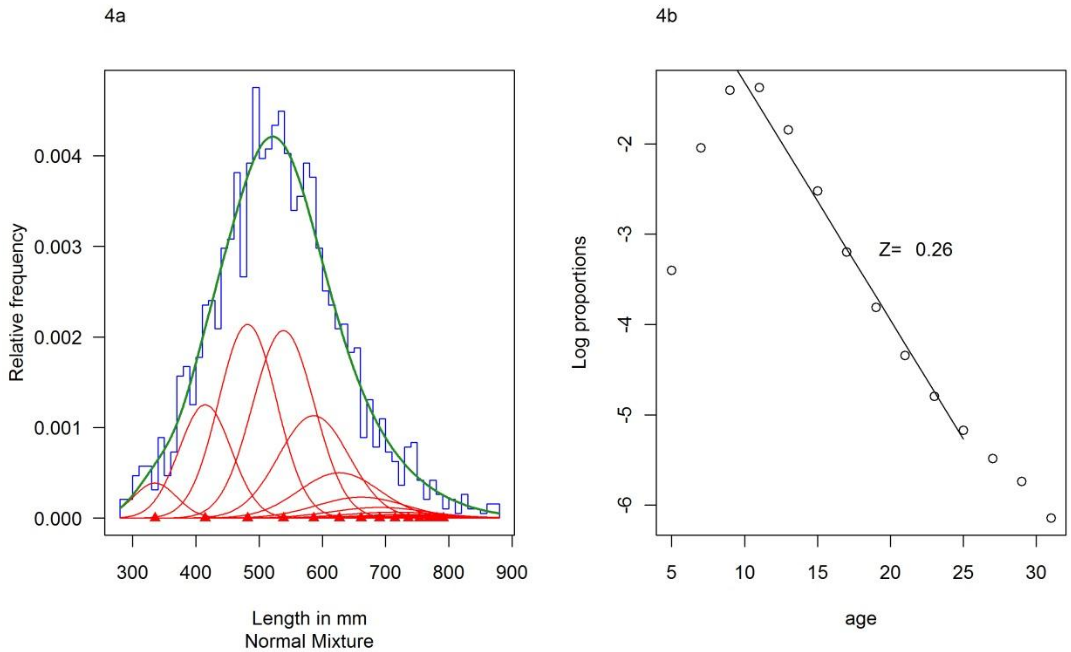

Figure 4.4 Estimation of total mortality (Z)
4a. An example of fit to mixture distribution. The blue bars are the histogram of the lengths from archaeological data, the green line is the cumulative length frequency distribution, and the red lines are the component distributions for each age. 4b. An example of the graph of log of proportions against age (L$_\infty$=850mm and k = 0.084). Total mortality Z (0.26) was calculated as the negative slope of regression.

The total mortality (Z) corresponding to each L_∞-k combination was estimated as the negative slope of the fitted proportions. Two problems with length frequency analysis are that sometimes knowledge about number of age-classes is not available, and that several combinations of parameters could fit the length frequency distribution (Schnute and Fournier 1980). This author experimented with several combinations of number of age-bins. At higher ages (>35), the log proportions did not decline in a straight line but became a curve (i.e. the proportions of older fish were being overestimated); so ages>35 were not used as age-bins. In the analysis presented here 15 age classes were used (alternate age from age 3 to 31) in the fitting process. Fewer age bins (<15) were not used because grouping ages together could lead to loss of information in the length frequency. The goal was to obtain an estimate of Z corresponding to each L_∞-k combination.

4.2.6 Proportion of large fish in the population

The proportion of fish of different lengths in a population depends on the growth curve of the species and the survivorship at age (the probability fish surviving to each age). The proportions of fish above three length (600 mm, 750 mm, and 800 mm) values in the population were calculated using both the modern published and the candidate growth curves. First the proportion of fish above the three length levels in each age class was calculated based on length at age and standard deviation of length at age (for example P_{800_20} represents fish at age 20 which are greater than 800 mm). The sum-product of survivorship schedule and the proportion of fish above the three length levels in each age class gave the proportion of fish above the three length levels in the population.

(3) Survivorship (l_{age}) is the probability of surviving to each age and is calculated from estimates of total mortality as follows: $l_{age} = \exp(-Z_{(age-1)})$. For each candidate growth curve, the Z estimated in the earlier section was used to calculate the survivorship.

(4) Proportion of fish greater than 800 mm: $P_{800} = P_{800\_age} . l_{age}$

It was assumed that after the fish fully recruited to the fishery, the proportion of fish of different lengths in the data represented their ratios in the fully recruited part of the population. In the estimation of total mortality it was found that fish above 600 mm had fully recruited to the fishery[23]. The proportions of fish greater than lengths 750 mm and 800 mm in the fully recruited part of the population were estimated. For example, the proportion of fish greater than 800 mm in the fully recruited part of the population P_{fr_800} was calculated as:

(5) $P_{fr\_800} = P_{800} / P_{600}$

[23] Figure 4.4b shows that fish above age 9 have fully recruited to the fishery, the log of proportions at age starts descending. Figure 4.3b shows that age 9 fish correspond to length around 500 mm, thus it is judged that fish above 600 mm should be fully recruited to the fishery. However, this author is not assuming that 600 mm is the smallest length at which the fish fully recruit to the fishery.

The estimated proportions of large fish (fully recruited) were compared with their respective proportions in the ancient data. The comparisons that fell within the proportion of large fish observed in the ancient data ± 10% were chosen as the probable L_∞-k combination for the ancient population.

By choosing the estimated Z to model mortality at all ages in the population this author made an assumption that the total mortality remained constant throughout all life stages. The results of P_{600}, P_{700}, and P_{800} were highly sensitive to different formulations of Z (i.e. combination of natural mortality (M) and fishing mortality (F) at smaller ages. But the results of P_{fr_600}, P_{fr_700}, and P_{fr_800} were not sensitive to the assumptions on Z (for the time before the fish fully recruited to the fishery) because after the fish fully recruited to the population, the decline in proportions was dependent on the estimated Z.

4.2.7 Ancient population size

An equilibrium age-structure model was used to estimate the ancient population size. The product of biomass per recruit and equilibrium number of recruits gives the equilibrium size of the population. Note that emphasis should not be placed on the precision or accuracy of the resulting estimates. Rather the intent was to provide approximate estimates of ancient population size in order to facilitate a discussion the difference between the ancient and the modern population biomass.

4.2.7.1 Biomass per recruit

The biomass per recruit (B/R) is calculated as the sum-product of weight at age and survivorship at age. Paul (1992) found the maximum age of a snapper to equal 60 years and this estimate was used for the analysis. The estimates of Z from the earlier section were used to model survivorship. I only had estimates of Z for the ancient population; I did not have any estimates of natural mortality (M) or fishing mortality (F). To split the estimated Z into its components and to estimate the probable ranges of B/R in the ancient population, it was assumed that M for the ancient population could range between the following two levels:

i. M=0.057 (F=Z-M) (M was equal to the natural mortality on the modern population (Davies et al. 2006; MFish New Zealand 2007),

ii. M=0.114 (F=Z-M) (M was double the natural mortality on modern population.)

Overestimating survivorship would lead to a higher estimate of biomass per recruit and therefore higher population biomass; underestimating survivorship would lead to a lower estimate of biomass per recruit and lower population biomass. In the estimation of total mortality in the earlier section, it was observed that the fish became fully vulnerable to fishing at around age 9 for all the candidate growth curves tested (see example for one growth curve in Figure 4.4b). The vulnerability at age of the ancient snapper population was modelled using a logistic curve (Figure 4.5); the length at which 50% fish were vulnerable (l_{50}) was 450mm.

(6) Vulnerability to fishing at age: $$vul_{age} = \frac{1}{1+\exp\left(\dfrac{-(l_{age}-l_{50})}{sig}\right)}$$

where, *sig* represents the steepness of the curve

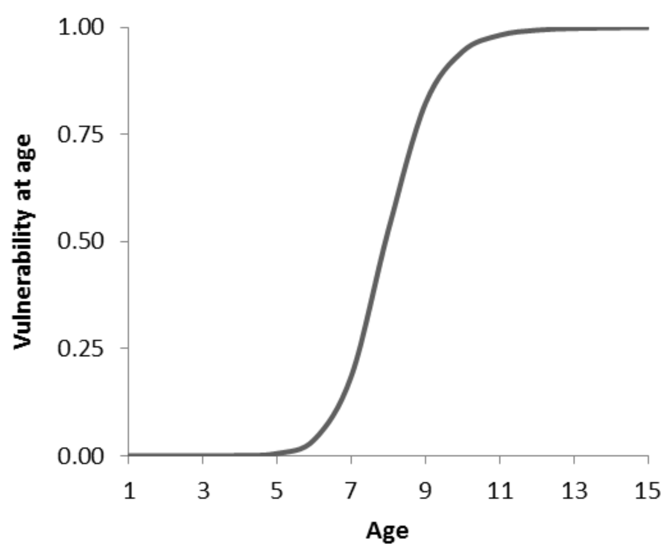

Figure 4.5 Vulnerability at age to fishing

(7) Fishing mortality at age: $F_{age} = F.vul_{age}$

(8) Survivorship at age: At age =1 $l_f = 1$

$$\text{At age}>1 \quad l_f = l_{f(age-1)} \exp(-M - F_{(age-1)})$$

(9) Weight (W_{age}) at age according to L-W relationship: $W_{age} = a(l_{age})^b$

Where, *a* was set to 0.0447 and *b* to 2.793 based on information in Fishbase (Paul 1976; as recorded in Freose and Pauly 2010) and l_{age} was calculated according to eq 1.

The B/R was calculated as the sum-product of the weight at age and the survivorship schedule.

(10) Biomass per Recruit: $B/R = W_{age}.l_{age}$

4.2.7.2 Stock and Recruitment

It was assumed that both the modern and the ancient population follow the same stock recruitment curve. Recruitment was described using the classic Beverton and Holt stock recruitment (BH-SR) pattern (Myers 2001):

(11) $R = \alpha S/(1+\beta S)$

where, R represents recruitment to age 1, and S breeding stock size. Parameter α is the slope at the origin, "α increases the height of the asymptote and reduces the curvature, and β increases the rate of approach to the asymptote" (Jennings et al. 2001). The 'α' was set to 1.287 based on Myers et al. (1999)[24]. The parameter β was calculated using the relation:

(12) $R_0 = \alpha / \beta$

R_0 was obtained from the plot of species summary for the New Zealand snapper from SNA 8 by Myers et al. (1995).

According to Beverton and Holt SR pattern, with an increase in number of spawners, the number of recruits increases to an asymptote. The equilibrium mean recruitment at a given level of exploitation is calculated by the formula (refer Walters and Martell (2004) for more details):

(13) $R_e = (\alpha\phi_e - 1) / \beta\phi_e$

where, R_e is the mean recruitment, and ϕ_e is the fecundity incidence function, which represents the fecundity per recruit in the population and is calculated from maturity at age and fecundity at age as follows:

(14) Maturity at age $mat_{age} = \dfrac{1}{1 + \exp\left(\dfrac{-(age - age_{mat})}{sig}\right)}$

(15) Fecundity at age $fec_{age} = w_{age}.mat_{age}$

(16) Fecundity per Recruit $\phi_e = fec_{age}.l_{age}$

[24] the 'α' value is estimated as â/$SPR_{F=0}$. (Myers et al. 1999) used $SPR_{F=0}$ = 50.956 (from Annala and Sullivan 1996) and estimated the â as 65.6. Annala, J.H. & Sullivan, K.J. (Comps) 1996: Report from the Fishery Assessment Plenary, April-May 1996: stock assessments and yield estimates. 308 p. (Unpublished report held in NIWA library, Welington). Enquiries: N.M. Davies, NIWA, P.O. Box 1043, Whangarei, New Zealand. Email: n.davies@niwa.cri.nz

It was assumed that above the age at maturity, the fecundity was proportional to body weight. Age at maturity was set at 3 years based on observations that snapper in SNA 8 mature at age 3 (Davies et al. 2006).

The population biomass was calculated as the product of B/R and mean recruitment

(17) $Biomass = (B/R) * R_e$

4.3 Results

4.3.1 Growth parameters of the ancient population

Corresponding to higher candidate L_∞ values for the ancient population, higher estimates of total mortality Z were obtained. Thus estimates of mortality were correlated with L_∞. The modern published growth curves were not able to explain the proportion of large fish (>750 mm and 800 mm) in the recruited fish population (Figure 4.6a and 4.6b). The proportion of fish greater than 750 mm was explained by candidate growth curves with L_∞ in the range from 767 to 853 mm; L_∞ higher than 853 mm overestimated the proportion of fish larger than 750 mm. The proportion of fish greater than 800 mm was explained by candidate growth curves with L_∞ in the range from 826 to 900 mm. When the uncertainty around the estimate of Z was considered, the range increased and included fish from 735 mm to 1050 mm.

For the fitting process used to estimate the Z, the initial values for the proportions at age were set to be equal. When different initial values for the proportions at age were tested[25]

[25] The different starting values were survivorship schedules corresponding to Z within the range 0.14 to 0.34. When the starting values were changed, different estimates of fitted proportions were obtained. For candidate growth curves in the range L_∞ 660 mm to 827 mm, the best fits were obtained when the initial proportions for all age classes were set to be equal. For candidate growth curves with L_∞ values 828 mm and above, better fits were obtained when different survivorship schedules were used as initializing values. These better fits for candidate growth curves with L_∞ values 828 mm and above resulted in slightly different (~1%) estimates of Z.

in the fitting process slightly different results were obtained; the proportion of fish greater than 750 mm were explained by candidate growth curves with L_∞ in the range from 767 to 841 mm; the proportion of fish greater than 800 mm were explained by candidate growth curves with L_∞ in the range from 826 to 884 mm. Since the difference in results was not large, the results obtained based on the earlier initialization of proportions (all age classes set to be equal) were used to calculate the estimates of biomass presented in the following sections.

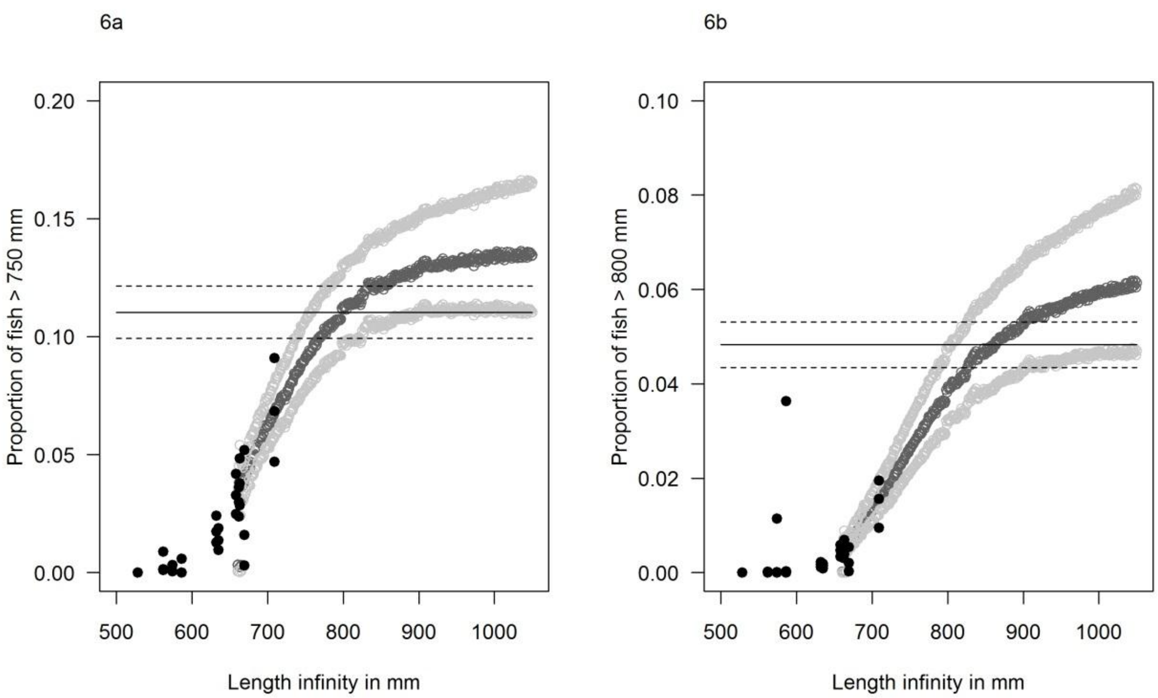

Figure 4.6 Proportions of large fish (>750 mm and 800 mm) estimated using modern published and candidate ancient growth curves.
The dark grey dots show the proportions estimated by the total mortality estimates from fitting; the light grey dots are obtained using the upper and lower bounds of the total mortality estimates. The black dots are estimates using modern published growth curves. The solid horizontal line is the proportion of fish above the corresponding length in the ancient data, and the dashed lines are proportions 10% above and below the solid line.

4.3.2 Comparison of modern and ancient population biomass

Current estimates for mean of snapper biomass calculated based on several stock assessment models for SNA 8 range from 11,200 to 12,900 tonnes with confidence limits ranging from 9,600 to 16,500 tonnes (Davies et al. 2006). Trawl survey estimates for year

2002 put the value at 10,442 tonnes with a coefficient of variation of 0.12 (Davies et al. 2006).

The range of the ancient population size depended on the growth parameters combinations. A larger L_{∞}- smaller k combination yielded a lower ancient population biomass estimate compared to a smaller L_{∞}- larger k combination (Figure 4.7). Results showed that the ancient population size (20,000 to 38,000 tonnes) was about 2 to 4 times higher than the modern population. Very little change (factor of 1.01) in the equilibrium recruitment was observed, and the factor by which the ancient population was higher than the modern population depended mainly on the ratio of biomass per recruit in ancient versus modern times.

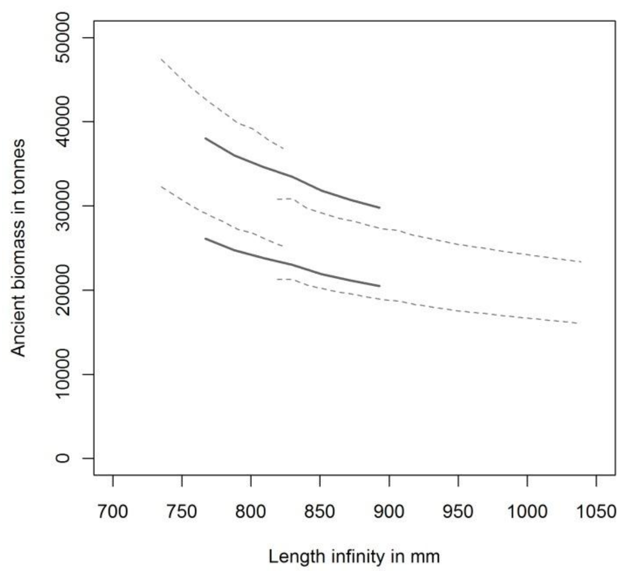

Figure 4.7 Ancient snapper population biomass.
The biomass estimates are plotted against the candidate growth parameters for the ancient population. The solid line shows the estimates corresponding to estimated Z (corresponding to dark grey dots in Figures 4.6a and 4.6b). The dashed lines show the estimates including the lower and upper bound on the estimates of Z (corresponding to light grey dots in Figures 4.6a and 4.6b).

4.4 Discussion

Steady state conditions were assumed in the analysis. The period from 1500 AD – 1770 AD is known as the classic Maori culture at which time the Maori population was at its peak. The midden data belonging to 1400 AD – 1500 AD were from a time period when the Maori population and their use of marine resources (including snapper) was changing. The snapper fishery was commercially exploited since the mid-1800s, with catches highest in the period 1960 to 1980 (Maunder and Starr 2001). The modern data belong to the period 1973 to 2007 AD, also a period which saw change in the exploitation pattern. Hence, in the short term perspective, population levels for both time periods were not in steady state. However, from a long term perspective and for the comparison of populations 600 years apart in time, the snapper populations could be assumed to be in steady state.

4.4.1 Ancient growth parameters

L_∞ was estimated to range between 767 mm and 900 mm, while modern published values range from 528 mm to 709 mm. The lower estimate (767 mm) is only 6 cm higher L_∞ than the highest modern published estimate for L_∞. The question is whether the difference between ancient and modern L_∞ is the result of evolutionary change or an artefact of the VBGF. If the fishing pressure on a population is high, then the proportion of large individuals in the population would be expected to be low (i.e., modern day scenario). "Extirpation of large specimens by intensive fishing" has made it difficult to estimate the maximum size of fish species (Binohlan and Froese 2009). Fitting a VBGF curve to age-length data from such a fishery catch can bias the L_∞ downward and the growth parameter k upward. The lower modern L_∞ could, therefore, be an artefact of fitting to data largely containing small age groups and rare large fish. The largest fish seen in the modern age length sample was 830 mm, but the fraction of such large fish was very small. So it is possible that the lower published estimates of L_∞ are a result of rarity of large individuals. Also growth curves of snapper followed a very similar pattern over the first 8 years of life and appeared almost identical in both the modern published and the candidate growth curves for the ancient population (Figure 4.2b). Therefore, it is possible that the L_∞ of modern snapper are higher than previously published estimates.

This author was thus unable to explore if the higher *k* seen in the modern estimations is the result of non-availability of large fish in the sample or the actual increase in the growth rate of the snapper. The more important question though is whether this barely discernable difference in growth parameters is of any significance for real management purposes. Answering the question would require analysis of response of population biomass at different combinations of growth parameters. Higher mortality risk has been shown to be associated with faster growing species (Lankford et al. 2001); however, these analyses refer to comparisons of different species or across same species at different latitudes and not to changes in growth in the same fish stocks. Some preliminary analysis indicates that if every other parameter remains the same, a higher *k* value would lead to a higher estimate of population biomass; a higher k would also underestimate the depletion of the population from the unfished level. But the parameter *k* is usually correlated with L_∞; thus, a sweeping generalization that a higher *k* could over-estimate the stock status cannot be made. An in-depth analysis is required in this area before any conclusions can be made.

As mentioned earlier, otoliths are difficult to extract from archaeological sites and it is also difficult to read annuli from archaeological otoliths. No corresponding age information was available with the length data for the ancient sample. If after extraction from middens, otoliths were weighed before sectioning then estimates of corresponding lengths (with limited confidence) could be made based on the otolith weight. Availability of some length at age would help validate the results of a study like ours. The authors, therefore, suggest that midden otoliths be weighed before being sectioned for further study, so that the length or weight of the fish (to which to otolith belonged) could be predicted.

4.4.2 Ancient population biomass

The ancient population biomass was estimated to be about 2 to 4 times higher than the modern snapper population biomass. The contribution to the difference in biomass was mainly from difference in biomass per recruit. The ratio between the equilibrium recruitment levels was close to unity. A relationship between temperature and recruitment in New Zealand snapper has been described (Paul 1976; Maunder and Starr 1988; Francis 1993; Maunder and Starr 2001). The dependence on temperature is an indication that recruitment was not dependent on stock size (Annala and Sullivan 1997; cited in Maunder and Starr 2001), indicating that spawning stocks have not fallen to levels at which the impact of a decline in spawners corresponds to lower levels of recruitment.

Annala (1994) as cited in Leach (2006) estimated the B_0 (virgin biomass or unfished biomass) of snapper in SNA 8 to be equal to 73,200 tonnes. A number of stock assessment models estimated the mean B_0 for SNA 8 for snapper to range between 117,000 and 135,000 tonnes (Davies et al. 2006) with lower and upper confidence intervals in the range from 113,000 to 142,000 tonnes. The stock assessment models assumed that in 1931 the population was in "unexploited equilibrium". The results for unfished biomass in the chapter were more uncertain (50,000 to 150,000 tonnes for L_∞ 767 mm and 50,000 to 190,000 tonnes for L_∞ 900 mm) depending on the assumption of natural mortality in the analysis. The lower values were obtained when natural mortality was twice the natural mortality on the modern population, and the higher values were obtained when natural mortality on the ancient population was the same as the natural mortality on the current population. The most important parameter influencing the results of population biomass was the natural mortality on the ancient population. Therefore, when trying to estimate the carrying capacity of a population in an ecosystem, it is necessary to understand the natural mortality experienced by the population.

The ancient system probably had higher abundance of predators of snapper, therefore, a higher natural mortality than the modern snapper. Under this circumstance, if the fishing mortality on the modern population is removed, the population could probably rebuild to levels higher than the levels observed in the history of the population. The unfished biomass 'B_0' is relative to the natural mortality. If the natural mortality has not changed with time, only then the B_0 would be equal to the population size at which the population was before the fishery began. However, when several predator species are exploited, the natural mortality constantly changes resulting in a change in the carrying capacity of the species in the ecosystem.

4.5 Conclusion

The results based on the proportion of large fish show that the L_∞ of the ancient population (767 mm to 900 mm) was higher than the published values of L_∞ for the modern population (528 mm to 709 mm). When estimating growth parameters, caution should be exercised against over-estimating k and under-estimating L_∞. The ancient snapper population in Maori times was 2 to 4 times larger than the modern snapper population. The estimates of unfished biomass (5 to 20 times larger) were highly influenced by the assumption on natural mortality on the ancient population and the uncertainty on the growth parameters in the analysis. If the natural mortality on the modern population is lower than the levels on the ancient population, then the modern population might rebuild to higher levels than the ancient population. The carrying capacity of the current ecosystem could also have changed due to factors such as loss of habitat, pollution, change in biomass of prey species, and competition. When trying to base rebuilding targets on unexploited levels of stock, the changes that have happened in the ecosystem should be evaluated in addition to evaluating the effect of removal of fishing pressure from the species.

4.6 References

Annala J. H. 1994. Draft reports from the Fishery Assessment Working Groups. February-March 1994. Unpublished report, MAF Fisheries.

Annala J. H., K. J. Sullivan. 1997. Report from the Fishery Assessment Plenary, May 1997: stock assessments and yield estimates. Unpublished report held in NIWA library, Wellington. 409p.

Binohlan C., R. Froese. 2009. Empirical equations for estimating maximum length from length at first maturity. Journal of Applied Ichthyology **25**(5):611-613.

Braje T. 2009. Modern Oceans, Ancient sites – archaeology and marine conservation on San Miguel Island, California. University of Utah Press, Salt Lake City, US. 160p.

Campbell L. M., N. J. Gray, E. L. Hazen, and J. M. Shackeroff. 2009. Beyond baselines: rethinking priorities for ocean conservation. Ecology and Society **14**(1):14.

Davies N. M., J. R. McKenzie. 2001. Assessment of the SNA 8 stock for the 1999-2000 fishing year. New Zealand Fisheries Assessment Report. **54**:1-57.

Davies N. M., J. R. McKenzie, and D. J. Gilbert. 2006. Assessment of the SNA 8 stock for the 2003-04 year. New Zealand Fisheries Assessment Report. **9**:1-58.

Davies N. M., B. Hartill, and C. Walsh. 2003. A review of methods used to estimate catch-at-age and growth in SNA 1 and SNA 8. New Zealand Fisheries Assessment Report. **10**:1-63.

Du Juan. 2002. Combined algorithms for fitting constrained estimation of finite mixture distributions with grouped data and conditional data. Master thesis, McMaster University, Canada.

Francis M. P. 1993. Does water temperature determine year class strength in New Zealand snapper (*Pagrus auratus*, Sparidae)? Fisheries Oceanography **2**(2):65-72.

Freose R., D. Pauly. 2010. FishBase. World Wide Web electronic publication. Available: <www.fishbase.org>, version (05/2010).

Jackson J. B. C., M. X. Kirby, W. H. Berger, K. A. Bjorndal, L. W. Botsford, B. J. Bourque, R. H. Bradbury, R. Cooke, Jon Erlandson, J. A. Estes, T. P. Hughes, S. Kidwell, C. B. Lange, H. S. Lenihan, J. M. Pandolfi, C. H. Peterson, R. S. Steneck, M. J. Tegner, and R. R. Warner. 2001. Historical overfishing and the recent collapse of coastal ecosystems. Science **293**(5530):629-638.

Jennings S., M. J. Kaiser, and J. D. Reynolds. 2001. Marine fisheries ecology. Blackwell Publications, Oxford.

Lankford Jr, T. E., J. M. Billerbeck, and D. O. Conover. 2001. Evolution of intrinsic growth and energy acquisition rates. II. Trade-Offs with vulnerability to predation in *Menidia menidia*. Evolution **55**(9):1873-1881.

Leach B. F., J. M. Davidson, and L. M. Horwood. 1997. Prehistoric Maori fishermen at Kokohuia, Hokianga Harbour, Northland, New Zealand. Man and Culture in Oceania **13**:99-116.

Leach F., J. Davidson. 2001. The use of size-frequency diagrams to characterize prehistoric fish catches and to assess human impact on inshore fisheries. International Journal of Osteoarchaeology **11**(1-2):150-162.

Leach F., A. Boocock. 1995. Estimation of live fish catches from archaeological bone fragments of the New Zealand snapper *Pagrus auratus*. Tuhinga: Records of the Museum of New Zealand **3**:1-28.

Leach F. 2006. Fishing in Pre-European New Zealand. Jointly published by New Zealand Journal of Archaeology (special publication) and Archaeofauna, New Zealand. 359p.

Leach F., J. Davidson. 2000. Pre-European catches of snapper (*Pagrus auratus*) in Northern New Zealand. Journal of Archaeological Science **27**(6):509-522.

Love M. S., M. Yoklavich, and L. K. Thorsteinson. 2002. The rockfishes of the Northeast Pacific. University of California Press, California.

Macdonald P. D. M., T. J. Pitcher. 1979. Age-groups from size-frequency data: a versatile and efficient method of analyzing distribution mixtures. Journal of the Fisheries Research Board of Canada **36**:987-1001.

Macdonald P. D. M., J. Du. 2004. Mixdist: Mixture Distribution Models. R package version 0.5-1. Available: http://www.r-project.org; http://www.math.mcmaster.ca/peter/mix/mix.html [Accessed on May 15, 2008].

Macdonald P. D. M., P. E. J. Green. 1988. User's guide to program MIX: An interactive program for fitting mixtures of distributions. Ichthus data systems. ISBN 0-9692305-1-6.

Martell S. 2005. An introduction to R for fisheries modeling. Available: s.martell@fisheries.ubc.ca, Fisheries Center UBC. Canada.

Maunder M. N., P. J. Starr. 2001. Bayesian assessment of the SNA1 snapper (*Pagrus auratus*) stock on the North-east coast of New Zealand. New Zealand Journal of Marine and Freshwater Research **35**(1):87-110.

Maunder M. N., P. J. Starr. 1988. Validating the Hauraki Gulf snapper pre-recruit trawl surveys and temperature recruitment relationship using catch at age analysis with auxiliary information. New Zealand Fisheries Research Document **94**(15):1-23.

McKechnie I. 2005. Five thousand years of fishing at a shell midden in the Broken Group Islands, Barclay Sound, British Columbia. Master thesis, Simon Fraser University, Canada.

McKenzie J. R., M. Haddon, C. Walsh, S. Carter, and N. M. Davies. 1992. Summary findings of commercial snapper (*Pagrus auratus*) market sampling from SNA 1 and SNA 8 (1989-1992). MAF Fisheries North Internal Report No. 8. 63p. (unpublished report held in NIWA library, Wellington).

MFish New Zealand. 2007. Snapper - Plenary report. Ministry of Fisheries New Zealand. **2007**:839:875.

Myers R. A., K. G. Bowen, and N. J. Barrowman. 1999. Maximum reproductive rate of fish at low population size. Canadian Journal of Fisheries and Aquatic Sciences **56**(12):2404-2419.

Myers R. A., J. Bridson, and N. J. Barrowman. 1995. Summary of worldwide stock and recruitment data. Canadian Technical Report on Fisheries and Aquatic Sciences. **2024**:1-327. Available: <www.mscs.dal.ca/~myers/welcome.html> [Accessed on August 11,2008]

Myers R. A. 2001. Stock and recruitment: generalizations about maximum reproductive rate, density dependence, and variability using meta-analytic approaches. ICES Journal of Marine Science **58**(5):937-951.

Paul L. J. 1992. Age and growth studies of New Zealand marine fishes, 1921-90: A review. Australian Journal of Marine and Freshwater Research **43**:879–912.

Paul L. J. 1976. A study on age, growth, and population structure of the snapper, *Chrysophrys auratus* (Forster), in the Hauraki Gulf, New Zealand. Fisheries Research Bulletin **13**:62.

Pauly D., V. Christensen, J. Dalsgaard, R. Froese, and F. Torres Jr. 1998. Fishing down marine food webs. Science **279**(5352):860-863.

Pitcher T. J., M. E. Lam. 2010. Fishful thinking: rhetoric, reality, and the sea before us. Ecology and Society **15**(2):1-12.

Pitcher T. J. 2001. Fisheries managed to rebuild ecosystems? Reconstructing the past to salvage the future. Ecological Applications **11**(2):601-617.

R Development Core Team. 2009. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. ISBN 3-900051-07-0, Available: <www.R-project.org>.

Roberts C. M. 2007. The unnatural history of the sea. Island Press, Washington DC. 435p.

Rosenberg A. A., W. J. Bolster, K. E. Alexander, W. B. Leavenworth, A. B. Cooper, and M. G. McKenzie. 2005. The history of ocean resources: modeling cod biomass using historical records. Frontiers in Ecology and the Environment **3**(2):78-84.

Schnute J., D. Fournier. 1980. A new approach to length-frequency analysis: growth structure. Canadian Journal of Fisheries and Aquatic Sciences **37**(9):1337-1351.

Walsh C., J. McKenzie, and H. Armiger. 2006. Spatial and temporal patterns in snapper length and age composition and movement, west coast North Island, New Zealand. New Zealand Fisheries Assessment Report. **6**:1-57.

Walters C. J., S. J. D. Martell. 2004. Fisheries ecology and management. Princeton University Press, New Jersey. 399p.

Wilmshurst J. M., A. J. Anderson, T. F. G. Higham, and T. H. Worthy. 2008. Dating the late prehistoric dispersal of Polynesians to New Zealand using the commensal Pacific rat. Proceedings of the National Academy of Sciences **105**(22):7676-7680.

5 Evaluation of Restoration Goals for Raja Ampat Coral Reef Ecosystem Using Influence Diagram Modeling[26]

5.1 Introduction

5.1.1 Need for marine restoration

Overfishing has led to declines of fish populations (Myers et al. 1996; Rose and Kulka 1999; Morris et al. 2000; Dulvy et al. 2003; Hutchings and Reynolds 2004) and has pushed several marine ecosystems towards collapse (Hughes 1994; Pauly et al. 1998; Jackson et al. 2001; Pandolfi et al. 2003). With the increase in marine resource declines, the emphasis on restoration has increased (Pitcher 2001; Fox et al. 2003; Russ and Alcala 2003; Lotze et al. 2006). Restoration efforts usually involve modifications to fishing gear, season length, quota allocation, species restrictions, or establishment of marine protected areas. The process of restoration is not easy because all manifestations of restoration require current extractions from the ecosystem to be limited, suspended or stopped. Lack of understanding between the fishers and the management agencies have often led to "adversarial relations" (Kaplan and McCay 2004) rendering any positive step towards restoration difficult (Charles 2002). Studies have highlighted that achieving "health and stewardship of coastal and marine seas" requires incorporating the concerns of different stakeholders (Leslie and McLeod 2007). Pairing marine tourism with fisheries restoration could offer options to protect both the ecosystem and the associated livelihoods

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[26] A version of this chapter will be submitted for publication. Varkey, D. A., Pitcher, T. J., McAllister, M., Sumaila, R. Restoration strategies for coral reef ecosystems – combining fisheries, tourism and conservation utilities using a Bayesian influence diagram model.

(Brunnschweiler 2009). In a comparison of stakeholders priorities in a Caribbean marine park, it was seen that all the stakeholders, from village council, fishers, recreational users and members of the assembly, weighted ecosystem health higher than economic and social concerns (Brown et al. 2001) but another comparison of perceptions in Florida Keys showed that these different groups often had different approaches to the use of the resource (Suman et al. 1999). Therefore, restoration efforts need to focus on management policies, which can incorporate the differing perspectives and attitudes of multiple stakeholders.

5.1.2 Raja Ampat coral reef ecosystem

Raja Ampat is an archipelago located inside Southeast Asian coral triangle, a hotspot of marine biodiversity and an area known for the high diversity and abundance of coral reef and fish species (McKenna et al. 2002; Donnelly et al. 2003). About 24,000 fishers depend on the reef and the adjacent coastal waters for their livelihood (Dohar and Anggraeni 2007). Analysis of fisher perceptions (Ainsworth et al. 2008a) and previous ecosystem modeling (Ainsworth et al. 2007) of the coral reef ecosystem shows that many fish populations in the region have declined due to high fishing pressure. Several non-governmental organisations (NGOs) working in the area publish pamphlets in the local language 'Bahasa' (Rabu 2006) containing information for general awareness of the fishers. Involvement with the NGOs has led to increased understanding of the harmful effects of destructive fishing. So, the fishers also are interested in protecting the ecosystem.

As a political entity, Raja Ampat is a relatively new regency (the administrative hierarchy of a regency is one level below the province and roughly corresponds to a district) in the province of Papua in eastern Indonesia. The region has also been declared a 'Kabupaten Bahari' (maritime regency) (Conservation International 2008) to safeguard the marine resource and to encourage tourism to the region. These new developments have led to increased interest in fisheries restoration in the region. The process for co-managing fisheries and tourism in Raja Ampat is based on management guidelines designed for and observed to be successful in Bunaken National Park located in Sulawesi Islands of

Indonesia (Erdmann et al. 2004). The Bunaken National Park was established in 1991 and initially it was only a "paper park"; after many years of mismanagement, which also saw increases in destructive fishing and declines in fish populations, an independent park management unit was established in 1997 (Erdmann et al. 2004). In a span of 5 years, in an iterative process of cooperation and consultation between the operators in the tourism industry, the local government, villagers, and park management body, a consensus was achieved on the management methods for the park. For in-depth understanding of the process, which brought all stakeholders on board, interested readers are encouraged to refer to the report by Erdmann et al. (2004).

5.1.3 Combining ecosystem model and Bayesian belief network

The restoration strategy for the coral reef ecosystem needs to balance the complex inter-species relations, expectations of the fishers, the needs of tourism industries, and the needs of conservationists. Instead of searching for the best policy, the more practical approach is probably to search for the most robust policy: this is the policy that will be suitable under the main sources of uncertainty and differences in utility functions between different interest groups. In this chapter, effort is made to identify restoration strategies for the Raja Ampat coral reef ecosystem that can be robust under different ecosystem states, different levels of tourism development, and different levels of interest in conservation. Bayesian influence diagrams, a special application of Bayesian Belief Networks (BBN) (Jensen 1997; Howard and Matheson 2005), are used to combine utilities of the different stakeholder groups dependent on Raja Ampat coral reef ecosystem.

Influence diagrams usually have one or more decision variables, which are informed by a combination of "knowledge" and "action" variables (Kuikka et al. 1999). Each variable in the BBN is associated with a set of probability tables for different states or values of the variable. The states of the ecosystem and fisheries catches are modeled using an Ecopath with Ecosim ecosystem simulation model for the Raja Ampat coral reef ecosystem. Other authors have used influence diagrams to model decision making in fisheries, for example, to decide on a robust management strategy when faced with environment driven uncertainty in cod recruitment (Kuikka et al. 1999); or to combine biological, social, and operational objectives for managing a herring fishery in the Bay of Fundy (Lane and Stephenson 1998). BBN have been used to decide on best allocation of resources available for management (Mantyniemi et al. 2009) and to compare different policies with respect to fisher commitment (Haapasaari et al. 2007).

5.2 Methods

5.2.1 Ecopath with Ecosim

The ecosystem simulation model was built using Ecopath with Ecosim (EwE) software. EwE is a mass balance food web simulation model that acts as a thermodynamic accounting system for marine ecosystems. Ecopath is a static snapshot of the system (Christensen 1992) that maps the energy flows in the system while Ecosim allows modeling of species composition changes as fishing effort varies over time (Walters et al. 1997). Marine flora and fauna are aggregated into functional groups based on similarity in their life history and trophic behaviour. The flows between groups are a result of predator-prey (predation mortality) interactions among functional groups and fishing mortality on the species. The EwE model (Ainsworth et al. 2008b presented in Appendix F) is used to explore various fishing restriction scenarios for the restoration of the system; the results in biomass, catch, and revenue are used to inform the probability tables needed for the bayesian influence diagram. However, readers interested in greater detail on the EwE models are referred to the online technical reports (Ainsworth et al. 2007) and (Ainsworth et al. 2008c) (see Appendix F).

5.2.2 Model structure of the influence diagram

The influence diagram is presented in Figure 5.1 and details of its component nodes (variables) are presented in the paragraphs below.

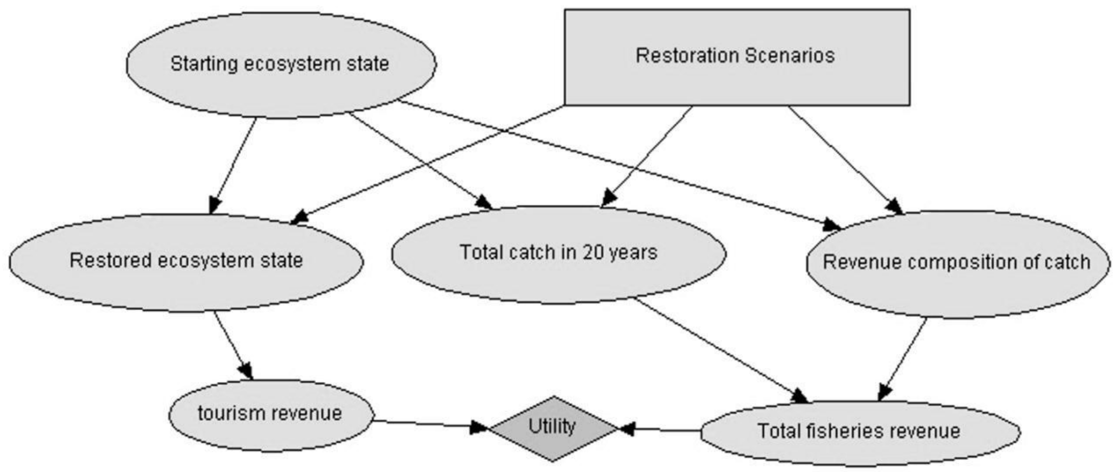

Figure 5.1 Structure of influence diagram.
The rectangular box shows the decision variable (restoration scenarios). The ellipses show all the other variables, unconditioned and conditioned that lead to the calculation of the utility. The different decisions (restoration scenarios) are compared based on the final utility values obtained.

5.2.2.1 Ecosystem restoration scenarios

The decision node 'Ecosystem restoration scenarios' consists of a group of fishing restriction scenarios. The restoration scenarios range from specific gear restrictions on the reef to marine protected areas (MPAs), which includes restricting (i) destructive fishing; (ii) fishing for live reef fish; (iii) net fisheries on the reef; (iv) shark fishery on the reef; and finally, imposing three levels of fishing closure: 25% of the model area; 50% of the model area; and 75% of the model area closed. Closures are simulated by reducing the effort 25%, 50% and 75%, respectively, in the Ecosim model[27]. The Raja Ampat Ecosim model is used to simulate each restoration scenario for 20 years. A 'status quo' scenario with no effort restriction is used for comparison. The 'status quo' scenario is the continuation of the base fishing effort for 20 years and does not depict the 'status quo' open access nature of fisheries management in the region wherein fishing effort could increase over the next 20 years. Since this chapter explores restoration scenarios, scenarios with increasing fishing effort are not explored. A choice of restoration

[27] By assuming that 25% reduction in effort corresponds to 25% closure, spatial redistribution of fishing effort that could occur after the MPA is established is ignored; the differences between closing different areas on the map is also ignored.

scenarios combined with the starting state of the ecosystem will lead to different levels of ecosystem restoration.

5.2.2.2 Starting and restored ecosystem states

These are two nodes in the network, which represent the exploited or restored state of the ecosystem. The ecosystem restoration goals are discretized into four levels, which are four different states of the ecosystem (Figure 5.2)—'highly exploited'; 'medium exploited'; 'partially restored'; and 'highly restored'. The node 'starting ecosystem state' is discretized into the first three ecosystem states—'highly exploited', 'medium exploited', and 'partially restored'. It is expected that the coral reef ecosystem will be between any of these three discrete levels when the restoration process begins. The node 'restored ecosystem state' can lie anywhere between all the four discrete ecosystem states. The alternate ecosystem states were obtained by running the Ecosim model forward for 20 years at different multiples of the current levels of fishing effort. When restoration is performed, depending on the starting state of the ecosystem and the restoration strategy adopted, the restored ecosystem state will lie somewhere in between the four ecosystem states. The states are described by the relative proportions of coral (hard vs. soft), reef fish biomass (large vs. small and medium), and pelagic fish (large and medium vs. small). A highly depleted ecosystem is low in hard coral, large reef fish, and large pelagic fish; on the contrary, a restored ecosystem has high levels of hard coral, large reef fish, and large pelagic fish. The probabilities for the node 'restored ecosystem state' are calculated using the biomass results, for coral, reef fish and pelagic fish, from the Ecosim model.

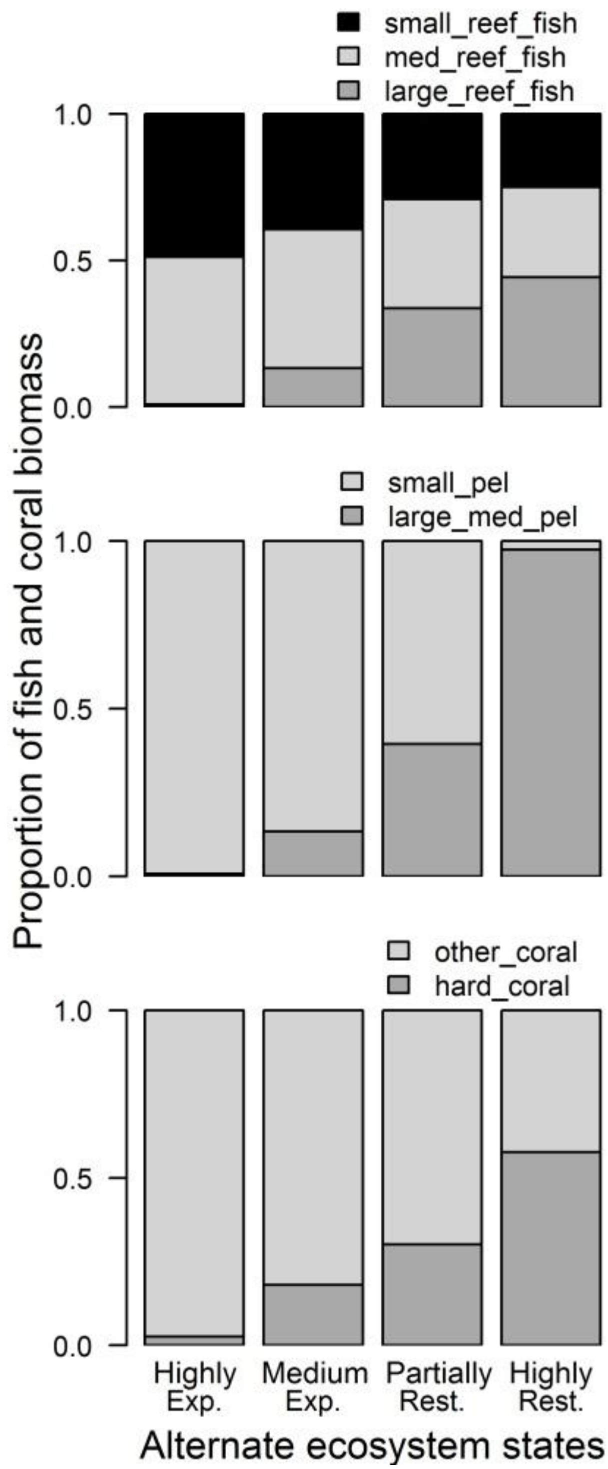

Figure 5.2 Ecosystem states and restoration goals.
The alternate ecosystem states are shown on the horizontal axis. The panels show the composition of reef fish, pelagic fish, and coral in each of the alternate states. The alternate ecosystem states were arrived at by simulating the results of different levels of fishing effort in the Ecosim model.

5.2.2.3 Fisheries catch, average price and total fisheries revenue

The node ‘fisheries catch’ represents the total fisheries catch obtained in 20 years over which different levels of fisheries restrictions were in place; it is calculated by summing the total fisheries catch in each simulation year of Ecosim. Depending on restoration scenario and the starting ecosystem state, the biomass and therefore the catch of different species would vary. To capture the difference in value at different species composition of the catch, the node ‘average price per unit catch’ is created. If the catch is constituted by highly valuable large reef fish then the average price per unit catch would be higher than if the catch is made up of lesser value species. The node ‘total fisheries revenue’ represents the landed value[28] received by the fishers for their catch; this node is dependent on the total fisheries catch and the average price per unit catch.

5.2.2.4 Tourism revenue

The node ‘tourism revenue’ is modeled to depend on the state of the ecosystem; it is expected that a highly restored ecosystem would be highly attractive to tourists while a poor and devastated ecosystem would bring less tourism benefits to the region (Cisneros-Montemayor and Sumaila 2010). In this chapter, it was arbitrarily assumed that the revenue from tourism is directly proportional to the state of the ecosystem: that in partially restored ecosystems, the revenue would be 75% of the revenue in fully restored ecosystems, and 50% and 25% in ‘medium exploited’ and ‘highly exploited’ reef ecosystems, respectively. Several other factors might influence the income from tourism. A review of entrance fees in over 900 marine parks has shown that fees were mainly dependent on the ‘general perception of prices in any country’ and lower fees were charged for parks with good quality reefs because the parks were located in poorer countries (Wielgus et al. 2009). The income was also dependent on coral cover and abundance of large reef fish (Wielgus et al. 2009) (but the analyses did not control for the effect of ‘general perception on price’, so it was not possible to ascertain how the revenue

[28] Here the landed value is used as a proxy for the economic benefits from fisheries. The landed value is chosen because of the limited information available on the costs associated with the different fisheries and limited information on how costs would change in the future years.

would change with change in reef condition). The revenue[29] is constituted by an entrance fee to the park and payments for diving and other activities in the region (Erdmann pers. comm.). A percentage (40%) of the entrance fees is distributed among the villages as a compensation for use of the resource, a practice that has helped to reduce conflict elsewhere (Brunnschweiler 2009). About 34% of the total revenue from the tourism industry goes to the local community (Mark Erdmann pers. comm.). Hence 34% of the total projected revenue is used in the model, especially because this revenue is assumed to be a replacement for the fisher's revenue forgone due to fisheries restrictions. Two projections for increase in revenue from tourism are considered in Raja Ampat.

Tourism Projection Low

In year 2009, around 4,000 tourists (mostly foreign) visited the region. In the first scenario, the number of visitors increases to 10,000 per year after which it stabilizes around this number (Mark Erdmann pers. comm.). Compared to the number of visitors at the Bunaken National Park (~20,000 visitors per year), this is a relatively modest scenario for tourism increase in Raja Ampat. There are, however, a few reasons why tourism increase in Raja Ampat would be modest. Raja Ampat is a remote area—the nearest airport to access Raja Ampat is in the neighbouring Sorong Regency whereas Bunaken in Sulawesi is only an hour away from an international airport (Manado). Accessibility has been stated as a reason for the current lack of interest in some potentially attractive tourist spots in Indonesia (Tourism Indonesia 2010a). It is also considered that the water currents are higher in Raja Ampat (Tourism Indonesia 2008); therefore, only very experienced divers come to Raja Ampat (Mark Erdmann, pers. comm.).

Tourism Projection High

The second scenario is an increase in the number of visitors to Raja Ampat at a constant rate of 25% per year for 20 years. This projection is chosen because of three reasons: (1)

[29] To be consistent with how the revenue from fisheries is modeled and because of lack of information on projected costs, the total revenue is used an indicator of the utility from tourism.

the projected increase for the next year is 1,000, which is about 25% increase from the visitors who arrived in the region in 2009 (~4000); (2) the regency is planning 3 airports in the region, which would increase the potential number of visitors, and there are also plans to set up an office of tourism industry in Bali for Raja Ampat; and (3) at this rate of increase, the maximum number of visitors arriving in Raja Ampat would be about 90,000 visitors per year in 20 years. This level is chosen here as an upper limit because of adverse ecological and social impacts of tourism (Harriott 2002) associated with very high number of visitors to a region. However, tourism could increase in Raja Ampat beyond this level (90,000 visitors per year), and the implications of higher than projected number of tourists is discussed later in the chapter.

5.2.2.5 Conservation interest

Conservation interest is not modeled as a node; it is modeled in combination with tourism, for a highly restored ecosystem would be more attractive to conservationists, similar to tourism. The health of hard coral has been shown to be an indicator to evaluate “human disturbance” (Fisher et al. 2008). Hard coral biomass is used as an indicator of progress in conservation. Conservation interest is modeled at two levels for Raja Ampat, both based on economic evaluation of natural resources in Raja Ampat (Dohar and Anggraeni 2007).

Willingness to Pay (WTP)

The ‘willingness to pay’ (WTP) for an environmental service is usually determined by polling a group of people to judge how much they would spend for a specified goal like ‘improving reef management or committing to increased biodiversity for conservation’ (Peters and Hawkins 2009). Measures of WTP of people in Raja Ampat were available from an economic valuation study conducted by Conservation International. Contingent valuation methods were used to calculate the WTP based on results of a survey conducted with the people of Raja Ampat (Dohar and Anggraeni 2007). The resulting indirect use value was estimated at about 1/30th of the fisheries revenue generated in the region.

Ecosystem Services (ES)

In the same study Dohar and Anggraeni (2007) conducted a valuation of ecosystem services (ES) from the various ecosystems and found that the 'total indirect use' values were about 5 times the direct use values (much higher than the 'perceived value' measured using willingness to pay). The Raja Ampat regency has committed to improving fisheries management, tourism and conservation in its vision statement. The rationale for modeling conservation utility based on ES is that the Regency government, having committed to conservation, might value both the direct (fisheries and tourism) and indirect benefits (ES) from the coral reef ecosystem.

5.2.2.6 Utility

The term 'utility' originally belonged to the field of economics: it measures how a 'particular attribute is valued' (Shotton 1999). This node combines, in essence, the different values and preferences associated with the use and maintenance of the ecosystem. Three competing interests are modeled by the 'utility' node in the influence diagram. Fishers' utility is captured by the fisheries revenue. The utility of the tourism industry is modeled by tourism revenue. Finally, the conservation interests stated in the policy statements of the government and other conservation minded entities are modeled as the utility derived from a restored ecosystem. Three types of utility functions are considered in the chapter (Figure 5.3). Risk neutral utility refers to a linear utility function—the utility is directly proportional to the variable of interest; risk averse utility function increases at a decreasing rate with increase in the value of the variable and converges asymptotically; and the risk prone utility function increases exponentially with increase in the value of the variable. To model the risk averse function, the calculated utility is raised to a power < 1 (0.3 and 0.5) and to model the risk prone function, the utility is raised to a power > 1 (3 and 5) (see Figure 5.3). The following is a description of various reasons for differing utility functions among the different stakeholders:

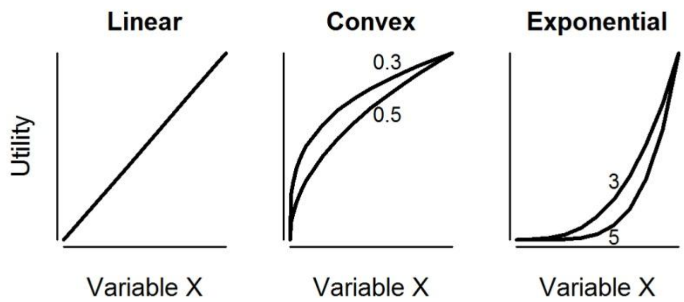

Figure 5.3 General shape of utility functions used in the analysis
Numbers show power value

Utility of fisheries revenue

It is considered that fisheries utility is predominantly linear. This utility function is depicted in the behaviour of fishers wherein they invest in bigger boats or equipment like motors to be able to catch more fish and increase their revenue. Fishers in Raja Ampat have increasingly integrated into the cash economy and have invested in similar measures to obtain higher catch and income from fisheries. This linear relationship can become a risk prone behaviour if the fisher is carrying a huge debt, because this might cause the fisher to take more risk to repay the loan. Also, a small component of fishers involved in destructive fishing might have a risk prone behaviour— their goal would be to maximise their revenue in each fishing trip without being detected or penalized by the management agency. For a section of the fishing community, which continues to adhere to 'adat' (customary law), the utility from fisheries revenue could be risk-averse[30].

[30] Raja Ampat is located in the Papuan province in eastern Indonesia. In the 1960s, the Dutch ceded control over this territory and the administration was taken over by Indonesia. Before becoming part of Indonesia, fishing was governed by 'adat' (customary law), and it was mostly for subsistence—the fisher communities were not integrated into the cash economy (Donnelly et al. 2003; Muljadi 2004). These situations changed when waves of immigrants from other parts of Indonesia arrived in the region under the influence of the government; Indonesians from other parts of the country were more integrated in the cash economy, and they did not recognize the 'adat' (customary law) systems since everything was now supposed to be owned by the state (Goram 2007). In spite of these changes, there are other communities who have resurrected their customary laws and traditional management practices; these laws specify certain timing and gear use in fishing activity. Adherence to such customary laws referred to as sasi adat and sasi gereja (McLeod et al. 2009) indicates risk averse behaviour by the fishing community. The fishers also perceive that the biomass of large reef species and sharks has declined over the years (Ainsworth et al. 2008a). Also there are still fishing communities in the region that are not fully integrated into the cash economy; for such small-scale subsistence oriented fishers, the utility function would be risk prone because these fishers would not

Utility of tourism revenue

The utility of tourism revenue in Raja Ampat, especially from the perspective of the Regency government is predominantly linear. This is reflected in the high interest and rapid development of the tourism industry in the initial stages of the industry from 2001 to 2005 (Mark Erdmann pers. comm.). The interest in developing the industry continues to increase. It is reflected in future plans, which include setting up an office in Bali to improve tourism in Raja Ampat and in the desire of the Raja Ampat Regency government to build airports in Raja Ampat (Tourism Indonesia, 2010b). However, communities associated with the Great Barrier Reef have not supported excessive growth in tourism due to adverse ecological and social impacts (Harriott 2002). High recreational use can damage coral reefs (Goreau 2009). The carrying capacity of coral reef for dive tourism is reported to depend on presence/absence of vulnerable species like coral reefs, training of divers, and presence of other anthropogenic stressors (review by Zakai and Chadwick-Furman (2002), and several studies have reported limits on the number of dives per year at reef sites (Dixon et al. 1993; Schleyer and Tomalin 2000; Hawkins et al. 2002). So, it is possible that after a certain level of growth in tourism, the value from tourism might have a risk averse utility function.

Utility of conservation

The utility of conservation in Raja Ampat is perhaps predominantly linear. The programs for conservation were initiated by COREMAP, The Nature Conservancy, Conservation International, World Wide Fund, almost all began in the early 2000s. At that time destructive fishing was more prevalent in the region than at the present; influenced by the awareness generated by these programs, several fishers have discarded these methods. With the increase in awareness, the NGOs began campaigning for marine protected areas and a network of MPAs was successfully established in 2007 in Raja Ampat. Today efforts continue to improve spatial zoning inside the MPAs. The consistent effort for improving conservation reflects linear utility for conservation. However, a risk averse

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regularly catch more than what is required for consumption in the fishing village—after which any surplus is not useful.

conservation utility[31] can also be argued for on the grounds that the established goals of various management agencies and NGOs for setting up MPAs are in the range of closing 20% to 30% of the marine area (Hoegh-Guldberg 2006; Ban 2008; Olsson et al. 2008) (after these goals are achieved, there might not be interest in closing larger areas). For conservation groups specifically interested in the protection of charismatic species (turtles, whales, etc.) the conservation utility is probably risk prone[32] because every single turtle or whale rescued results in increasingly higher satisfaction.

The different stakeholders are modeled using predominantly linear (risk neutral) functions. In the real world, the utility functions are more complex. This complexity is addressed to a limited extent by modeling each stakeholder with the other predominant utility functions: high risk averse, low risk averse, high risk prone and low risk prone. The goal here is to map the range within which the results would vary under different formulations of utility functions; a better understanding of utility functions would have to be based on surveys of the respective stakeholders. There are a few reasons to adopt this approach in the analysis: (1) A Risk neutral function is easy to model and allows an easy comparison of the different variables in consideration; (2) Risk neutral behavior is midway between the extremes of risk prone and risk averse behaviour (Binmore 1992) and so is a better choice as a base model; and (3) subsequent modeling of risk averse and risk prone behavior show how the difference in utility influences the results and provides a better understanding of the influence utility functions have on the results. Similar assumptions of risk neutral behavior have been made for the sake of simplicity in fisheries analysis (McKelvey et al. 2007).

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[31] Similarly, several biological reference points used as thresholds in fisheries management (Mace 1994; Hilborn 2002) and the reference points trigger responsive measures only when the biological indicators approach these thresholds.

[32] Risk prone conservation utility is also shown by the general public when they may be willing to pay to protect a well-known diverse region, but may not be interested in trying to conserve a less diverse ecosystem because they do not see the value of protecting a system that is very depleted or they believe that the system will not recover.

5.2.3 Discounting

Usually the benefits in the future are not valued the same as benefits in the present (Clark 1973), and this 'time preference' or 'impatience' is captured by discounting (Clark 1990; Sumaila 2004). The higher the discount rate, the higher current benefits are valued relative to the future ones. Different levels of discounting favour different "time streams" of benefits and so have considerable impact on the policy choice (Berman and Sumaila 2006). In the analysis, fisheries revenue, tourism revenue, and conservation benefits, each are discounted at four rates 3%, 7%, 10% and 24% for 20 years. The four levels of discount rate are taken from different sources. The 3% discount rate is used as a proxy for intergeneration discounting: this is based on the findings in Sumaila (2004) that conventional discounting rates between 0 to 3% give similar results as intergenerational discount rates. Bailey (2007) uses 7% discount rate in a principle agent analysis of destructive fishing in Raja Ampat. The discount rate 10% is obtained from economic valuation of natural resources in Raja Ampat (Dohar and Anggraeni 2007). A discount rate of 24% is obtained from Buchary (2010), and it is the official social rate of discounting for Indonesia. The net present value (NPV) from the flow of fisheries revenue, tourism revenue, and conservation benefits in the next 20 years is calculated as follows (Sumaila and Walters 2005):

Discount factor $d = \frac{1}{1+r}$ where, r is the discount rate

Weight on benefits in each year: $W_t = d^t$

Net present value: $NPV = \sum_{t=0}^{T} V_t W_t$

Where, V_t is the revenue in a given year t, and T denotes the total number of years over which the discounted benefits are calculated. Figure 5.4 shows the flow of discounted benefits with change in time.

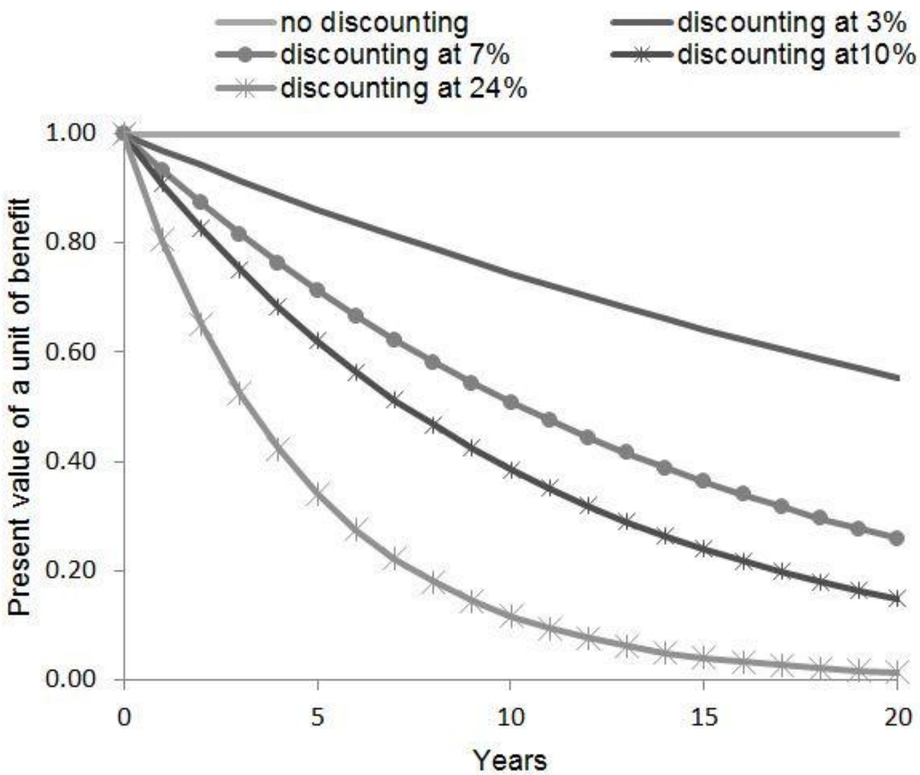

Figure 5.4 Decay of future benefits at different discount rates used in the analysis. Area under the curves corresponds to net present value.

5.3 Results

The results are analyzed to see which scenarios are favored depending on different stakeholders and different specifications (linear or non-linear) of the utility functions considered in the analysis to see if any scenarios emerged as robust under the existing sources of uncertainty. Linear utility functions are explored first followed by non-linear utility functions.

5.3.1 Linear utility functions

If **fisheries revenue** is the only source of utility (Figure 5.5) or the only source of revenue for the regency management, then implementing marine protected areas is not a favored option. In this case, scenarios with minimum restrictions on fisheries are favored. For a highly exploited ecosystem, the utilities of status quo fishing and minimum restrictions on fisheries are very similar. When the ecosystem state is “medium exploited”, ‘restricting net fisheries’ and implementing a ‘25% closure’ are observed to be slightly better than the status quo scenario. When the starting ecosystem state is “partially restored”, ‘restricting net fishing’ and ‘no shark fishing’ appear to be the best options.

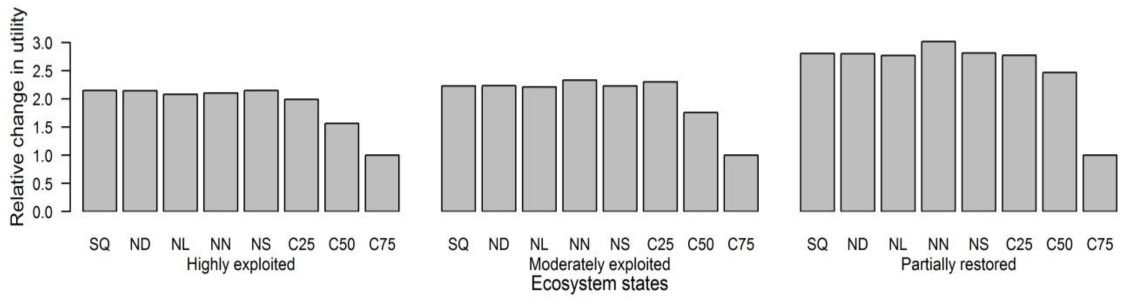

Figure 5.5 Utility of fisheries revenue.
Fisheries revenue is modeled as a linear function. The three panels show the results for the three starting ecosystem states. The vertical axis shows the relative difference in utility (the scenario with the minimum utility in any ecosystem state is assigned a value 1 and the remaining scenarios are shown relative to that scenario). The abbreviations for each bar describe the restoration scenario (SQ Status Quo, ND No destructive fishing, NL No live fish fishing, NN No Net fishing, NS No shark fishing, C25 25% closure, C50 50% closure, C75 75% closure)

When all the weight is placed on **tourism revenue** (Figure 5.6), and the ecosystem is “highly exploited”, then the most favored policies are fishing closures using MPAs, with 75% closure observed to be twice as good as fishing gear restrictions. The favored policies are the same for “moderately exploited” and “partially restored” ecosystems, but with improvement in ecosystem state, the difference in utility between the best and worst options declines. The response is similar whether the revenue from tourism is modeled according to the ‘low’ or ‘high’ scenario.

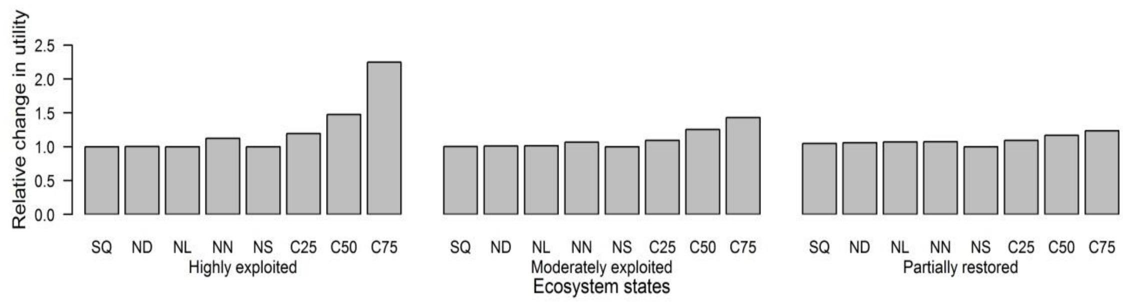

Figure 5.6 Utility of tourism revenue.
Tourism revenue is modeled as a linear function. The three panels show the results for the three starting ecosystem states, and the abbreviations for each bar describe the restoration scenario (SQ Status Quo, ND No destructive fishing, NL No live fish fishing, NN No Net fishing, NS No shark fishing, C25 25% closure, C50 50% closure, C75 75% closure)

Modeling **conservation utility** alone**,** with all non-negative values for utility (Figure 5.7), favors fishing closures across all ecosystem states. For ecosystem state "highly exploited", the protection scenarios are more strongly favored than for ecosystem states "medium exploited" and partially restored". The reason for the observation is that there is a higher increase in conservation benefit from protecting a "highly exploited" ecosystem than protecting a "medium exploited" or partially restored". The results obtained are very similar whether conservation is modeled according to WTP or ES. Modeling **conservation utility** alone, with a negative value for low biomass for hard coral, also favors fishing closures across all ecosystem states. However, 75% closure of a "partially restored" ecosystem would be probably only realistic when considering waters around uninhabited islands where the fishers would not prefer to fish.

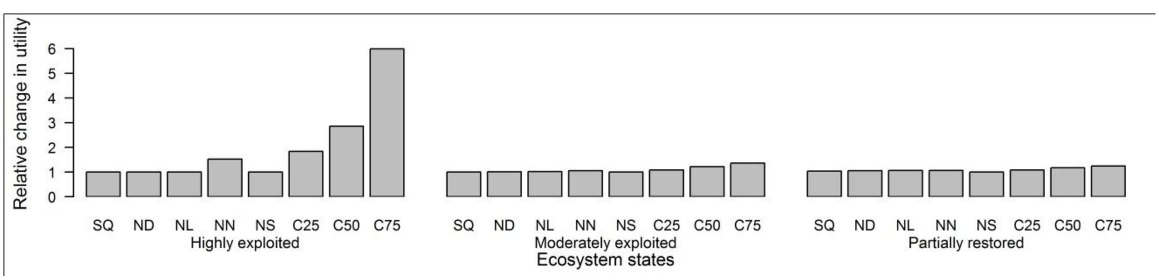

Figure 5.7 Utility from conservation benefits.
Conservation benefits are modeled as a linear function (non-negative) in the top panel. In the lower panel conservation benefits are modeled with linear (negative values for depleted ecosystem) utility function.

When **fisheries and tourism revenue** contribute to utility, the MPAs become more favorable than when only fisheries revenue was considered, and less favorable than when tourism alone contributed to utility. If the tourism scenario 'low' is considered, and ecosystem state is "highly exploited", then 'status quo' and minimal fishing restriction scenarios are favored. For "moderately exploited" and "partially restored" ecosystem states, the most favored scenarios are restricting net fisheries and MPA scenario with 25% closure. In the model, it was assumed that in partially restored ecosystem, the tourism revenue would be 75% of the tourism revenue from a fully restored ecosystem, and 50% and 25% in 'medium exploited' and highly exploited' reef ecosystems respectively, and the results are sensitive to this assumption. For "highly exploited" and "medium exploited" ecosystems, the tourism revenues (low) are lower compared to fisheries revenue. So, for "highly exploited" and "medium exploited" ecosystems, the scenarios with highest utility are similar to the scenarios that had the highest utility when utility from fisheries alone were considered. A change in the preferred scenario is seen only when the ecosystem state is "partially restored" because the tourism revenue from a "partially restored" ecosystem is higher.

If tourism is expected to follow scenario 'high', then across all ecosystem states, 'restricting net fishing' emerges as the most favorable scenario. For "highly exploited" ecosystem state, 'restricting net fishing' is only slightly better than scenarios with minimum fishing restrictions. For "moderately exploited" and "partially restored" ecosystem states, the second best scenario is '25% closure'.

When **fisheries and tourism revenue and conservation benefits** contribute to utility, 24 alternative combinations of results can arise; these combinations derive from ecosystem state, tourism modeled according to low or high scenario, and conservation modeled according to WTP or ES. Because of the large number of combinations, the results showing the two best scenarios for each utility combination are shown in Table 5.1. When conservation is modeled based on WTP, the results are very similar to the results obtained when only fisheries and tourism were considered. The reason is that conservation modeled as WTP is only about 1/30th of the revenue from fisheries and so has only a small influence on the results. When conservation is modeled according to ES, then the conservation benefits override all other sources of utility—fisheries revenue, low or high tourism revenue—and consistently favor 75% fishing closure[33].

Table 5.1 Utility from fisheries revenue, tourism revenue, and conservation benefits when all the three sources of utility are modeled with linear utility functions
The abbreviations describe the restoration scenario (SQ Status Quo, ND No destructive fishing, NL No live fish fishing, NN No Net fishing, NS No shark fishing, C25 25% closure, C50 50% closure, C75 75% closure)

Ecosystem states	Tourism	Conservation			
		WTP		ES	
		Linear - non negative	Linear negative	Linear - non negative	Linear negative
Highly exploited	Low	SQ NS	SQ NS	C75 C50	C75 C50
Medium exploited		NN C25	C25 NN	C50 C25	C75 C50
Partially restored		NN C25	C25 NN	C75 C50	C75 C50
Highly exploited	High	NN SQ	NN SQ	C75 C50	C75 C50
Medium exploited		NN C25	C25 NN	C50 C25	C75 C50
Partially restored		NN C25	NN C25	C75 C50	C75 C50

5.3.2 Discounting

When only the fisheries revenue contributes to utility, the NPV from status quo and minimum fishing restrictions scenarios are greater than the NPV from protection scenarios because the revenue from protection scenarios increases later in the time period (from rebuilding fish populations). Protection scenarios are less favoured at high discount rates. When fisheries revenue and tourism contribute to utility, the NPV of both fisheries and tourism revenues decline in comparison to undiscounted revenues, but the tourism revenues are more affected than fisheries revenues (Figure 5.8). The result is observed because tourism in Raja Ampat is a developing industry, and a greater portion of the

[33] There is a big difference between modeling conservation as WTP or ES. WTP measures what amount an individual is willing to pay and therefore will usually be a value lower than his/her income. When income comes from direct benefits from use of a resource, this value tends to be considerably lower than ES which combines both direct and indirect benefits.

revenue is expected in the later years after the industry becomes fully established. The contribution from tourism revenue to the decisions diminishes; notice, as the discount rate increases, the tourism NPV becomes a smaller fraction of the total NPV. Because the relative contribution of tourism to fisheries revenue declines, the scenarios preferred are similar to scenarios preferred when fisheries revenue is modeled as the only source of utility. Therefore, when discount rates are considered, fishing restrictions (i.e., restoration strategies) are not favoured; this influence on decision is more predominant at higher rates of discounting. When conservation benefits are also added to the calculations of utility, the results obtained are similar for conservation modeled according to the WTP because the values of WTP for conservation benefits are very small (1/30th) in comparison to fisheries revenues. Conservation modeled as ES consistently favours protection scenarios even at the highest discounting rates.

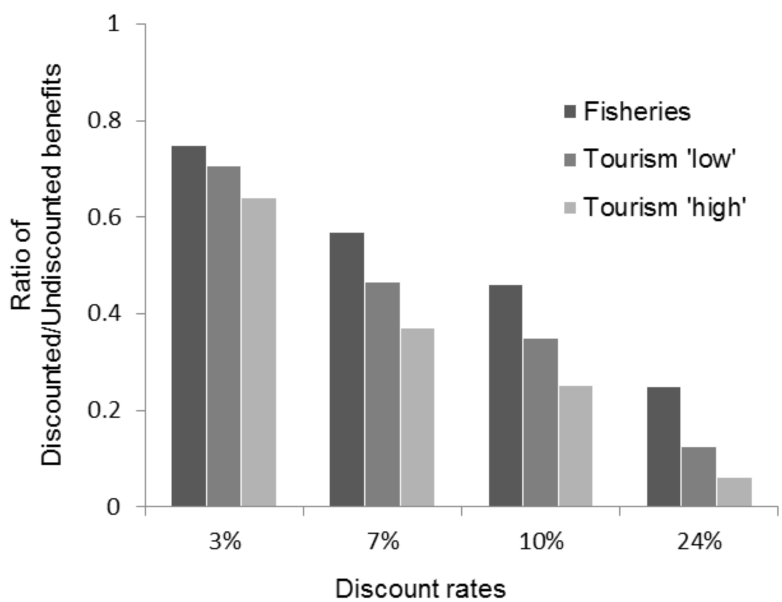

Figure 5.8 Comparison of discounted benefits from fisheries and tourism.
The horizontal axis shows different discount rates. The vertical axis shows the ratio of discounted benefits to undiscounted benefits.

5.3.3 Non-linear utility functions

In the analysis, it is assumed that the utility of all the stakeholders is predominantly linear and that some stakeholders within each group could have non-linear utility functions. Here, the implications of non-linear utility functions are briefly explored.

Fisheries risk-averse utility function

If the utility of fisheries and conservation are modeled as risk averse functions, then it is difficult to choose between different restoration scenarios because all the scenarios have very similar values of utility. Whether the policy favors scenarios with slightly less or more fishing mortality depends on the power on the utility functions—if the power on the fisheries revenue is lower (0.3) than the power on the conservation utility (0.5), then the utility values slightly favor conservation and vice versa.

If the utility of fisheries are modeled as risk averse function and tourism-conservation combination as risk prone function, then all scenarios that allow high levels of conservation (setting up MPAs and restricting net fisheries) are very highly favored compared to status quo or minor fishing restrictions. For a 'highly exploited' ecosystem, depending on whether conservation is modeled based on WTP or ES, the factor by which 75% closure is better than the worst choice maybe 11 to 178 times (power on conservation utility equals 3) or 60 to 14000 times (power on conservation utility equals 5). For a 'medium exploited' ecosystem, the utility for the MPAs is about 2 to 6 times the scenario that results in least utility, depending on the model used to value conservation benefits (WTS or ES) and power on conservation utility function (3 or 5). Similarly MPA scenarios have the highest utility when the ecosystem state is "partially restored". The power on the utility function affects the degree to which one decision is favored over the other (e.g., 3 times vs. 8 times), but it does not change the decision.

Fisheries risk prone utility function

If fisheries utility is modeled as a risk prone function and tourism-conservation combination using risk averse function, then scenarios with minimal restrictions are preferred across all ecosystem states.

The results are interesting when both fisheries and tourism-conservation combination are modeled as risk prone utility functions. The results are illustrated in Table 5.2.

Table 5.2 Utility from fisheries revenue, tourism revenue, and conservation benefits. All are modeled with risk prone utility function. The table shows the scenarios which result in highest utility values. The abbreviations describe the restoration scenario (SQ Status Quo, ND No destructive fishing, NL No live fish fishing, NN No Net fishing, NS No shark fishing, C25 25% closure, C50 50% closure, C75 75% closure)

		Exponent on Conservation (WTP or ES) and Tourism (Low or High)							
		3		5		3		5	
Ecosystem states	**Exponent on fisheries utility**	**WTP & Low**	**WTP & High**	**WTP & Low**	**WTP & High**	**ES and Low**	**ES and High**	**ES and Low**	**ES and High**
Highly exploited	3	SQ NS	C75 NN	C75 C50	C75 C50	C75 C50	C75 C50	C75 C50	C75 C50
Medium exploited		NN C25	C75 C50	C75 C50	C75 C50	C75 C50	C75 C50	C75 C50	C75 C50
Partially restored		NN C50	C75 C50	C75 C50	C75 C50	C75 C50	C75 C50	C75 C50	C75 C50
Highly exploited	5	SQ NS	SQ NS	SQ NS	C75 SQ	SQ NS	SQ NS	C75 C50	C75 C50
Medium exploited		NN C25	NN C25	NN C25	C75 C50	NN C25	NN C25	C75 C50	C75 C50
Partially restored		NN NS	NN NS	C75 C50	C75 C50	NN NS	NN NS	C75 C50	C75 C50

When conservation is modeled using ES (for tourism-conservation combination), then irrespective of the exponent on the utility functions, the results always favor setting up the largest MPA, except when the exponent on fisheries utility (5) is greater than the exponent on tourism-conservation utility (3). When conservation is modeled using WTP (for tourism-conservation combination), but the exponent on conservation utility is higher than the exponent on fisheries utility, then again the highest levels of protection of the ecosystem are favored. When conservation is modeled using WTP (for tourism-conservation combination), but the exponent on conservation utility is smaller than the exponent on fisheries utility, then scenarios that allow for minimum fishing restrictions are favored for a “highly exploited” ecosystem. For a “medium exploited system”, “no net fishing” and “25% closure” are favored, and for a partially restored ecosystem, “no net fishing” and “no shark fishing” are favored. When the exponent on both the fisheries and conservation utility is the same (3 or 5), then the results are dependent on the projections for tourism industry. When the projection for tourism is ‘high’, then protection scenarios are favored. When tourism projection is ‘low’ and the ecosystem is “highly exploited”, the results favor minimal fishing restrictions, and in “medium exploited” ecosystem the results favour protection measures such as ‘no net fishing’ and ‘25% closures’. In “partially restored” ecosystem the scenarios ‘no net fishing’ and ‘50% closure’ and ‘75% closure’ are favoured.

Thus, non-linear utility functions can considerably influence the results obtained from the analysis.

5.4 Discussion

5.4.1 Policy choice

When only one type of stakeholder—the fisher—is considered the scenarios that maximize fishing revenues are favored. When other stakeholders, tourism industry and conservationists are included, protection scenarios become increasingly favorable. Other studies have indicated that high tourism revenues have the potential to improve coral conservation (White et al. 2000; Depondt and Green 2006) and that alternative source of incomes can reduce the opposition to reserves (Smith et al. 2010). When tourism is modeled according to scenario ‘low’, then for a highly exploited ecosystem, the ‘status quo’ is scenario is preferred. The same result is observed even when conservation (WTP) is added to the utility. Similar results are observed when fisheries are modeled with risk prone utility function. The reason for this difference in choice based on the state of the ecosystem is that the tourism benefits for a highly exploited ecosystem even when restoration efforts are made are not high enough to replace the revenue lost from fisheries. Higher levels of tourism revenue (more than scenario 'low') or conservation benefits (more than WTP) are able to offset losses in fisheries and make MPAs a favorable decision also for highly depleted ecosystems. Thus, when industries that depend on a conserved ecosystem are brought into the decision making process, then policies that prevent ecosystem degradation and favor protection can be seen as preferable. From the perspective of a decision maker, it is easier in this case to arrive at a compromise between different stakeholders because losses in one source of revenue are compensated by gains in the other sources.

The results suggest that the most robust policies are restricting net fisheries in Raja Ampat followed by 25% closure. Here by using Ecosim, the habitat choices associated with MPAs is ignored; a much more rigorous analysis would involve spatial models of ecosystems in different states of exploitation. These policies are favored in most situations modeled using linear utility for a medium exploited, partially restored, and highly exploited ecosystem when the tourism is modeled according to scenario 'high'. Even when the utilities are modeled using risk averse and risk prone functions, no net fishing and 25% closure are the favored policies for 'medium exploited' ecosystem. No net fishing is also a favored policy in several instances for a partially restored ecosystem. It is an interesting result that the robust scenario for MPA is similar to 20 to 30% closure based on other research (Gaines et al. 2010). These policies compromise the utilities of the fisheries revenue, but the scenarios emerge as robust because the gains in tourism revenue and conservation benefits compensate the losses in fisheries revenue.

5.4.2 Discounting

When discounting is included in the calculations of the net benefits, then restoration scenarios become less favourable. The 'twofold' problems with discounting net benefits of marine restoration identified in Berman and Sumaila (2006) are highlighted in this chapter. The first is valuing only the 'production' or direct benefits from the system, and the second is the lower value attributed to future benefits from a restored ecosystem.

When indirect benefits are modeled by including conservation benefits (ES) in the utility, then protection scenarios are favoured. Global survey of ecosystem services has shown that use values are only a small percentage of the total value from an ecosystem (Costanza et al. 1997; Jansson et al. 1999; Boumans et al. 2002). Not accounting for non-use values from an ecosystem ignores the "cost resource depletion imposes on future generations" (Howarth and Farber 2002). Though quantification of ecosystem services is not straightforward (Beaumont et al. 2007; Meyerson et al. 2008), the results in this chapter show that including non-use values would significantly influence the cost-benefit calculations of marine management policy in favour of restoration. When only direct benefits are considered, scenarios that provide maximum benefits from fisheries in the short-term are preferred. The preferences for short-term benefits are greater at higher discount rates. Four discount rates are used in the analysis, but higher discount rates may be more appropriate in an evaluation of strategies for coral reef ecosystems in developing countries considering that poverty among fishers is associated with high discount rates (Pauly et al. 1989; Sumaila 2003). If, in addition to the time preferences of the current generation, the perspective of future generations are included in the evaluation (~3% discount rate), then long-term benefits become more favourable. Stern and Taylor (2007) advocate low discount rates in valuation of ecosystems because unlike 'roads or railways', ecosystems would be valued 'as long as the planet and its people exist'. Economic evaluations using high discount rates on direct benefits will by their nature not support restoration strategies; including industries that depend on sustainable use of the ecosystem in the calculations of benefits favour restoration strategies.

5.4.3 Utility functions

It is observed that when the utility is raised to powers less than 1 (risk averse function), then only very big changes in the variable are captured by the utility function. Modeling the utility with power > 1 leads to the opposite result; small changes in the variable are magnified. For example, when revenue from fisheries is modeled as a linear function, then the highest utility is twice the value of the lowest utility; when the power on the utility function is 0.3, the highest utility value is only 26% better than the lowest utility value; when the power on the utility function is 5, the highest utility value is 27 times the lowest utility value. So risk averse utility functions should be used when it is expected that the variable in question will show a high response to change in the model, but the change needs to be dampened in the utility function (for example, in a model, the plankton biomass might change by higher orders of magnitude than marine mammals, and the modeler might wish to reduce the influence of plankton biomass). Risk prone utility functions should be used when small changes need to be magnified (for example, in a climate model, small changes in temperature could have a high significance compared to other changes in the model). A linear utility function should always be modeled for 2 reasons: (1) it can be used to observe the trend in utility from different sources; and (2) it can be used to error check the model.

5.4.4 Tourism revenue

In this analysis, tourism and fisheries revenue have been viewed as interchangeable sources of revenue to the community. However, one of the criticisms against the tourism industry is that local people are often sidelined, and only a small portion of the revenues "trickle down to the local population" (Dixon et al. 1993). In Raja Ampat, the local community receives about 34% of the revenue; it seems that the tourism development is progressing in an equitable direction. About 4 years ago, the local people in Raja Ampat were not familiar with visits of tourists, the villagers did not speak English, and the general perception was that because of these difficulties, they would not be able to adapt to tourism as an alternate source of revenue. It seems to have changed, with several diver resorts reporting that tourists will find people conversant in English, Spanish, and other foreign languages; it might change further in future years when the younger, more educated individuals of Raja Ampat enter the industry. The process for marine management including fishers and tourism adopted in Raja Ampat is the same as the model developed in Bunaken National Park through a process that evolved after a long process of consultation and involvement of the different stakeholders. The proportion of tourism revenue received by the local population might then increase to above 34%; however, this possibility is not considered in the chapter. Tourism growth is modelled for two extreme 'low' and 'high' scenarios based on the best information available. The variation is shown by the results obtained under the two extreme scenarios. Additional work is needed to build more realistic models for growth of tourism in Raja Ampat because tourism will depend on several factors ranging from perception of health and safety in the region, ease of access, visa and other regulations, local support, and general political stability, etc.

5.4.5 Conservation utility

If conservation benefit is modeled based on ecosystem services, then this source of utility completely over-rides the other sources of utility showing that the value of the indirect benefits from the ecosystem are higher than direct extractive benefits. However, it is unrealistic to forego the direct benefits for the indirect benefits. The utility from the fishing industry is not only the revenue from fish catch, but it also a source of food (Brunner et al. 2008). Our dependence on food, as one of the most basic needs of life, cannot be captured by a utility function based on fishing revenues, but would probably need a different metric (for example, caloric needs).

5.4.6 Other anthropogenic impacts

The Ecopath with Ecosim model is able to capture the changes in biomasses of fish population based on changes in fishing effort. However, it is not able to capture other changes because of development of other industries. Interest in the mining industry could result in siltation and poisoning due to tailings which could lead to habitat changes and detrimental effects on fish populations. These sources of variation are not considered in the model.

5.5 Conclusion

The analysis is able to successfully identify two policy options for restoration that are quite robust against the uncertainty surrounding complex utility functions of the various stakeholders. The scenarios are restricting net fisheries and 25% closure. Bringing multiple stakeholders into the decision making process is not easy—In Bunaken National Park, it took several years of effort and cooperation, and entailed several failures before a management method was identified that was considered equitable by all stakeholders, the fishers, tourism industry, local management body, and tourists. Building a model of the utilities of the different groups can speed up the process by providing scenarios that are at least theoretically robust to expected uncertainty and differences in utility functions between groups. Socially successful restoration strategies influence long-term ecological success (Christie 2004); therefore model-based studies like this, which can explore equitable resource allocation and use, can provide useful directions for testing actual responses in the field.

5.6 References

Ainsworth C. H., T. J. Pitcher, and C. Rotinsulu. 2008a. Evidence of fishery depletions and shifting cognitive baselines in Eastern Indonesia. Biological Conservation **141**(3):848-859.

Ainsworth C. H., D. A. Varkey, and T. J. Pitcher. 2008b. Ecosystem simulations supporting ecosystem-based fisheries management in the Coral Triangle, Indonesia. Ecological Modelling **214**(2-4):361-374.

Ainsworth C. H., D. A. Varkey, and T. J. Pitcher. 2008c. Ecosystem simulation models of Raja Ampat, Indonesia, in support of ecosystem based fisheries management. In: Bailey, M. and T. J. Pitcher, editors. Ecological and economic analyses of marine ecosystems in the Birds Head Seascape, Papua, Indonesia: II. Fisheries Centre Research Reports **16**(1):3-122.

Ainsworth C. H., D. A. Varkey, and T. J. Pitcher. 2007. Ecosystem simulation models for the Bird's Head Seascape, Papua. In: Pitcher T.J., C.H. Ainsworth, and M. Bailey, editors, Ecological and economic analyses of marine ecosystems in the Birds Head Seascape, Papua, Indonesia. Fisheries Centre Research Reports **15**(5):6–172.

Bailey M. 2007. Economic analysis of unregulated and illegal fishing in Raja Ampat, Indonesia. Master thesis submitted to University of British Columbia, Canada.

Ban N. 2008. Multiple perspectives for envisioning marine protected areas. PhD Thesis, University of British Columbia, Canada.

Beaumont N. J., M. C. Austen, J. P. Atkins, D. Burdon, S. Degraer, T. P. Dentinho, S. Derous, P. Holm, T. Horton, E. van Ierland, A. H. Marboe, D. J. Starkey, M. Townsend, and T. Zarzycki. 2007. Identification, definition and quantification of goods and services provided by marine biodiversity: Implications for the ecosystem approach. Marine Pollution Bulletin **54**(3):253-265.

Berman M., U. R. Sumaila. 2006. Discounting, amenity values, and marine ecosystem restoration. Marine Resource Economics **21**(2):211.

Binmore, K.G. 1992. Fun and games. A text on game theory. Published by D.C. Heath and Company. 602p.

Boumans R., R. Costanza, J. Farley, M. A. Wilson, R. Portela, J. Rotmans, F. Villa, and M. Grasso. 2002. Modeling the dynamics of the integrated earth system and the value of global ecosystem services using the GUMBO model. Ecological Economics **41**(3):529-560.

Brown K., W. N. Adger, E. Tompkins, P. Bacon, D. Shim, and K. Young. 2001. Trade-off analysis for marine protected area management. Ecological Economics **37**(3):417-434.

Brunner E. J., P. J. S. Jones, S. Friel, and M. Bartley. 2008. Fish, human health and marine ecosystem health: policies in collision. International Journal of Epidemiology **38**(1):93-100.

Brunnschweiler J. M. 2010. The Shark Reef Marine Reserve: a marine tourism project in Fiji involving local communities. Journal of Sustainable Tourism **18**(1):29-42.

Buchary E. A. 2010. In search of viable policy options for responsible use of sardine resources in the Bali Strait, Indonesia. PhD Thesis, University of British Columbia, Canada.

Charles A. T. 2002. The precautionary approach and'burden of proof'challenges in fishery management. Bulletin of Marine Science **70**(2):683-694.

Christensen V. 1992. ECOPATH II--A software for balancing steady-state ecosystem models and calculating network characteristics. Ecological Modelling **61**(3):169-185.

Christie, P. 2004. Marine protected areas as biological successes and social failures in Southeast Asia. American Fisheries Society Symposium **42**:155-164.

Cisneros-Montemayor A., R. S. Sumaila. 2010. A global estimate of benefits from ecosystem-based marine recreation: Potential and implications for management. Journal of Bioeconomics **12**(3):245-268.

Clark C. W. 1973. The economics of overexploitation. Science **181**(4100): 630-634.

Clark C. W. 1990. Mathematical Bioeconomics: The optimal management of renewable resources. Wiley, New York. 368p.

Conservation International. 2008. Environmental Friendly Development in Raja Ampat. Available:
<http://www.conservation.or.id/home.php?catid=30&tcatid=178&page=g_peluang.detail> [Accessed on July 30, 2008].

Costanza R., R. d'Arge, D. R. Groot, S. Farber, M. Grasso, B. Hannon, K. Limburg, S. Naeem, V. R. O'Neill, J. Paruelo, G. R. Raskin, P. Sutton, and M. v. d. Belt. 1997. The value of the world's ecosystem services and natural capital. Nature **387**:253-260.

Depondt F., E. Green. 2006. Diving user fees and the financial sustainability of marine protected areas: Opportunities and impediments. Ocean and Coastal Management **49**(3-4):188-202.

Dixon J. A., L. F. Scura, and T. V. Hof. 1993. Meeting ecological and economic goals: marine parks in the Caribbean. Ambio: A Journal of the Human Environment **22**(2-3):117-125.

Dohar A. G., D. Anggraeni. 2007. Valuasi Ekonomi Sumberdaya Alam di Kepulauan Raja Ampat. Conservation International, Bekerjasama dengan, Universitas Negeri Papua. (in Indonesian) Economic valuation report for Raja Ampat, Indonesia; Conservation International Indonesia.Available: Rotinsulu C. Gunung Arfak CI. Jl. 45. Sorong, Papua, Indonesia. Available: chris@conservation.or.id.

Donnelly R., D. Neville, and P. J. Mous. 2003. Report on a rapid ecological assessment of the Raja Ampat Islands, Papua, Eastern Indonesia. The Nature Conservancy Coral Triangle Center. Indonesia. Final Draft. 250p.

Dulvy N. K., Y. Sadovy, and J. D. Reynolds. 2003. Extinction vulnerability in marine populations. Fish and Fisheries **4**(1):25-64.

Erdmann M. V., P. R. Merrill, M. Mongdong, I. Arsyad, Z. Harahap, R. Pangalila, R. Elverawati, and P. Baworo. 2004. Building effective co-management systems for decentralized protected areas management in Indonesia: Bunaken National Park case study. Natural Resources Management Program. NRM III Program Secretariat. Jakarta. 149p.

Fisher W. S., L. S. Fore, A. Hutchins, R. L. Quarles, J. G. Campbell, C. LoBue, and W. S. Davis. 2008. Evaluation of stony coral indicators for coral reef management. Marine Pollution Bulletin **56**(10):1737-1745.

Fox H. E., J. S. Pet, R. Dahuri, and R. L. Caldwell. 2003. Recovery in rubble fields: long-term impacts of blast fishing. Marine Pollution Bulletin **46**(8):1024-1031.

Gaines S. D., C. White, M. H. Carr, and S. R. Palumbi. 2010. Designing marine reserve networks for both conservation and fisheries management. Proceedings of the National Academy of Sciences doi/10.1073/pnas.0906473107.

Goram Y. 2007. Interview (Sawen, Sawaki, J., Raubaba, S. and L. Metansan, 10-11 February, 2007).

Goreau T. J. 2009. Tourism and sustainable coral reefs. Global coral reef alliance white paper. Available: <www.globalcoral.org/Ecotourism%20Biorock%20complete.pdf>.[Accessed on May 14, 2010].

Haapasaari P., C. G. J. Michielsens, T. P. Karjalainen, K. Reinikainen, and S. Kuikka. 2007. Management measures and fishers' commitment to sustainable exploitation: a case study of Atlantic salmon fisheries in the Baltic Sea. ICES Journal of Marine Science **64**(4):825-833.

Harriott V. J. 2002. Marine tourism impacts and their management on the Great Barrier Reef. CRC Reef Research Centre Technical Report **46**:1-41.

Hawkins J. P., C. M. Roberts, T. V. T. Hof, K. De Meyer, J. Tratalos, and C. Aldam. 2002. Effects of recreational scuba diving on Caribbean coral and fish communities. Conservation Biology **13**(4):888-897.

Hilborn R. 2002. The dark side of reference points. Bulletin of Marine Science **70**(2):403-408.

Hoegh-Guldberg O. 2006. Ecology: complexities of coral reef recovery. Science **311**(5757):42-43.

Howard R. A., J. E. Matheson. 2005. Influence diagrams. Decision Analysis 2(3):127-143.

Howarth R. B., S. Farber. 2002. Accounting for the value of ecosystem services. Ecological Economics **41**(3):421-429.

Hughes T. 1994. Catastrophes, phase-shifts, and large-scale degradation of a Caribbean coral reef. Science **265**:1547-1551.

Hutchings J. A., J. D. Reynolds. 2004. Marine fish population collapses: consequences for recovery and extinction risk. Bioscience **54**(4):297-309.

Jackson J. B. C., M. X. Kirby, W. H. Berger, K. A. Bjorndal, L. W. Botsford, B. J. Bourque, R. H. Bradbury, R. Cooke, Jon Erlandson, J. A. Estes, T. P. Hughes, S. Kidwell, C. B. Lange, H. S. Lenihan, J. M. Pandolfi, C. H. Peterson, R. S. Steneck, M. J. Tegner, and R. R. Warner. 2001. Historical overfishing and the recent collapse of coastal ecosystems. Science **293**(5530):629-638.

Jansson Å., C. Folke, J. Rockström, L. Gordon, and M. Falkenmark. 1999. Linking freshwater flows and ecosystem services appropriated by people: the case of the Baltic Sea drainage basin. Ecosystems **2**(4):351-366.

Jensen, F.V. 1997. Bayesian networks and influence diagrams. Paper presented at First European Conference for information technology in agriculture, Copenhagen, Denmark, 15-18 June, 1997. Available: http://www.dina.dk/efita-conf/program/paperspdf/ix_2.pdf

Kaplan I. M., B. J. McCay. 2004. Cooperative research, co-management and the social dimension of fisheries science and management. Marine Policy **28**(3):257-258.

Kuikka S., M. Hilden, H. Gislason, S. Hansson, H. Sparholt, and O. Varis. 1999. Modeling environmentally driven uncertainties in Baltic cod (*Gadus morhua*) management by Bayesian influence diagrams. Canadian Journal of Fisheries and Aquatic Sciences **56**(4):629-641.

Lane D. E., R. L. Stephenson. 1998. A framework for risk analysis in fisheries decision-making. ICES Journal of Marine Science **55**(1):1.

Leslie H. M., K. L. McLeod. 2007. Confronting the challenges of implementing marine ecosystem-based management. Frontiers in Ecology and the Environment **5**(10):540-548.

Lotze H. K., H. S. Lenihan, B. J. Bourque, R. H. Bradbury, R. G. Cooke, M. C. Kay, S. M. Kidwell, M. X. Kirby, C. H. Peterson, and J. B. C. Jackson. 2006. Depletion, degradation, and recovery potential of estuaries and coastal seas. Science **312**(5781):1806.

Mace P. M. 1994. Relationships between common biological reference points used as thresholds and targets of fisheries management strategies. Canadian Journal of Fisheries and Aquatic Sciences **51**(1):110-122.

Mantyniemi S., S. Kuikka, M. Rahikainen, L. T. Kell, and V. Kaitala. 2009. The value of information in fisheries management: North Sea herring as an example. ICES Journal of Marine Science **66**(10):2278.

McKelvey, R., P. V. Golubtsov, G. Cripe and K.A. Miller, 2007. The Incomplete Information Stochastic Split-Stream Model: An Overview. Pages 159-183 In: T. Bjørndal, D. V. Gordon, R. Arnason and U. R. Sumaila (eds.) Advances in Fisheries Economics: Festschrift in Honour of Professor Gordon R. Munro. Blackwell: Oxford, U.K.

McKenna S. A., G. R. Allen, and S. Suryadi. 2002. A marine rapid assessment of the Raja Ampat Islands, Papua Province, Indonesia. Conservation International, Washington, DC. RAP Bulletin of Biological Assessment 22. 193p.

McLeod E., B. Szuster, and R. Salm. 2009. Sasi and marine conservation in Raja Ampat, Indonesia. Coastal Management **37**(6):656-676.

Meyerson L. A., J. Baron, J. M. Melillo, R. J. Naiman, R. I. O'Malley, G. Orians, M. A. Palmer, A. S. P. Pfaff, S. W. Running, and O. E. Sala. 2008. Aggregate measures of ecosystem services: can we take the pulse of nature? Frontiers in Ecology and the Environment **3**:56-59.

Morris A. V., C. M. Roberts, and J. P. Hawkins. 2000. The threatened status of groupers (Epinephelinae). Biodiversity and Conservation **9**(7):919-942.

Muljadi A. 2004. Coastal Rural Appraisal report in Kofiau and Misool area of Raja Ampat Islands. Sorong, Indonesia: The Nature Conservancy, South East Asia Centre for Marine Protected Areas. 12p.

Myers R. A., J. A. Hutchings, and N. J. Barrowman. 1996. Hypotheses for the decline of cod in the North Atlantic. Marine Ecology Progress Series **138**(1):293-308.

Olsson P., C. Folke, and T. P. Hughes. 2008. Navigating the transition to ecosystem-based management of the Great Barrier Reef, Australia. Proceedings of the National Academy of Sciences **105**(28):9489-9494.

Pandolfi J. M., R. H. Bradbury, E. Sala, T. P. Hughes, K. A. Bjorndal, R. G. Cooke, D. McArdle, L. McClenachan, M. J. H. Newman, G. Paredes, R. R. Warner, and J. B. C. Jackson. 2003. Global trajectories of the long-term decline of coral reef ecosystems. Science **301**:955-958.

Pauly D., G. Silvestre, and I. Smith. 1989. On development, fisheries and dynamite: a brief review of tropical fisheries management. Natural Resource Modeling **3**(3):307-329.

Pauly D., V. Christensen, J. Dalsgaard, R. Froese, and F. Torres Jr. 1998. Fishing Down Marine Food Webs. Science **279**(5352):860-863.

Peters H., J. P. Hawkins. 2009. Access to marine parks: A comparative study in willingness to pay. Ocean and Coastal Management **52**(3-4):219-228.

Pitcher T. J. 2001. Fisheries managed to rebuild ecosystems? Reconstructing the past to salvage the future. Ecological Applications **11**(2):601-617.

Rabu P. 2006. Tabloid Raja Ampat. Conservation International, Indonesia. **6**: 1-4.

Rose G. A., D. W. Kulka. 1999. Hyperaggregation of fish and fisheries: how catch-per-unit-effort increased as the northern cod (*Gadus morhua*) declined. Canadian Journal of Fisheries and Aquatic Sciences **56**(S1):118-127.

Russ G. R., A. C. Alcala. 2003. Marine reserves: rates and patterns of recovery and decline of predatory fish, 1983-2000. Ecological Applications **13**(6):1553-1565.

Schleyer M. H., B. J. Tomalin. 2000. Damage on South African coral reefs and an assessment of their sustainable diving capacity using a fisheries approach. Bulletin of Marine Science **67**(3):1025-1042.

Shotton R. 1999. Decision theory and its role in fisheries management. Fisheries training programme. UNU, Reykjavik (Iceland). Published by FAO, Rome. Available: <ftp://ftp.fao.org/docrep/fao/article/244908.pdf> [Accessed on May 20, 2010].

Smith M. D., J. Lynham, J. N. Sanchirico, and J. A. Wilson. 2010. Political economy of marine reserves: Understanding the role of opportunity costs. Proceedings of the National Academy of Sciences doi/10.1073/pnas.0907365107.

Stern N., C. Taylor. 2007. Climate change: risk, ethics and the Stern Review. Science **317**(5835):203-204.

Sumaila U. R. 2004. Intergenerational cost–benefit analysis and marine ecosystem restoration. Fish and Fisheries **5**(4):329-343.

Sumaila U. R. 2003. An economic indicator for monitoring fishing pressure on marine ecosystems. Sustainability Indicators EcoFish Workshop, June 2003, Dakar, Senegal. Available: <http://filaman.uni-kiel.de/ecofish/reports/rashid.pdf> [Accessed on June 3, 2010]

Sumaila U. R., C. Walters. 2005. Intergenerational discounting: a new intuitive approach. Ecological Economics **52**(2):135-142.

Suman D., M. Shivlani, and J. W. Milon. 1999. Perceptions and attitudes regarding marine reserves: a comparison of stakeholder groups in the Florida Keys National Marine Sanctuary. Ocean and Coastal Management **42**(12):1019-1040.

Tourism Indonesia. 2010a. Indonesia's toursim: a national tragedy. Available: <http://www.tourismindonesia.com/2010/01/indonesias-tourism-national-tragedy.html> [Accessed on April 12, 2010].

Tourism Indonesia. 2010b. Raja Ampat turns to Bali to help promote tourism. Available: <http://www.tourismindonesia.com/2010/04/raja-ampat-turns-to-bali-to-help.html> [Accessed on April 11, 2010].

Tourism Indonesia. 2008. Scuba diving Raja Ampat trip report Available: http://www.tourismindonesia.com/2008/02/scuba-diving-raja-ampat-trip-report.html [Accessed on April 17, 2008].

Walters C. J., D. Pauly, and V. Christensen. 1997. Structuring dynamic models of exploited ecosystems from trophic mass-balance assessments. Reviews in Fish Biology and Fisheries **7**:139-172.

White A. T., H. P. Vogt, and T. Arin. 2000. Philippine coral reefs under threat: the economic losses caused by reef destruction. Marine Pollution Bulletin **40**(7):598-605.

Wielgus J., A. Balmford, T. B. Lewis, C. Mora, and L. R. Gerber. 2009. Coral reef quality and recreation fees in marine protected areas. Conservation Letters **3**(1):38-44.

Zakai D., N. E. Chadwick-Furman. 2002. Impacts of intensive recreational diving on reef corals at Eilat, northern Red Sea. Biological Conservation **105**(2):179-187.

6 The Influence of Life History Parameters in Fish Population Restoration[34]

6.1 Introduction

Increase in fishing mortality has led to changes in fish community structure; fishing of forage fish has negatively affected the abundance of birds and marine mammals (Jennings and Kaiser 1998). In communities with top predatory fish species, the phenomenon of "fishing down the food web" (Pauly et al. 1998)—smaller faster growing species increasingly dominate the fish catch. It has been estimated that 90% of the large predatory fish have declined (Christensen et al. 2003); several fish stocks have declined (Myers et al. 1996; NAS 1998; Rose and Kulka 1999; Morris et al. 2000; Dulvy et al. 2003; Hutchings and Reynolds 2004; Mullon et al. 2005). Large scale transitions and alternate ecosystem states have been observed (Daskalov et al. 2007). Fish communities have shifted towards relatively homogenous states (Jackson 2008). Several studies reported that large slow growing species declined faster than small fast growing species (Jennings et al. 1999; Ault et al. 2005). International conventions have promoted marine protected areas (MPAs) to restore marine biodiversity (Spalding et al. 2008; Halpern et al. 2010), and as of 2008, about 1 million km^2 (4.9%) of the total continental shelf area was reported to be protected (Spalding et al. 2008). MPAs are valuable sites for empirical studies on species recovery. One of the "primary" results of protection was an increase in fish body size inside the MPAs (Tetreault and Ambrose 2007; Anticamara et al. 2010). Studies on reef fish recovery in MPAs in Kenya showed that recovery patterns were different among different families and size classes (McClanahan et al. 2007). Meta-analyses of recovery inside MPAs found, "large fished species responded strongly to protection and small fished species showed weaker responses" (Molloy et al. 2009);

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similarly, higher trophic level fish showed greater responses to protection (Lester et al. 2009). Some recovery is due to habitat restoration, which is not considered in this chapter.

The available empirical studies suggested relationships between recovery and life history, but at the same time, recovery of a species could extend over a few years or decades (Halpern 2003; Russ and Alcala 2004). Therefore, there is value in pursuing the influence of life history on recovery; the greatest advantage is the ability of "demonstrating broad trends across species" (Vetter 1988). Jennings et al. (2008) combined life history theory with macroecology and food web ecology to predict potential fish biomasses and global trophic structure. Basic life history information is available for a large number of fish species (courtesy: Fishbase); therefore, any broad generalizations reached can be extended over a wide range of species (for example, the use of life history to predict natural mortality in fish Pauly 1980; Hoenig 1983; Vetter 1988). Understanding the relationships between "life history traits and population dynamics is a central goal in ecology" (Goodwin et al. 2006). Fish life history studies show that short-lived species have a higher population growth rate than long-lived species (Jennings et al. 1999; Denney et al. 2002). It has been suggested that life history parameters such as maximum size (Jennings 2000) and age at maturity (Myers et al. 1997; Denney et al. 2002), could be used to predict population recovery rates. In this chapter, the influence of life history parameters of fish species on the potential magnitude of recovery (not the recovery rate i.e. slower or faster recovery) of fish populations is explored.

Two factors determine the magnitude of recovery of any fish population; biomass per recruit and increase in the number of recruits. The biomass per recruit is calculated at various levels of mortality on the fish. Analysis of recruitment is not straightforward because of the compensation in recruitment that occurs at low population levels; the basic phenomenon that is modeled using Beverton-Holt (Beverton and Holt 1957), Ricker (Ricker 1954) and other stock recruitment models. There are five main difficulties associated with estimating the shape of the stock recruitment relations: (1) the curvature of the stock recruit relation emerges only when the population has declined to low levels of stock size; (2) studies, therefore, require long time series of spawner and recruit data (He et al. 2006); (3) the available data show variability and noise in recruit data (Rothschild 2000; He et al. 2006); (4) errors due to time series bias and errors in variable case from error in measurement of spawner data (Walters and Martell 2004); (5) overcompensatory and depensatory responses (Liermann 2001). To analyze the change in mean recruitment with mortality, an index is devised which can be used to express the mean recruitment at any level of total mortality as a percentage of the unfished recruitment. This allows comparison of recruitment in different species at different levels of mortality on a scale from 0 to 100. Then two extremes are assumed for the range within which the recruitment compensation could vary for most species. Finally fish species are grouped according to their life history parameters to explore any overarching patterns in recruitment that are influenced by the life history parameters of the fish species.

6.2 Methods

The patterns in biomass per recruit and number of recruits predicted by standard fisheries assessment models are explored. In the analysis of both biomass per recruit and recruitment, several simplifying assumptions were made. Later, in the results and discussion sections, the implications of the assumptions are discussed. The analysis of recruitment is based on growth parameter data and natural mortality data for around 1800 species obtained from the Fishbase database (Freose and Pauly 2010).

6.2.1 Biomass per recruit (B/R)

The biomass per recruit (B/R) is based on the classic “steady state model” of Beverton and Holt (1957) that describes the state of the stock and the catch. The steady state model is used in a situation where the current level of fishing pressure has been consistent for a long time such that all the fish alive have been exposed to it since they recruited. Hence this chapter is restricted to the analysis of equilibrium conditions and cannot predict time dynamic changes in the species biomass per recruit. The biomass per recruit model expresses the annual average biomass of the exploited part of the cohort.

$$\frac{B}{R} = W_\infty \sum_{n=0}^{n=3} \left\{ U_n [\exp(-(nk(t_r - t_0)))][\frac{1-\exp(-(M+nk)(t_c - t_r)}{M+nk} + \frac{(\exp(-(m+nk)*(t_c - t_r)))*(1-\exp(-((F+M+nk)*(tl - tc))))}{F+M+nk} \right\}$$

where, F = instantaneous rate of fishing mortality, M = instantaneous rate of natural mortality, R = number of recruits, W_∞ = asymptotic weight (grams) calculated as: a*L_∞^b, where a and b are the length-weight parameters, L_∞ is the asymptotic length (cm) of the fish, *k* = von Bertalanffy metabolic growth coefficient (yr^{-1}), t_0 = the initial condition parameter is the theoretical age (yr) at which fish have zero length, t_r = age (yr) at recruitment to fishable stock, t_c = actual age (yr) at first capture with given gear, t_l = maximum age (yr) of fish in stock, and U_n = integration constant necessitated by use of the von Bertalanffy growth model, $U_0 = 1$, $U_1 = -3$, $U_2 = 3$, $U_3 = 1$ (n is only an index used to denote the 4 values of U). For a detailed description of the model and the underlying assumptions, please refer to the fisheries classic ‘Beverton and Holt (1957)’.

B/R can also be modeled using an equilibrium age structure model. The above described formulation is chosen because it requires fewer parameters. The parameter L_∞ is not needed in the model. In the analysis, the t_0 is assumed to be -1. The W_∞ is assumed to be 10,000 g in the calculations. The t_l of the fish is assumed to be 10 years. The implications of these assumptions of asymptotic weight and maximum age are discussed later. The age ‘10 years’ is chosen because roughly 50% of ~1800 species from Fishbase used in the analysis have a maximum age ‘10 years’ or lower. It was also assumed that t_r and t_r are equal to 1. Thus the calculations represent fish populations fully vulnerable to fishing pressure from age 1 onwards. The B/R estimates are calculated at specific values of average annual survival rate and any trend in the relationship between B/R and survival rates were observed. The survival rates range from (1 to 95% per year) and correspond to instantaneous annual total mortality (Z) range from 4.6 to 0.05.

6.2.2 Recruitment

Recruitment is modelled in the analysis using the Beverton and Holt stock recruitment (BH-SR) relationship because it is a very common representation of stock recruitment relationship; additionally, BH-SR relationship is easier to parameterize and discuss compared to other stock recruitment relationships. The BH-SR curve is described as:

$$R = \alpha S / (1 + \beta S)$$

where, R = recruitment, S = breeding stock size. The parameter α is the slope at the origin of the function and represents the maximum recruits per unit breeding stock size at low stock size. Using this equation, the curve approaches an asymptote equal to α/β at increasing levels of spawning stock biomass. "Parameter α increases the height of the asymptote, and β increases the rate of approach to the asymptote" (Jennings et al. 2001).

6.2.2.1 Unfished recruitment R_0

The unfished recruitment R_0, denotes the number of recruits in an unfished population. The standard equation[35] (Walters and Martell 2004) for calculation of R_0 is:

$$R_0 = (\alpha \phi_{e0} - 1) / \beta \phi_{e0}$$

where, ϕ_{e0} the fecundity incidence function represents the fecundity per recruit in an unfished population. The ϕ_{e0} is calculated as the sum over ages of unfished survivorship and fecundity at age. The survivorship at age (l_{age}) is the probability of surviving to each age and is calculated from estimates of mortality (natural mortality (M) is used to calculate the unfished survivorship).

Unfished Survivorship: At age =1, $l_{age} = 1$

At age > 1 $l_{age} = l_{(age-1)} \exp(-M_{(age-1)})$

[35] The formula for R_0 is derived by integrating the area under the curve across the numbers at age, fecundity at age, and relative effect of density-dependent mortality at each age; the parameterization is based on incidence functions (Botsford 1981). Incidence functions are used to describe any property on a per recruit basis. The incidence functions have been further updated to calculate analytical relationships between the parameters (α and β) of the BH-SR function and other parameters R_0, E_0 (unfished eggs per recruit) (Walters and Martell 2004).

The fecundity at age is calculated as the sum-product of weight at age and maturity at age.

Weight at age: $W_{age} = a(l_{age})^b$

Maturity $mat_{age} = \dfrac{1}{1 + \exp\left(\dfrac{-(age - age_{mat})}{sig}\right)}$

The factor *sig* in the above equation refers to the shape of the maturity at age curve, and it can be understood as the standard deviation around the age at maturity. At small values of sig (0.2), the maturity at age curve is steep (almost knife-edge at age at maturity); at high values (~5) the curve almost becomes linear with age. The sig is chosen to be 10% of the age at maturity; the value is arrived at after observing the plots of maturity schedule for several combinations of age at maturity and the ratio of *sig* to the age at maturity.

Fecundity $fec_{age} = w_{age} mat_{age}$

Unfished Fecundity per Recruit $\phi_{e0} = fec_{age} l_{age}$

The R_0 can be expressed as recruitment relative to the asymptote of the Beverton- Holt SR curve. As an arbitrary situation, let us assume that the R_0 is 95% of the asymptote.

$$(\alpha\phi_{e0} - 1)/\beta\phi_{e0} = 0.95 * \alpha/\beta$$

$$\alpha\phi_{e0} - 1 = \beta\phi_{e0} * 0.95 * \alpha/\beta$$

$$\alpha\phi_{e0} - 1 = 0.95\alpha\phi_{e0}$$

$$\alpha\phi_{e0} = 20$$

Hence, the R_0 is 95% of the asymptote when the product of the α parameter and the ϕ_{e0} is 20. The index $(100/\alpha\phi_{e0})$ (here 100/20 =5) represents the percentage difference between R_0 and the asymptote. Therefore, $100-(100/\alpha\phi_{e0})$ =100-5=95 expresses R_0 as a fraction of the asymptote. As the value of $\alpha.\phi_{e0}$ increases above 20, the R_0 approaches the asymptote; if $\alpha.\phi_{e0}$ is 50, then ratio of R_0 is 98% of the asymptote.

Interestingly the product $\alpha.\phi_{e0}$ calculated when a population is unfished (total mortality equals natural mortality) is referred to as compensation ratio. This index is referred to by several notations: CR (Goodwin et al. 2006; Forrest et al. 2008), â (Myers et al. 1999), and K (*kappa*) (Martell et al. 2008; Walters et al. 2008); in this chapter it is referred to as CR. The index was originally proposed and named by Goodyear 1977), and it measures "the relative improvement in juvenile survival at low stock sizes" (Forrest et al. 2008). It is the ratio of the juvenile survival at very low stock size and the juvenile survival at unfished stock size. CR can be calculated as the product of α and ϕ_{e0}, since the reciprocal of ϕ_{e0} represents juvenile survival at unfished stock size (Forrest et al. 2008). The CR has also been defined as the product of α and unfished spawners per recruit ($SPR_{F=0}$) (Myers et al. 1999). But, as explained in Forrest et al. (2008), when the "relative fecundity is described as the product of mean weight-at-age and maturity-at-age, the ϕ_{e0} is the same as $SPR_{F=0}$.

6.2.2.2 Extending the algebra to mean recruitment levels below R_0

The mean equilibrium recruitment at any level of total mortality for a Beverton-Holt SR curve is calculated by the equation (Walters and Martell 2004):

$$R_e = (\alpha\phi_e - 1)/\beta\phi_e$$

This equation is the same as the equation for R_0 except that unfished fecundity per recruit (ϕ_{e0}) is replaced by fished fecundity per recruit (ϕ_e). The ϕ_e is calculated using the same equations as ϕ_{e0} except that in the calculation of survivorship, natural mortality (M) is replaced by total mortality (Z).

Fished Survivorship: At age =1 $l_f = 1$

At age> 1 $l_f = l_{f(age-1)} \exp(-Z_{(age-1)})$

Weight at age: $W_{age} = a(l_{age})^b$

Maturity $mat_{age} = \dfrac{1}{1+\exp\left(\dfrac{-(age-age_{mat})}{sig}\right)}$

Fecundity $fec_{age} = w_{age}.mat_{age}$

Fished Fecundity per Recruit $\phi_e = fec_{age}.l_{age}$

More details about the function and the formula can be obtained from Walters and Martell (2004). Again, the index $\alpha\phi_e$ can be used to express mean recruitment as a percentage of recruitment at the asymptote of the Beverton-Holt SR curve. As an arbitrary situation, let us assume that the mean recruitment is 90% of the asymptote (similar to the earlier formulation) then:

$$(\alpha\phi_e - 1)/\beta\phi_e = 0.9 * \alpha/\beta$$

$$\alpha\phi_e = 10$$

According to the equation, a 10% change from the mean recruitment happens when the product of α and ϕ_e equals to 10. The index $(100/\alpha\phi_e)$ represents the percent decrease in equilibrium recruitment from the asymptote. Thus, if the product $\alpha.\phi_e$ is calculated at any level of mortality on the population, then the mean recruitment at that level can be expressed as a percentage of the asymptote.

6.2.2.3 Steepness

The steepness parameter *h* (Mace and Doonan 1988), also referred to as *z* (Myers et al. 1999), is the ratio of recruitment at 20% spawner abundance to the unfished recruitment. Since the parameter *h* represents the recruitment at a lower (20%) spawning stock biomass, it represents the curvature of the stock recruitment curve. For a Beverton and Holt SR curve, the *h* is analytically related to CR (Myers et al. 1999) as:

$$h = CR/(4 + CR)$$

Also for a Beverton and Holt SR curve, the value of steepness ranges from 0.2 to 1. A value of steepness equal to 0.2 corresponds to a compensation ratio equal to 1; which means recruitment is linearly related to spawning stock biomass (Beddington and Kirkwood 2005) and that there is no compensation. The other extreme value of *h* (1) denotes infinite compensation. Here it is assumed that for most species, the value of the steepness parameter *h* varies from 0.33 to 0.9. Please see Figure 6.1 for interpretation of BH-SR curves at different values of steepness. Of the estimates of steepness available for 56 species (Myers et al. 1999), only 6 species lay outside this range. A steepness value equal to 0.9 specifies that when the spawning stock biomass has decreased to 20%, the mean equilibrium recruitment is 90% of R_0. Thus a steepness value of 0.9 denotes a high level of compensation. At *h*=0.9, the corresponding estimate for CR is 36. If it is assumed that for most of the species the range of steepness parameter is between 0.33 and 0.9, then the range within which the Beverton and Holt α parameter would vary for each species can be calculated by dividing the CR estimates at *h*=0.33 and *h*=0.9 by the estimates of ϕ_{e0}. (because as mentioned earlier CR is a product of Beverton-Holt α parameter and ϕ_{e0}).

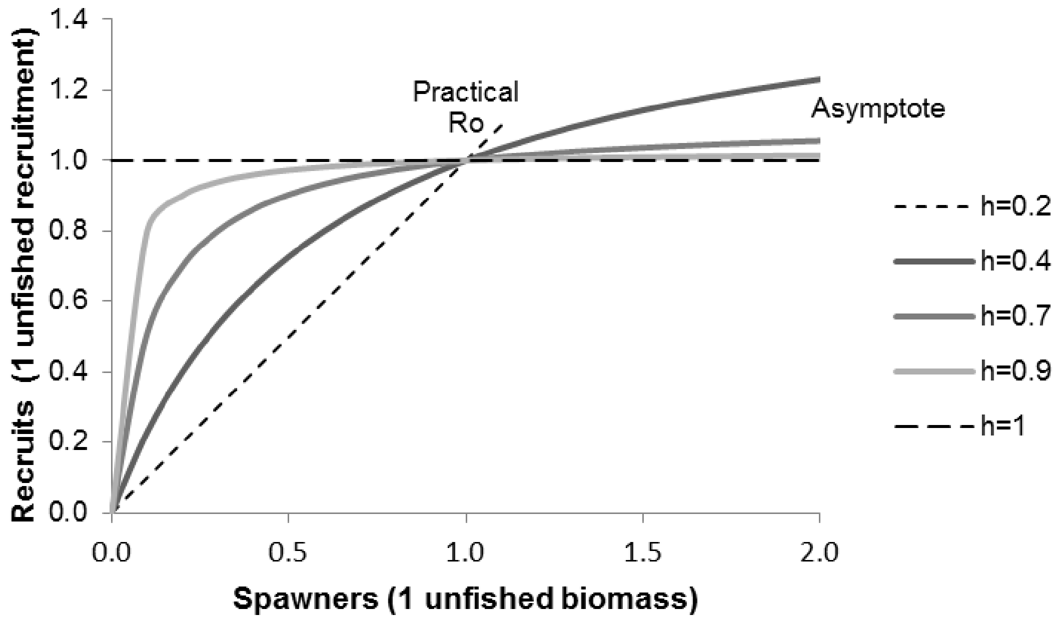


Figure 6.1 Beverton-Holt stock recruitment curves at different values of steepness.
At high levels of steepness the value of practical R_0 is very close to the recruitment at asymptote R_0.is the unfished recruitment and h (steepness) is the is the ratio of recruitment at 20% spawner abundance to the unfished recruitment.

6.2.2.4 Mean equilibrium recruitment for ~1800 species

Unfished fecundity per recruit and alpha parameter (ϕ_{e0})

Life history parameters (L_∞, *k*, t_0, parameters *a* and *b* of length-weight relationship, maximum age, age at maturity) needed for the estimation of ϕ_{e0}, and estimates of natural mortality are available for ~1800 species from the Fishbase database. The unfished fecundity per recruit, ϕ_{e0}, is calculated using equations described in the section 6.2.2.1. It is assumed that the natural mortality remains constant throughout the lifespan of the fish. The two assumed extremes within which the Beverton and Holt α parameter would vary for ~1800 species are calculated as:

$\alpha_{low} = 2/\phi_{e0}$ and

$\alpha_{high} = 36/\phi_{e0}$.

Fished fecundity per recruit (ϕ_e)

i. The fished fecundity per recruit, ϕ_e, is calculated across a range of total mortality (Z) values (0.05 to 4.6) using equations described earlier in section 6.3.2.2. A constant total mortality throughout the lifespan of the fish is assumed. Thus, similar to the calculation of B/R, in the calculation of ϕ_e, it is assumed that the fish have fully recruited to the fishery at age 1.

ii. The product of Beverton-Holt α parameter and ϕ_e are calculated for low compensation and high compensation BH curves as $\alpha_{low}.\phi_e$ and $\alpha_{high}.\phi_e$ respectively.

iii. The mean equilibrium recruitment (measured as a percent of the asymptote) at the low and high compensation levels are calculated as $100-(100/(\alpha_{low}.\phi_e))$ and $100-(100/(\alpha_{low}.\phi_e))$ respectively.

6.3 Results

The results for biomass per recruit and mean recruitment are for the simplified situation when the age 1 fish is fully vulnerable to fishing.

6.3.1 Biomass per recruit

The log of B/R shows a linear trend against total annual survival within a range of survival from 0.1 to 0.7 (Figure 6.2); outside this range of survival values, the relationship becomes curvilinear. The slope of the regression lines decreases exponentially with the increase of von Bertalanffy growth coefficient *k*. At high value of *k* (>2 yr^{-1}), the slope reaches an asymptote (see Figure 6.3a). The high slope for slow growing species indicates that a change in survival has a considerable impact on the size of the population. For fast growing species, the change in B/R with change in survival is comparatively smaller. This indicates that fast growing species can tolerate wider ranges of mortality compared to slow growing fish. The *k* values used in the analysis range from 0.01 to 10 based on the range of *k* values in Fishbase (Freose and Pauly 2010).

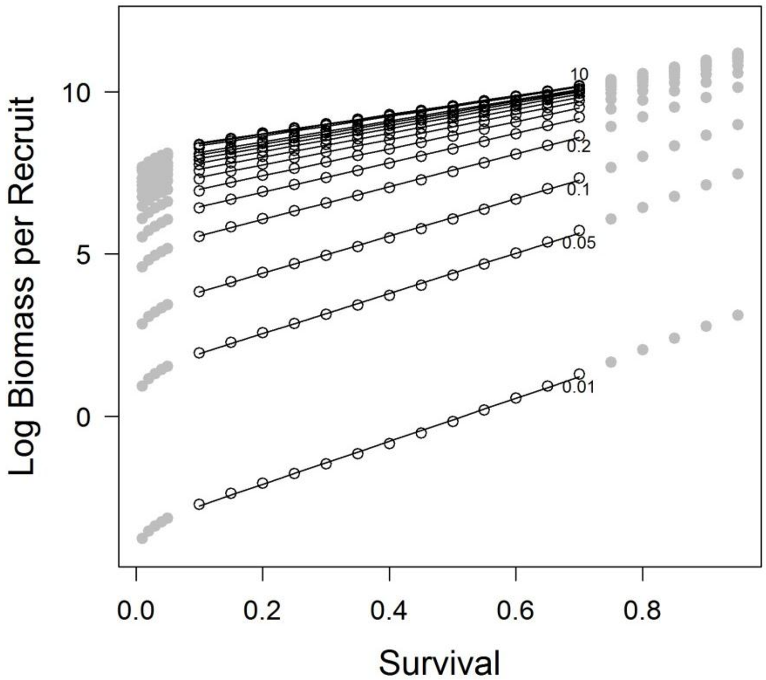


Figure 6.2 Plot of log biomass per recruit against survival.
Only the open dots were used in the regression. For the results shown here W_∞ is 1000g and maximum age of the fish is 10 years. The values adjacent to the lines are the values for growth parameter k.

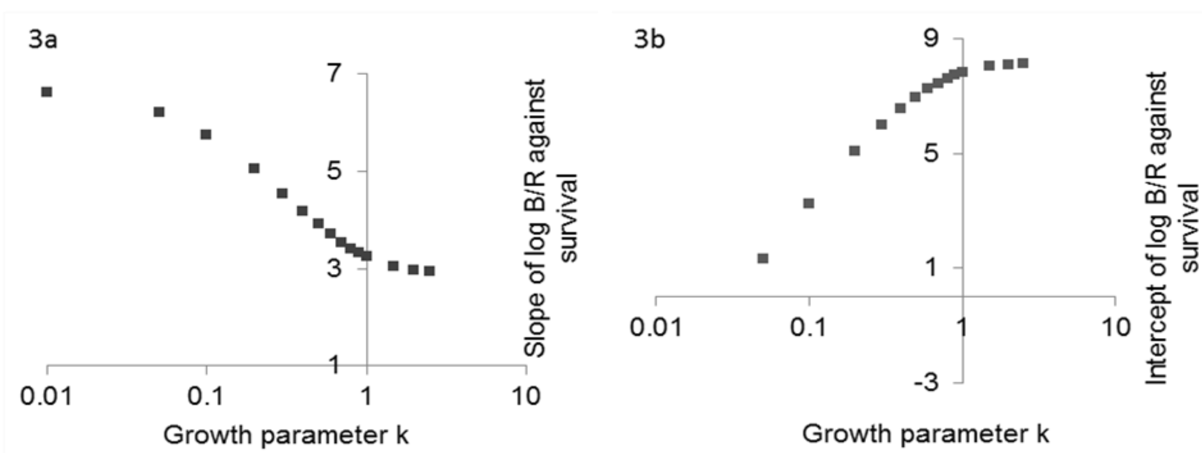


Figure 6.3 Slope and intercept of log biomass per recruit against growth coefficient* k*.
Note* k *is on log scale in both the plots. 3a Plot of change in slope (log B/R against survival) with growth coefficient* k*. 3b Plot of change in intercept (log B/R against survival) with growth parameter* k*. The plot only shows the component governed by growth parameter. The value of the intercept is the sum of log (W_∞) and the component shown in Figure 3b.

For the results seen in Figure 6.2, W_∞ was fixed at 10,000 g. Changing W_∞ only changes the intercept of the regression line and does not change the slope of the relationship between B/R and survival. Thus, the absolute change depends on the W_∞, but the rate of change does not depend on W_∞. The intercept is directly proportional to the sum of $\log(W_\infty)$ and a component that varies negatively with an exponent of *k* (Figure 6.3b).

Changing the maximum age of the fish in the analysis affects the estimates of log B/R obtained at high levels of survival (see grey dots at high levels of survival in Figure 6.4a and 6.4b). High maximum age combined with high survival rates gives higher estimate of B/R. The reason for this observation is that at low rates of survival, the numbers surviving to high ages (>10) are so low that they do not cause a large change in the pattern; therefore, the changes become evident at high survival rates. For slow growing species, the slopes calculated in the section above increase slightly with increase in the maximum age of the fish.

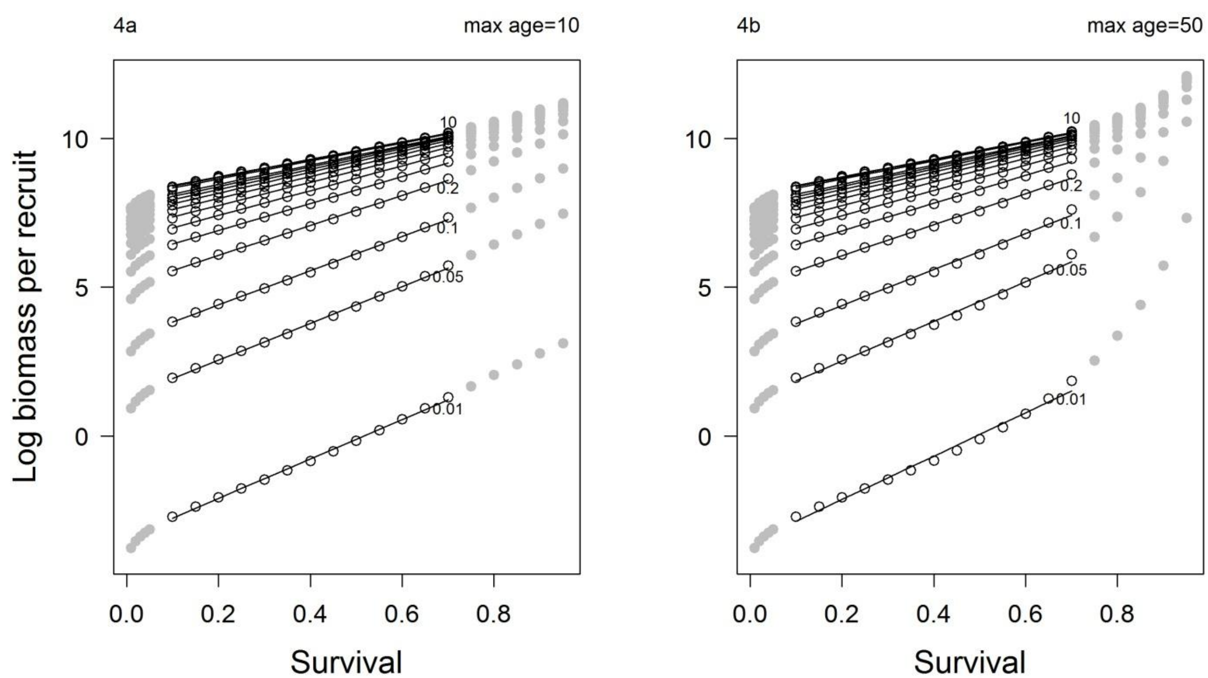


Figure 6.4 Plot of log biomass per recruit against survival at 2 levels of maximum age. Panel a, maximum age equals 10. Panel b, maximum age equals 60. Note the difference in the grey dots at high survival.

6.3.2 Recruitment

The mean recruitment declines with increase in mortality; the decline is faster at low (0.33) value of steepness. When recruitment compensation is high (steepness is high), then the decline in mean recruitment is relatively slower. Curves can be drawn with percent recruitment on the vertical axis and the instantaneous total mortality on the horizontal axis (Figure 6.5). The grey lines in the figure show the mean recruitment for a species at low and high levels of compensation. The black line shows the equilibrium recruitment according to Myer's (1999) estimate of *h*. Such curves allow the comparison of recruitment pattern across species because on the vertical axis, recruitment is referred to as a percentage of the recruitment at the asymptote.

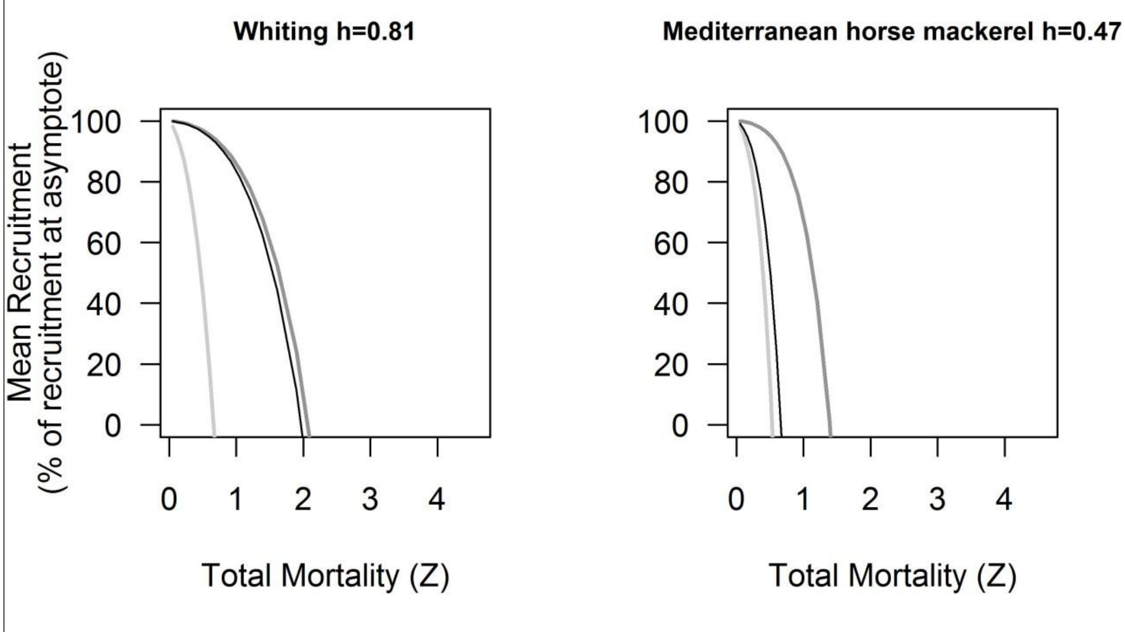
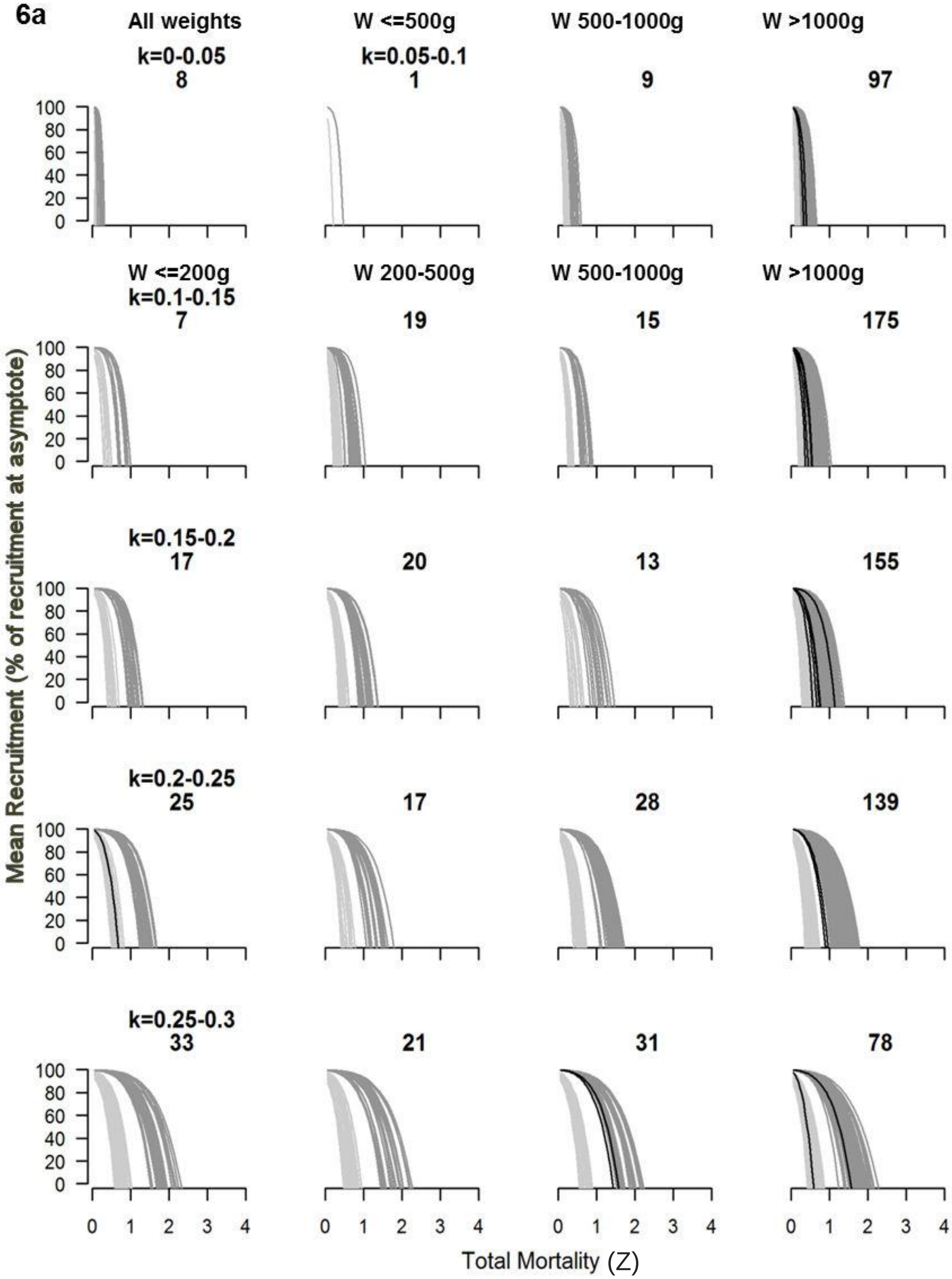


Figure 6.5 Change in equilibrium mean recruitment with change in survival for 2 species. The light grey lines show the mean recruitment at steepness h=0.33 and the dark grey lines show the mean recruitment at steepness h=0.9. The black lines show the lines for mean recruitment based on steepness estimates for the species in Myers et al. (1999).

Similar to the examples above, mean equilibrium recruitment curves at low and high compensation are obtained for ~1800 species. The results are grouped according to the growth parameters (*k* and W_∞) for the purposes of presentation and for exploring any overarching patterns. Figure 6.6a shows the results for species with *k* $<=0.3yr^{-1}$ (for example, Greenland halibut *Reinhardtius hippoglossoide*, Haddock *Melanogrammus* *aeglefinus*, Red snapper *Lutjanus campechanus*, Striped bass *Morone saxatilis*) while Figure 6.6b shows the results for species with k>0.3 yr^{-1} (for example, Whiting *Merlangius merlangus*, Gulf menhaden *Brevoortia patronus*, Anchovy *Engraulis encrasicolus*).


6a
All weights
W <=500g
W 500-1000g
W >1000g
k=0-0.05
k=0.05-0.1
8
1
9
97
W <=200g
W 200-500g
W 500-1000g
W >1000g
k=0.1-0.15
7
19
15
175
k=0.15-0.2
17
20
13
155
k=0.2-0.25
25
17
28
139
k=0.25-0.3
33
21
31
78
Mean Recruitment (% of recruitment at asymptote)
Total Mortality (Z)

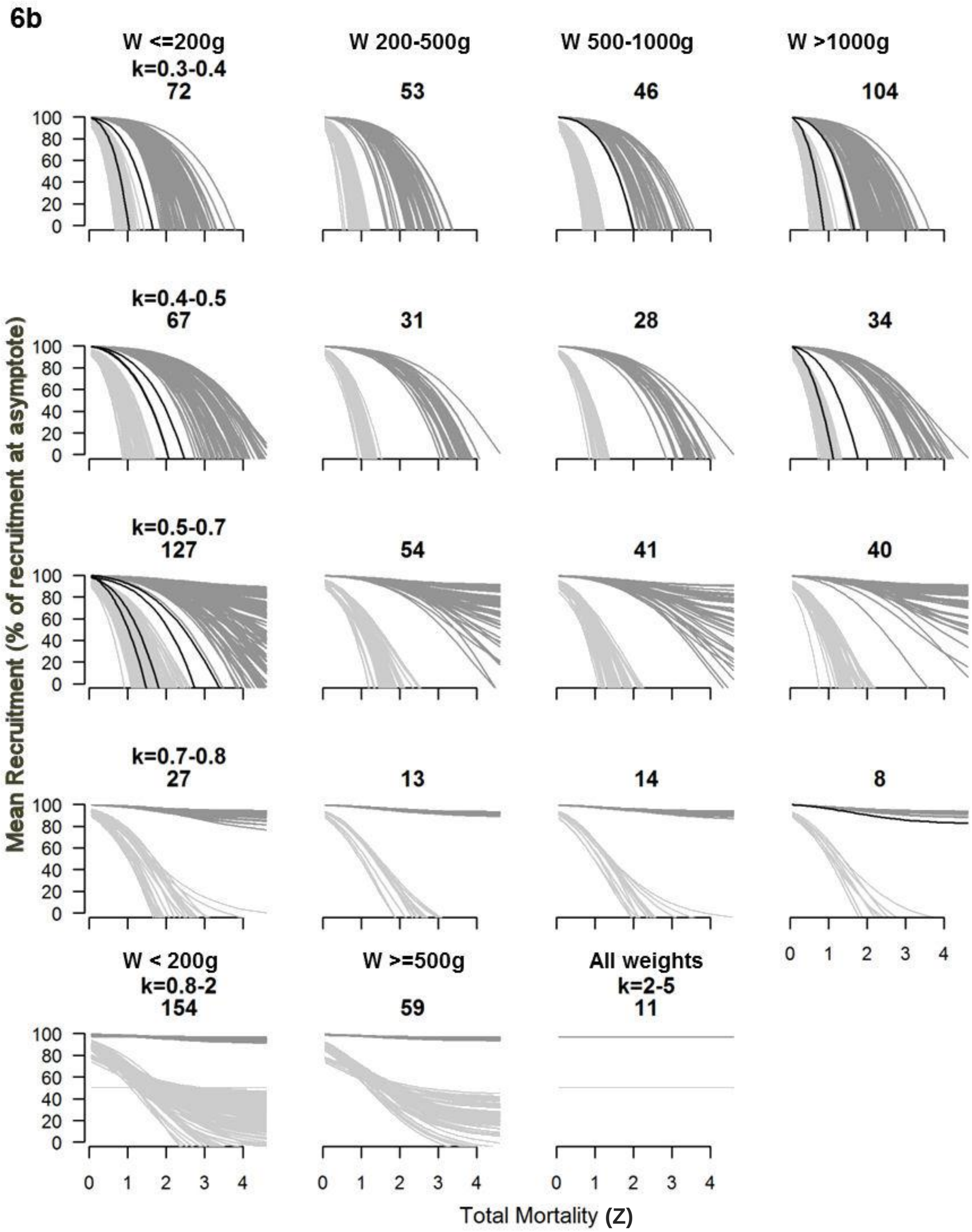


Figure 6.6 Mean recruitment curves for ~1800 species plotted against total mortality Z.
The species are grouped by von Bertalanffy k (rows) and W_∞ (columns). The light grey lines show the calculated mean recruitment when h=0.33 and dark grey lines show the calculated mean recruitment when h=0.9. The black lines show the mean recruitment based on Myer's estimate of h for some species within the range of k and W_∞ in the respective panel . The number on top of each panel shows the number of species plotted. Figure 6a shows species with k<0.3. The first panel of Figure 6a, shows the only 9 species in FishBase database with k<0.05, the next 3 panels (across) show fish species with k in range 0.05 to 0.1. From the second row onwards, the range of k is same across the panel (shown on top of the 1st graph) and the range of W_∞ is same downwards (shown on the top of columns in panel 2). Figure 6b shows species with k>0.3.

The results show that expectations of mean recruitment levels are influenced by the life history parameters. At very low values of *k* and W_{∞}, even high levels of recruitment compensation do not offer a high tolerance to mortality. For such species (under the assumptions on selectivity in the analysis) recruitment rapidly decreases at very low levels of mortality ($Z<0.5$). With increase in *k*, the scope of compensation in recruitment increases.

6.4 Discussion

6.4.1 Selectivity

Fishing gear selectivity is important for stock assessment and management (McClanahan and Mangi 2004; Walters and Martell 2004; Pitcher and Ainsworth 2010). In order to simplify the complications arising from combining the effects of selectivity and maturity, a simplistic situation in which age 1 fish are fully vulnerable to the fishery is modelled.

With increase in the age at first capture, the B/R in the field would be higher for a given level of fishing mortality (to a certain extent before the natural mortality drives the exponential population decline). The calculation of fecundity per recruit 'ϕ_e' also depends on the age at which fish recruit to the fishery. Thus the results discussed in the following sections pertain to the situation when age 1 fish is fully vulnerable to fishing. Species that recruit to the fishery much earlier than they mature are highly vulnerable; the "limits of exploitation" are related to the difference between the age at maturity and the age at capture (Myers and Mertz 1998). A higher age at recruitment into the fishery would push the estimate of ϕ_e higher because greater numbers of individuals would reach the age at maturity. Thus if the fish population is selected to the fishery at a later age (>1) then the population would be able to tolerate higher values of fishing mortality. Therefore, the assumption regarding selectivity is of greatest consequence to the results for large fish that mature later and that are caught by a selective method of fishing.

The assumption made here regarding selectivity would be realistic to some extent for tropical fisheries where the catches (of small fish) from several pelagic fisheries are directed to production of fish meal. Tropical demersal fisheries (mesh size of trawls less than 2 cm) target shrimp for export resulting in bycatch and discards (Pauly et al. 1989). Similarly blast fishing on reefs is not selective for the size of fish. These problems in tropical fisheries are exacerbated by the open access and multi-species (a gear may catch mature fish of one species but immature fish of other species (Gobert 1994), multi-gear nature of the fisheries (one gear may catch mature fish but another gear may catch immature fish). Global scale of by-catch and discards of under-sized fish (Alverson 1994) show that the selectivity determined from landed catch may be different from the real extraction from the fish population. At high fishing pressure, when the older fish reduce in number, then fisheries tend to select smaller age groups within the population; this is known to have led to the collapse of several stocks (Myers and Mertz 1998). The discussion pertaining to restoration, in the following sections, relates to response of species responding to removal of fishing pressure from situations similar to the ones highlighted above.

6.4.2 Biomass per recruit

6.4.2.1 The importance of growth coefficient k

The slope of change in log B/R depends only on the growth coefficient *k*. The implications of the results are that for slow growing species, the effect of decrease in mortality values (for example achieved through setting up of an MPA) would be a relatively steep increase in the B/R of the population. Alternatively for fast growing species, a smaller response (increase) in population size would be observed with decrease in mortality. Strategies for rebuilding should be, therefore, focused on slow growing species. Several studies discuss trophic cascades with respect to restoration dynamics in MPAs (Walters et al. 1999; Graham et al. 2003). The prey species are usually faster growing fish. The analysis shows that because the slope of change in log B/R with survival is less steep for fast growing fish, fast growing fish are relatively more stable against changes in mortality. For the same reason, the response to protection of fast growing fish would be weaker. The result is corroborated by the findings by Ault et al. (2005) who concluded, “overfishing appeared most severe for long-lived, slow growing fish”. Empirical studies of recovery in MPAs have also shown a greater magnitude in response of slow growing fish (Claudet et al. 2006; Molloy et al. 2009). The results also lessen the concern about increased predation mortality inside MPAs, but this author suggests that expected increase in predation mortality on smaller species should be compared against the expected reductions in fishing mortality in the MPAs.

The regressions also show the implications of change in growth. If under fishing pressure the growth rate of the species increases (i.e. *k* increases), then the species would climb upward on Figure 6.2. At the higher *k*, the slope of log B/R would be lower (exponentially) and would allow greater tolerance to mortality. If after the fishing pressure is released, the growth rate does not revert to the earlier lower level, then the B/R attained at higher survival rates would be lower than the B/R levels attained at the same survival rate had the *k* not changed.

6.4.2.2 The importance of Weight infinity and maximum age

The intercepts of the regression of log B/R against survival are dependent on W_∞. Therefore in a comparison of two species with equal *k* but different W_∞, the species with higher W_∞ will show a greater absolute increase in B/R. At any value of *k*, a higher W_∞ signifies a higher B/R; this might imply a higher advantage against change in mortality. Fish species growing to older ages show an exponential increase in log B/R at survival rates higher than 80%. Higher maximum age is related to lower *k* and lower natural mortality rates. If it is expected that survival rates of more than 70% are possible in the no-take areas, then no-take areas will give a quicker and greater increase than allowing limited fishing mortality (effort reduction inside protected area) for a prolonged period. For all ranges of the *k* parameter, it is observed that at average annual survival rates lower than 20% (Z=2.7), the log B/R curve decreases rapidly compared to decrease in survival rate (under the assumption of selectivity in the analysis). If the fisheries in a region are non-selective in nature then at such mortality rates effort reductions or MPAs should be implemented to arrest the decline of the population biomass.

6.4.3 Recruitment

6.4.3.1 Implications of assumptions

In this section, first the implications of the data used and the assumptions made in section 2.2 on recruitment are discussed. Before proceeding with the discussion, this author would like to stress that these assumptions have allowed a clear pattern to emerge.

i. Estimates of M in Fishbase are not based on empirical relationships; however a few of the estimates of natural mortality have probably been calculated using empirical relationships (for example the Pauly (1980) formula which relates M to growth parameters k and L_∞). Examples of such relationships have been reviewed in Vetter (1988). In such a case, any variations in natural mortality around the empirical regression relationships have been missed in this analysis. Estimates of M obtained from direct observations in the field could disperse the patterns observed here.
ii. The ϕ_{e0} is calculated as the sum over ages of survivorship and fecundity at age; the fecundity is calculated as the product of weight at age and maturity at age. Therefore, it is assumed that above the age at maturity, the fecundity is proportional to body weight. This assumption has been made in similar studies (Goodwin et al. 2006; Forrest et al. 2008) and the assumption increases the correlation between ϕ_{e0} and W_∞ for any species. Constant natural mortality is also a common assumption used for the calculations of ϕ_{e0}.
iii. No uncertainty was allowed on the life-history or natural mortality parameters from Fishbase. Small changes in life history parameters (L_∞, k, t_0, parameters a and b of length-weight relationship, maximum age, age at maturity) can affect the weight at age used in the calculation of ϕ_e and bias the value upward or downward. The results show only the deterministic equilibrium. Here it is hoped that the uncertainty is accounted for by using a large number of fish species in the analysis.
iv. The Beverton-Holt stock recruitment curve was used to model recruitment because the steepness and compensation ratio are analytically related by a

simple relationship, and the steepness ranges between the extremes 0.2 and 1. For species that follow overcompensatory (Ricker) relationships, the mean recruitment at high survival rates (lower levels of mortality) will be lower than the highest values for mean recruitment for the species. Complex patterns like depensation effects at low stock sizes have also been ignored (Liermann 2001).

6.4.3.2 Overarching patterns relating life history with recruitment

Species with the lowest *k* and highest W_∞ combination (large, slow growing species) are those which have the longest lifespans (50-100 years) and the lowest natural mortality rates (0.04 to 0.12). Compensation in recruitment allows the population to survive at total mortality levels almost double the natural mortality (0.1 to 0.25) on the population. When these species experience much high mortality rates, the ϕ_e declines rapidly, and the compensation offered by increased numbers of recruits at low population levels is small relative to the decline in ϕ_e. This is the reason why even with high compensation, for species with $k<0.1$ yr^{-1}, the mean equilibrium recruitment declines much rapidly compared to species with higher *k*. It was assumed that fish were fully vulnerable to fishing from age 1. The result shows that if slow-growing late maturing species are vulnerable to fishing from age 1, then these species would decline at very low levels of fishing pressure. The ϕ_e declines rapidly because very few fish reach the age at the maturity. Therefore, it is for such species that the age at which the fish become vulnerable to fishing is of utmost importance. If the fish become vulnerable at a later age, then the decline in ϕ_e would be more gradual and the species will be more tolerant to higher fishing pressure. The results obtained here stress the influence selectivity has on the amount of fishing pressure which can be exerted on fish populations, especially long-lived species like rockfish and orange roughy.

The ratio between the estimate of BH α parameter at high compensation (h=0.9, CR=36) and at low compensation (h=0.33, CR=2) for any species in this analysis is always 18 (36/2) irrespective of whether the actual value for α is 10 or 1000 or higher. For example, at two levels of compensation CR = 2 and CR=36, species A has ϕ_{e0}=10, and species B has ϕ_{e0}=2. For species A, the α at low and high compensation levels will vary between 0.2 and 3.6. For species B, the α at low and high compensation levels will vary between 1 and 18. Increase in W_∞ within any given range of *k* corresponds to a longer lifespan and lower natural mortality, both factors that lead to a higher estimate of ϕ_{e0}. A higher estimate of ϕ_{e0} leads to lower estimate of BH α parameter. The negative correlation between BH-SR α parameter and ϕ_{e0} is also reported from analysis using real spawner recruit data (Denney et al. 2002; Goodwin et al. 2006)

In the earlier section on growth, the results showed that the slope of change in B/R depended on *k*. Besides the information used in calculation of B/R—the weight at age and survivorship at age—the maturity schedule is only the other information required to calculate ϕ_e. Consequently, the shape of decline in ϕ_e also depends on *k*. Since the percentage change in mean recruitment is dependent on the product of BH-SR α parameter and ϕ_e, the shape of the decline in mean recruitment is highly influenced by the shape of decline of ϕ_e with mortality (the shape of ϕ_e as altered by the levels of compensation). In the example above, if species A and B have the same value for *k*, then the mean recruitment curves at low and high compensation will be similar. This is the reason why the shapes of the mean recruitment curves are very similar within the same range of *k* (across the panels in Figures 6.6a and 6.6b). It is to be noted that the results do not imply that species within the same range of *k* have the same recruitment pattern. The results show that if species within the same range of *k* have the same level of recruitment compensation, then the pattern of change in recruitment will be similar.

In the section on biomass per recruit, the results showed that with increase in *k*, the rate of change in B/R with change in survival decreased (i.e. the slope decreased). Similarly, the rate of change in ϕ_e with change in survival decreases with increase in *k*. Therefore as growth rate increases, the ϕ_e spreads across a larger range of mortality values. For fast growing species (k >0.4 yr^{-1}) depending on the level of compensation, recruitment curves spread over a wider range of total mortality from 0.5 to 4 or higher. Even at low levels of compensation (h=0.33), the decline in mean recruitment (with increase in mortality) for species in this category is gradual. Also, for the fast growing species, a small increase in compensation can cause a huge difference in the shape of mean recruitment curve. The response of biomass per recruit and mean recruitment against change in mortality are steeper for slow growing fish, therefore, the magnitude of decline and recovery would be larger. The results from recruitment also suggest a higher (in terms of magnitude) response from protection of slow growing species, the result corresponds with empirical findings (Claudet et al. 2006; Molloy et al. 2009).

6.4.3.3 Compensation Ratio

The ϕ_{e0} is correlated with body size (Goodwin et al. 2006) because it is derived from growth parameters. In Figures 6.6a and 6.6b, the highest estimates for ϕ_{e0} are for the species in the top right panel because these species have the highest range for W_∞ and the longest lifespans. The estimate of ϕ_{e0} decreases from right to left with decrease in W_∞. Based on a comparison of 54 stocks, Goodwin et al. (2006) found that stocks with high ϕ_{e0} ("large bodied late maturing") had high CR and concluded that ϕ_{e0} "proved to be the best single predictor of both α and CR". A meta-analysis of ~200 North American freshwater and marine species (Rose et al. 2001) categorized fish species as "*periodic* – large highly fecund fish with long life spans", "*opportunistic* – small rapidly maturing short lived species", and "*equilibrium* – intermediate size producing relatively large offspring and showing parental care" (Rose et al. 2001). Cod and tuna are examples of *periodic* species; anchovies, killifishes are examples of *opportunistic* species, sculpins and marine catfish are examples of *equilibrium* species (Winemiller and Rose 1992). Rose et al. (2001) analyzed the steepness parameter for these categories of species and found that the average steepness value was highest for the category *periodic* (0.7), and lower for the categories *opportunistic* (0.55) and *equilibrium* (0.57). *Opportunistic* strategists "inhabited highly variable environments and seldom approached environmental carrying capacity and were expected to show high inter-annual variation"; indicating lower levels of compensation for *opportunistic* species (Rose et al. 2001). If these observations (high unfished fecundity per recruit indicates high compensation and vice versa) could be extended to all species, then it would mean that in Figures 6.6a and 6.6b, the actual mean recruitment curves would be closer to the dark grey lines in the panels on the right, and those curves would be closer to the light grey lines in the panels on the left. The results from this analysis can be used only to adjudge broad overarching patterns. Species adopt a large number of strategies for recruitment compensation. The results in this chapter indicate the following hypothesis: The influence of environment can be confused, so it is categorized into 2 forms: (1) The influence of the environment improves the reproductive condition or size of the spawning stock resulting in an improved recruitment; (2) Recruitment is related to environment because it seems to be independent of spawning stock size. At high compensation, recruitment is relatively unchanging with change in spawning stock; recruitment one year becomes the spawning stock years (age at maturity) later when the cohort matures. *Therefore, after controlling for extremes in fishing pressure, the spawning stock (in numbers) that shows low variation probably indicates high compensation; a high variation in spawning stock probably indicates low compensation.*

Compensation ratio is the ratio of recruit survival at low versus high stock sizes. A compensatory increase in the number of recruits at low stock sizes can be a result of higher survival at low stock sizes owing to lesser competition and predation at these levels. Increased compensation can also be due to relatively improved recruit production, i.e., increase in ϕ_e. Several species exhibit compensation by change in age and size at maturity, fecundity, spawning frequency, sex, etc (see review by Rose et al. 2001). All these changes would lead to a deviation from the model estimate for the ϕ_e, but these changes are not considered in the work here.

The results are based on the assumption that the steepness parameter for most species ranges between 0.33 and 0.9. Myers et al. (1999) reported 4 species (Ayu *Plecoglossus altivelis*, Scup *Stenotomus chrysops*, New Zealand snapper *Pagrus auratus*, Red snapper) with higher values of steepness. Assuming no compensation (h=0.2) is highly precautionary and for the same reason highly uneconomical (Rose et al. 2001). Low probabilities have been associated with steepness values lower than 0.3 calculated using life history information, recruitment variability and "low critical abundance of the population (He et al. 2006); however the results were sensitive to the choice of low critical abundance. It is possible that several species have the biological ability for high compensation (h>0.9). Expecting higher levels of compensation also means expecting that all other environmental factors (for example timing of plankton bloom (Cushing 1990; Minto et al. 2008), optimum temperature (Myers 1998), and abundance of predatory species) would also be in perfect harmony in the years when higher compensation is expected to materialize. Study of survival variability (Minto et al. 2008) at low spawning stock shows that strong density dependence (high CR) is associated with high survival variability; the "increased variance results in high extinction risk". The extreme (h=1) is "inconsistent biologically and inconsistent with precautionary approach" (Mangel et al. 2010).

Here only the equilibrium change from the perspective of a unit fish stock is considered, but in practice actual recovery would be dependent on several factors (each MPA would not contain a unit stock and spatial parameters like migration would influence biomass change) which are not discussed. Also, issues related to food web structure (Walters et al. 2008), meta-population connectivity (Jennings 2000), habitat, or the difficulty of "reducing fishing mortality on collapsed populations to zero" (Hutchings 2000; Hutchings and Reynolds 2004) are not discussed here.

6.5 Conclusion

The main findings— in both biomass per recruit and mean recruitment for fish populations which are fully vulnerable to fishing from age 1 onwards—are that fast growing species are much more resilient against changes in mortality either from fishing or predation pressure. When fishing pressure is decreased, an increase in biomass per recruit will be observed; the increment will be larger for slow growing fish. Whether recruitment will increase at lower fishing pressure will depend on how much the recruitment has declined from the unfished level. The mean recruitment for slow growing species declines at lower levels of mortality, so it could be expected that even small declines in fishing mortality would result in improved recruitment of slow growing fish. This analysis provides approximate estimates of expected change in equilibrium population biomass due to restoration. In conclusion, fast growing species would show a quicker (Denney et al. 2002) but smaller response to protection and slow growing species would show a slower but larger response to protection.

6.6 References

Alverson D. L. 1994. A global assessment of fisheries bycatch and discards. FAO Fisheries Technical Paper **339**:1-233.

Anticamara J. A., D. Zeller, and A. C. J. Vincent. 2010. Spatial and temporal variation of abundance, biomass and diversity within marine reserves in the Philippines. Diversity and Distributions **16**(4):529-536.

Ault J. S., S. G. Smith, and J. A. Bohnsack. 2005. Evaluation of average length as an estimator of exploitation status for the Florida coral-reef fish community. ICES Journal of Marine Science **62**(3):417.

Beddington J. R., G. P. Kirkwood. 2005. The estimation of potential yield and stock status using life–history parameters. Philosophical Transactions of the Royal Society B: Biological Sciences **360**(1453):163-170.

Beverton R. J. H., S. J. Holt. 1957. On the dynamics of exploited fish populations. Chapman and Hall, London, 1957; Facsimile reprint 1993.

Botsford L. W. 1981. The effects of increased individual growth rates on depressed population size. The American Naturalist **117**(1):38-63.

Christensen V., S. Guenette, J. J. Heymans, C. J. Walters, R. Watson, D. Zeller, and D. Pauly. 2003. Hundred-year decline of North Atlantic predatory fishes. Fish and Fisheries **4**(1):1-24.

Claudet J., D. Pelletier, J. Y. Jouvenel, F. Bachet, and R. Galzin. 2006. Assessing the effects of marine protected area (MPA) on a reef fish assemblage in a northwestern Mediterranean marine reserve: Identifying community-based indicators. Biological Conservation **130**(3):349-369.

Cushing D. H. 1990. Plankton production and year-class strength in fish populations: an update of the match/mismatch hypothesis. Advances in Marine Biology **26**:249-293.

Daskalov G.M., A.N. Grishin, S. Rodionov, and V. Mihneva. 2007. Trophic cascades triggered by overfishing reveal possible mechanisms of ecosystem regime shifts. PNAS 104(25): 10518-10523.

Denney N. H., S. Jennings, and J. D. Reynolds. 2002. Life–history correlates of maximum population growth rates in marine fishes. Proceedings of the Royal Society of London. Series B: Biological Sciences **269**(1506):2229-2237.

Dulvy N. K., Y. Sadovy, and J. D. Reynolds. 2003. Extinction vulnerability in marine populations. Fish and Fisheries **4**(1):25-64.

Forrest R. E., S. J. D. Martell, M. C. Melnychuk, and C. J. Walters. 2008. An age-structured model with leading management parameters, incorporating age-specific selectivity and maturity. Canadian Journal of Fisheries and Aquatic Sciences **65**(2):286-296.

Freose R., D. Pauly. 2010. FishBase. World Wide Web electronic publication. Available: <www.fishbase.org, version >(05/2010).

Goodwin N. B., A. Grant, A. L. Perry, N. K. Dulvy, and J. D. Reynolds. 2006. Life history correlates of density-dependent recruitment in marine fishes. Canadian Journal of Fisheries and Aquatic Sciences **63**(3):494-509.

Goodyear C. P. 1977. Assessing the impact of power plant mortality on the compensatory reserve of fish populations. Pages 186-195 In: W. van Winkle, editor. Proceedings of the Conference on Assessing the Effects of Power Plant Induced Mortality on Fish Populations, 3–6 May 1977. Pergamon Press. New York.

Graham N. A. J., R. D. Evans, and G. R. Russ. 2003. The effects of marine reserve protection on the trophic relationships of reef fishes on the Great Barrier Reef. Environmental Conservation. **30**:200-208.

Halpern B. S. 2003. The impact of marine reserves: do reserves work and does reserve size matter? Ecological Applications **13**(sp1):117-137.

Halpern B. S., S. E. Lester, and K. L. McLeod. 2010. Placing marine protected areas onto the ecosystem-based management seascape. Proceedings of the National Academy of Sciences doi:10.1073/pnas.0908503107.

He X., M. Mangel, and A. MacCall. 2006. A prior for steepness in stock-recruitment relationships, based on an evolutionary persistence principle. Fishery Bulletin **104**(3):428-433.

Hoenig J. 1983. Empirical use of longevity data to estimate mortality rates. Fisheries Bulletin **82**:898-903.

Hutchings J. A. 2000. Collapse and recovery of marine fishes. Nature **406**(6798):882-885.

Hutchings J. A., J. D. Reynolds. 2004. Marine fish population collapses: consequences for recovery and extinction risk. Bioscience **54**(4):297-309.

Jackson J. B. C. 2008. Ecological extinction and evolution in the brave new ocean. Proceedings of the National Academy of Sciences **105**(Supplement 1):11458-11465.

Jennings S. 2000. Patterns and prediction of population recovery in marine reserves. Reviews in Fish Biology and Fisheries **10**(2):209-231.

Jennings S., M.J. Kaiser., 1998. The effects of fishing on marine ecosystems. Advances in Marine Biology **34**:201–352.

Jennings S., M. J. Kaiser, and J. D. Reynolds. 2001. Marine fisheries ecology. Blackwell Publications, Oxford. 417p.

Jennings S., S. P. R. Greenstreet, and J. D. Reynolds. 1999. Structural change in an exploited fish community: a consequence of differential fishing effects on species with contrasting life histories. Journal of Animal Ecology **68**(3):617-627.

Jennings S., F. Melin, J. L. Blanchard, R.M. Forster, N. K. Dulvy, and R.W. Wilson. 2008. Global-scale predictions of community and ecosystem properties from simple ecological theory. Proceedings of the Royal Society of London. Series B: Biological Sciences **275**(1641):1375-1383.

Gobert B. 1994. Size structures of demersal catches in a multispecies multigear tropical fishery. Fisheries Research **19**:87-104.

Lester S. E., B. S. Halpern, K. Grorud-Colvert, J. Lubchenco, B. I. Ruttenberg, S. D. Gaines, S. Airamé, and R. R. Warner. 2009. Biological effects within no-take marine reserves: a global synthesis. Marine Ecology Progress Series **384**:33-46.

Liermann H. 2001. Depensation: evidence, models and implications. Fish and Fisheries **2**(1):33-58.

Mace P. M., I. J. Doonan. 1988. A generalized bioeconomic simulation model for fish population dynamics. New Zealand Fisheries Assessment Research Document. **88**(4).

Mangel M., J. Brodziak, and G. DiNardo. 2010. Reproductive ecology and scientific inference of steepness: a fundamental metric of population dynamics and strategic fisheries management. Fish and Fisheries **11**(1):89-104.

Martell S. J. D., W. E. Pine, and C. J. Walters. 2008. Parameterizing age-structured models from a fisheries management perspective. Canadian Journal of Fisheries and Aquatic Sciences **65**(8):1586-1600.

McClanahan T. R., N. A. J. Graham, J. M. Calnan, and M. A. MacNeil. 2007. Toward pristine biomass: reef fish recovery in coral reef marine protected areas in Kenya. Ecological Applications **17**(4):1055-1067.

McClanahan T. R., S. C. Mangi. 2004. Gear-based management of a tropical artisanal fishery based on species selectivity and capture size. Fisheries Management and Ecology **11**:51–60.

Minto C., R. A. Myers, and W. Blanchard. 2008. Survival variability and population density in fish populations. Nature **452**(7185):344-347.

Molloy P. P., I. B. McLean, and I. M. Côté. 2009. Effects of marine reserve age on fish populations: a global meta-analysis. Journal of Applied Ecology **46**(4):743-751.

Morris A. V., C. M. Roberts, and J. P. Hawkins. 2000. The threatened status of groupers (Epinephelinae). Biodiversity and Conservation **9**(7):919-942.

Mullon C., P. Freon, P Cury. 2005. The dynamics of collapse in world fisheries. Fish and Fisheries **6**:111-120.

Myers R. A., J. A. Hutchings, and N. J. Barrowman. 1996. Hypotheses for the decline of cod in the North Atlantic. Marine Ecology Progress Series. **138**(1):293-308.

Myers R. A., K. G. Bowen, and N. J. Barrowman. 1999. Maximum reproductive rate of fish at low population size. Canadian Journal of Fisheries and Aquatic Sciences **56**(12):2404-2419.

Myers R. A. 1998. When do environment–recruitment correlations work? Reviews in Fish Biology and Fisheries **8**(3):285-305.

Myers R. A., G. Mertz. 1998. The limits of exploitation: a precautionary approach. Ecological Applications **8**(1) Supplement:S165-S169.

Myers R. A., G. Mertz, and P. Fowlow. 1997. Maximum population growth rates and recovery times for Atlantic cod, *Gadus morhua*. Fishery Bulletin **95**(4):762-772.

National Academy of Sciences 1998. Review of Northeast Fishery Stock Assessments. Committee to Review Northeast Fishery Stock Assessments, National Research Council Committee constituted by Commission on Geosciences, Environment and Resources (CGER), Ocean Studies Board (OSB) and Earth and Life Studies (DELS). Published by the National Academics Press. 134p.

Pauly D. 1980. On the interrelationships between natural mortality, growth parameters, and mean environmental temperature in 175 fish stocks. ICES Journal of Marine Science **39**(2):175-192.

Pauly D., G. Silvestre, and I. R. Smith. 1989. On development, fisheries and dynamite, a brief review of tropical fisheries management. Natural Resource Modeling **3**(3):307-329.

Pauly D., V. Christensen, J. Dalsgaard, R. Froese, and F. Torres Jr. 1998. Fishing Down Marine Food Webs. Science **279**(5352):860-863.

Pitcher T. J., C. H. Ainsworth. 2010. Resilience to change in two coastal communities: Using the maximum dexterity fleet. Marine Policy **34**(4):810-814.

Ricker W. E. 1954. Stock and recruitment. Journal of Fisheries Research Board of Canada **11**:559-623.

Rose G. A., D. W. Kulka. 1999. Hyperaggregation of fish and fisheries: how catch-per-unit-effort increased as the northern cod (*Gadus morhua*) declined. Canadian Journal of Fisheries and Aquatic Sciences **56**(S1):118-127.

Rose K. A., J. H. Cowan, K. O. Winemiller, R. A. Myers, and R. Hilborn. 2001. Compensatory density dependence in fish populations: importance, controversy, understanding and prognosis. Fish and Fisheries **2**(4):293-327.

Rothschild B. J. 2000. " Fish stocks and recruitment": the past thirty years. ICES Journal of Marine Science **57**(2):191-201.

Russ G. R., A. C. Alcala. 2004. Marine reserves: long-term protection is required for full recovery of predatory fish populations. Oecologia **138**(4):622-627.

Spalding M., L. Fish, and L. Wood. 2008. Toward representative protection of the world's coasts and oceans—progress, gaps, and opportunities. Conservation Letters **1**(5):217-226.

Tetreault I., R. F. Ambrose. 2007. Temperate marine reserves enhance targeted but not untargeted fishes in multiple no-take MPAs. Ecological Applications **17**(8):2251-2267.

Vetter E. F. 1988. Estimation of natural mortality in fish stocks: A review. Fishery Bulletin **86**(1):25-43.

Walters C., D. Pauly, and V. Christensen. 1999. Ecospace: prediction of mesoscale spatial patterns in trophic relationships of exploited ecosystems, with emphasis on the impacts of marine protected areas. Ecosystems **2**(6):539-554.

Walters C. J., S. J. D. Martell. 2004. Fisheries ecology and management. Princeton University Press, New Jersey. 399p.

Walters C. J., R. Hilborn, and V. Christensen. 2008. Surplus production dynamics in declining and recovering fish populations. Canadian Journal of Fisheries and Aquatic Sciences **65**(11):2536-2551.

Winemiller K. O., K. A. Rose. 1992. Patterns of life-history diversification in North American fishes: implications for population regulation. Canadian Journal of Fisheries and Aquatic Sciences **49**:2196-2218.

7 Conclusion

Chapter 1

Chapter 1 of the dissertation reviewed the history of fisheries management, history of resource management in Raja Ampat and set up the theoretical and practical background for the dissertation. The context for the thesis was set in 2 major worldviews in fisheries research: (1) A long period of exploitation has led to decline of marine fish populations, and thus effective marine restoration has become an important concern in current fisheries research. (2): The interactions between fish populations in the ecosystem and the human influence are to be considered simultaneously.

Chapter 2

Chapter 2 of my thesis explored the technological, socio-economic, and political history of Raja Ampat to measure the illegal, unreported and unregulated IUU catch in the region. Results showed that IUU catch exceeded the reported catch by more than 40,000 tonnes (or a factor of 1.5) for the year 2006. The average annual revenue from the IUU catch in Raja Ampat during 2003-2006 was 40 million USD. The IUU fishing by local fishers either due to lack of proper enumeration methods or due to sale to middlemen from elsewhere was a large proportion of total extraction from the resource; a similar result in Indonesia was observed for the sardine fishery in Bali strait (Buchary et al. in press). A global analysis of the scope of illegal fishing showed that incentives to indulge in IUU were far greater than the “costs of being apprehended” (Sumaila et al. 2006). On account of the remoteness of the region (reduced policing) and the relatively shorter history of exploitation (better resource status), it could be imagined that the advantages of IUU fishing in remote parts of Eastern Indonesia would be greater than the global average.

Study of history of events in the region led to an understanding of the management challenges in the region. The chapter recounted several factors that led fishers to engage in IUU and showed that several factors considered extraneous to the resource management problem influenced IUU. The main management challenges included population influx and technological advances that led to a shift from a predominantly subsistence based lifestyle to a cash based economy (WWF/IUCN 1996). Disappointment over unregulated access, and perception that their paradise was being plundered by outsiders led young fishers in Raja Ampat to engage in destructive fishing (Halim and Mous 2006). In addition, depletion of marine resources in other parts of Indonesia augmented with increased demand for live reef fish, sea cucumber, and lobsters from markets outside Indonesia led to increased pressure on resources in Raja Ampat. The wide geographic spread of Raja Ampat Regency across ~600 islands and limited management capacity of the government further exacerbated the challenges of managing the marine resources in the region.

Conservation minded non-governmental organizations (NGOs) considerably increased public awareness on the destructive effects of cyanide and blast fishing. In fact, many Raja Ampat fishers stopped blast and cyanide fishing and shifted to longlines and gillnets. The villagers and the NGOs created informal partnerships to patrol the region; this showed the villagers' willingness to participate in management. The work in Chapter 2 showed that there was value in educating the fishers about resource use and in engaging them in the management process. It emphasized the need for considering and addressing the social and ecological aspects of the marine ecosystem together. A similar process of educating fishers and encouraging dialogue between different stakeholder groups was successful in Bunaken National Park in Indonesia; the fishers, NGOs, and local management body entered into partnership to monitor the reserve, and increased success was observed when fishers were part of the monitoring process (Erdmann et al. 2004). Decentralization of political power gave considerable rights to the Raja Ampat Regency government to manage the coastal marine resources. The regency government entered into collaboration with NGOs to develop ecosystem based management (EBM) of the region and established a network of marine protected areas in the region. Indonesia as a country performed poorly in an evaluation of progress in EBM, but successful implementation of collaborative management projects is expected to lead to considerable improvement (Pitcher et al. 2009 presented in Appendix F). Integration of ecological, economic, social, cultural, and political factors with "constructive articulation of top-down approaches and development of bottom-up or grassroots initiatives" is required for sustainable development (Gallopín et al. 2001; Olsson et al. 2008).

Research in Indonesia showed that commitment from the local residents was essential for effective management (Alder et al. 1994); co-management improved enforcement by encouraging compliance (Elliott et al. 2001; Crawford et al. 2004). Success in management of no-take reserves was observed in Sumilon and Apo islands in Philippines where the local communities, the local management body, and the National government "comanage" the marine resources (Alcala and Russ 2006). Compliance with management goals was correlated with "democratic decision making" at the community level (Pollnac et al. 2001). Chapter 2 of my thesis re-emphasized that economic and social factors could have considerable impact on resource use. Understanding the perceptions of the local community towards the environment and the management regime are very important in both identification and successful implementation of management methods.

An interesting direction for further research in this area could be to explore the influence of cohesiveness in fisher community in the adherence to fisheries management regulations. Here the word community means "fishers working in a certain manageable geographic unit—might be a village or a district"; however, this might be different from how the fishers associate or identify themselves as "community" due to ethnic, income, religious, or other reasons (Townsley 1998). This issue is of importance in Raja Ampat because the communities in Raja Ampat are a mix of ethnic Papuans and migrants from western parts of Indonesia. Also because considerable efforts are underway (for example the Papuan Congress 2002) to integrate the population in Raja Ampat within the traditional adat (system of law), and revive the traditional marine tenure in the region in collaboration with the fisheries management department in Raja Ampat.

A similar and related issue of interest is the influence of local leaders among the fisher community. It has been found that identification of centers of decision making in the community especially in cohesive communities can aid management effort (Townsley

1998). During my field visit to Kofiau in Raja Ampat, I saw that the declaration of the MPA began in the church courtyard with a sermon from the priest. The entire village was at the meeting, and there was a mood of celebration; the ceremony indicated the value of religious leadership in promoting management. In Raja Ampat, a study conducted in Misool Island found that in villages where the traditional management systems had been adopted and promoted by the church; there was better cooperation on management issues. In another village where similar integration had not happened, the influence of traditional management systems continued to decline (McLeod et al. 2009). Exploring these questions could offer valuable insight into management tactics that could be successful in the region.

Chapter 3

In chapter 3, I used spatial ecosystem models to explore the ecological benefits of implementing marine protected areas (MPAs) for coral reef ecosystems in Raja Ampat. The results showed that rapid rebuilding of reef fish populations required no-take areas. The model results also predicted trophic cascades inside the MPAs. When some forms of fishing were allowed inside the MPA, rebuilding was a slower process. It has been suggested that higher fishing effort in the spillover region encourages spillover (Walters et al. 2009), but high spillover could drain the MPA of the rebuilding fish biomass (Watson et al. 2000). A distinct tradeoff was observed between allowing some fisheries to operate inside the MPAs and expecting spillover effects from the MPAs.

After ascertaining that some no-take area was essential, the next goal was to analyze the size of no-take areas. From an ecological perspective, large no-take areas would be favoured for rebuilding to higher levels of precautionary biomass, for encompassing wide range of habitats (Lauck et al. 1998), for rebuilding of species with high dispersal rates (Walters 2000; Hilborn et al. 2004), and for protection of biodiversity and greater resilience against natural perturbations. Smaller no-take areas would allow design of a network of relatively closely spaced protection zones; this would enhance connectivity especially of larvae (Roberts et al. 2003; Hamilton et al. 2010). The disadvantage of very small MPAs is that the benefits of rebuilding could be lost through spillover, resulting in no effective protection (Walters 2000; Roberts et al. 2003).

I proposed an 'ideal minimum size' of no-take area: the size of MPA at which the increase in biomass density of reef fish reached an asymptote; it was thus the minimum size needed to ensure full rebuilding of reef fish. The results from the Ecospace models in Raja Ampat showed that for no-take areas of 16 to 25 km^2 in size or larger, the biomass density of reef fish asymptoted. The results showed that after reaching a certain size of 'no-take', increase in size of 'no-take' did not increase the biomass density of reef fish inside the MPA. Therefore, several zoning options were possible that offered flexibility in the consideration of non-ecological aspects in placement and design of MPAs. The estimates of 'ideal minimum size' (16 to 25 km^2) have to be considered in at least two perspectives: (1) the estimates were calculated based on response of reef fish to protection—species that disperse further than reef fish might need larger size of no-take areas, and (2) the exploitation status of the ecosystem. Raja Ampat reef ecosystems have declined considerably from pristine conditions; however, on a global scale of reef degradation (Pandolfi et al. 2003), Raja Ampat reefs were relatively less exploited. With reference to the 'ideal minimum size' of no-take area, an interesting question for future research is how the estimates change with change in dispersal rates and exploitation status of the ecosystems. A total of more than 100 EwE models have been built—at least one EwE model exists for almost all large marine ecosystems (LMEs). Database driven EwE models have been built for all LMEs (Christensen et al. 2009). Further work using Ecospace models from representative ecosystems from the different oceans would lead to improved findings on the 'ideal minimum size' for a no-take.

Larval dispersal studies suggested optimal size of MPAs to be in the range of 4 to 6 km diameter spaced at distances of about 10 to 20 km (Shanks et al. 2003). Size of no-take was correlated with population growth rate of species—slower growing populations required larger no-take areas (Mangel 1998). Roughly 33% of the Great Barrier Reef Marine Park was set as no-take areas with each no-take area being at least 10 to 20 km across (Fernandes et al. 2005). Targets of 10 to 50% closure (Lauck et al. 1998; Dahlgren and Sobel 2000; Botsford 2001; Roberts et al. 2003; Parnell et al. 2006; Stewart et al.

2007) offer broad guidelines for policy; in comparison, the 'ideal minimum size' offers a more specific local management guideline for zoning inside reserves.

Chapter 4

The analysis on size of 'no-take' areas showed that biomass density approached asymptote levels in MPAs. The results indicated that the populations inside the MPAs were probably approaching carrying capacity levels. In order to understand the concept of carrying capacity of a species in a system, in chapter 4, I reconstructed the ancient (c.1400 AD) population biomass of snapper in western North Island of New Zealand. The results showed the ancient snapper population in Maori times was about 2 to 4 times higher than the modern population biomass. The unfished population biomass was estimated to be about 5 to 20 times higher than the modern population. The results were sensitive to my assumptions about natural mortality on the ancient snapper population—I showed the estimates of ancient population biomass at different levels of natural mortality. In terms of carrying capacity, the results indicated that the carrying capacity was capped at it upper end by the natural mortality on the population in the system. Thus carrying capacity of an ecosystem for any species is not an invariable physical property, but rather it varies with the state of the ecosystem. Studies of historical and archaeological evidence (Jackson et al. 2001; MacKenzie et al. 2002; Lotze and Milewski 2004; Roberts 2007; Rose 2007), surveys with participation from older fishers (Sáenz-Arroyo et al. 2005; Ainsworth et al. 2008; Lozano-Montes et al. 2008), and comparisons of systems experiencing different levels of exploitation (Bellwood et al. 2004; Sandin et al. 2008) highlighted the difference between the current and less exploited ecosystem states. The current status of ecosystems needs to be seen in perspective with the historical abundances and diversity (Holm 2002). In the different states of the same ecosystem, the carrying capacity of a species would be different. The work in Chapter 4 also provided a methodology for archaeologists to estimate ancient population biomasses.

The estimates of ancient snapper population biomass were dependent on the estimates of total mortality in the archaeological period. If biomasses in ancient ecosystems are used as goals for rebuilding, then these estimates need to be considered in light of the total mortality on the species in the past. For example, on the snapper population in New Zealand, if the fishing pressure is released, the trajectory of rebuilding will be determined by the abundance of the predators in the current system. Historical baselines should be used as references to guide restoration effort in principle—historical baseline will help to explore, for example, that the predators of the concerned species are less abundant, and the population will rebuild to higher levels than the historical baseline, or that all the spawning grounds of the species have been destroyed, and achieving historical levels of abundance for the species is not a realistic expectation. Actual targets of rebuilding must be based on evaluation of carrying capacity of the current ecosystem.

On a scale of exploitation status, at one extreme is a pristine coral reef ecosystem with high abundance of sharks, groupers, other predatory fish, and luxuriant corals (Sandin et al. 2008). At the other extreme are ecosystems dominated by microbes (Jackson 2008). Though the systems have different levels of ecosystem maturity, both the ecosystems are at their carrying capacity irrespective of whether the ecosystem constituents are desirable or not; in the pristine reef, sharks are at their highest carrying capacity; in the other, the microbes are at their highest carrying capacity. In either case the ecosystem state is dependent mainly on the level of nutrients, the solar energy, and the inhabitants of the system in the previous time step. Following the decline of several exploited fish species (cod, haddock, and hake) in the North Atlantic, the abundance of shrimp in the system increased (Worm and Myers 2003; Frank et al. 2005). Historical (pre 1980) baseline for shrimp biomass would suggest a lower shrimp carrying capacity than the shrimp biomass exploited today. Similarly in ecosystems where jellyfish were not abundant in 1950s and 1960s, after the collapse of small pelagic species, jellyfish have increased to nuisance levels (Mills 2001; Lynam et al. 2006). This indicates that the abundance of its predators is one of the important factors that affect the carrying capacity of jellyfish. This is also one good reason to build trophic models and to explore species interactions in a system.

For the same reasons, carrying capacity needs to be considered in the light of "fishing down the food web". To some extent fishing down the food web was inevitable. Until very recently it was believed that the seas were inexhaustible (Smith 1994). Based in this belief, it was not difficult to overexploit any accessible marine mammal or fish population. Consequent to decrease in population size of the exploited predator species, the species in the next trophic level increased. This led to a shift in the exploitation pattern and repetition of the same pattern at the next trophic level. Therefore, historical baselines from different points in time will be different. If we are able to track the history of fishing down the food web, then we might be able to see the transformation in natural mortality levels on successive trophic levels. Analysis of natural mortality at different levels of species abundance might lead to an improved understanding of levels to which species will rebuild from their abundances in the current ecosystems.

Chapter 5

As discussed in the introductory chapter, fisheries played a large role in the decline of fish populations. Chapter 2 showed how the socio-political, economic, and technological changes influence fisheries resource use. Fisheries play a central role in marine restoration, but fishers are not the only stakeholders in the issue of fish population restoration. In Chapter 5, I found that there were advantages to restoration in combining the utilities of fishers, the tourism industry, and the conservationists. The main reason was that the two other stakeholders associated with ecological aspects of marine restoration —the tourism industry and the conservationists— were non-extractive in nature; though, several researchers have placed limits on tourism growth for ecological (Dixon et al. 1993; Schleyer and Tomalin 2000; Hawkins et al. 2002) and social equity reasons (Dixon et al. 1993; Harriott 2002). The scenarios—restricting net fisheries and 25% fisheries closure—were robust against uncertainty from ecosystem states, projections for tourism, specification of conservation benefits, and complex utility functions. I found that combination of fisher utilities with the utilities of the other stakeholders showed that gains in one industry (tourism) could, theoretically, encourage conservation by offsetting the losses of the other industry (fisheries). In Raja Ampat, tourism is a developing industry; when future benefits were discounted, the revenues from tourism were less effective in offsetting the losses of the fishing industry. At high discount rates (10% and 24%), I found that scenarios which favoured high fishing revenues provided the highest utility. High levels of protection of the ecosystems were favoured only when the conservation benefits were modeled according to 'ecosystem services' (e.g. including the function of coral reefs in shore protection).

In this chapter, I modeled the fisher utility considering only the fisheries revenue. The decisions of fishers about the fishing method they adopt or the area they fish in are not dependent on fisheries revenue alone. Even when fishers are convinced about the need for conservation, whether they would adopt the fisheries restrictions would depend on several factors such as: presence/provision for alternative livelihoods, education and skill to adopt new opportunities, and risk involved due to change in a fishing patterns (Townsley 1998). I have made several assumptions about the projections for tourism growth and tourism revenues from ecosystems in different states of exploitation; thus, my results are not directly applicable to choosing the management solution. My findings in the chapter highlighted the uncertainties associated with making decisions about restoration scenarios, and provided a basic framework which with better information on the different sectors could be adopted in decision making on fisheries restoration

Chapter 6

The analysis on ecological changes in MPAs using Ecospace model in Chapter 3 showed that trophic cascades could cause depression of medium trophic level species. Snapper reconstruction in Chapter 4 highlighted that carrying capacity was related to the natural mortality on the population. To investigate how different species would respond to restoration efforts, Chapter 6 investigated the response of species with different life histories against change in mortality. The analysis showed that fast growing species were more resilient to change in fishing mortality than slow growing species. Slow growing species—if they became vulnerable to fishing at age 1— even high levels of recruitment compensation did not increase their tolerance to very high levels of mortality. This could probably explain why slow growing species declined faster than fast growing species. The analysis showed that slow growing species would show a greater magnitude of recovery under lowered fishing pressure. The findings agreed with empirical evidence; large fished species showed strongest response to protection based on meta-analysis of fish densities inside and outside MPAs (Molloy et al. 2009). Large slow growing species, however, might not be the first group to respond to protection because several studies suggested that recovery rate was lower for slow growing species. In conclusion, fast growing species would show a quicker but smaller response to protection while slow growing species would show a slower but larger response to protection.

Coastal ecosystems are threatened by habitat degradation, shorefront infrastructure, fisheries, and decline of water quality (Olsen and Christie 2000). Some fish species might have the biology that allows for very high compensation, but allowing very high compensation ratios in stock assessments places high confidence in the biology of the population. Other ecological factors or anthropogenic factors (for example, abundance of spawners, upwelling, temperature, wind patterns, and degradation of stream habitat) could negatively affect the recruitment in the years when high recruitment compensation is needed to sustain the population. From the perspective of precautionary approach, it is dangerous to believe in very high levels of compensation.

Chapter 6 showed the magnitude to which recovery could be expected for different species and these results can be used to set restoration goals. The work could be used to develop management reference points specific to the growth parameters of the population, especially in data limited ecosystems or for data limited species.

Chapter 6

My thesis explored some theoretical aspects related to restoration: carrying capacity and influence of life history in species response to change in mortality. From an application perspective, the thesis re-emphasized the influence of socio-political and other extraneous factors on IUU fishing in Raja Ampat. The thesis developed a framework, using BBN, within which the needs of multiple stakeholders could be considered. The thesis also developed a measure referred to as the ‘ideal minimum size’ of a marine protected area which, after careful analysis across different marine ecosystems, could be a basis for global standards on size of individual no-take areas.

7.1 References

Ainsworth C. H., T. J. Pitcher, and C. Rotinsulu. 2008. Evidence of fishery depletions and shifting cognitive baselines in Eastern Indonesia. Biological Conservation **141**(3):848-859.

Alcala A. C., G. R. Russ. 2006. No-take marine reserves and reef fisheries management in the Philippines: a new people power revolution. Ambio: A Journal of the Human Environment **35**(5):245-254.

Alder J., N. A. Sloan, and H. Uktolseya. 1994. A comparison of management planning and implementation in three Indonesian marine protected areas. Ocean and Coastal Management **24**(3):179-198.

Bellwood D. R., T. P. Hughes, C. Folke, and M. Nyström. 2004. Confronting the coral reef crisis. Nature **429**(6994):827-833.

Botsford H. 2001. Dependence of sustainability on the configuration of marine reserves and larval dispersal distance. Ecology Letters **4**(2):144-150.

Buchary E. A., T. J. Pitcher, and U. R. Sumaila. (in press). Unaccounted values: under-reporting sardine catches as a strategy against poverty in the Bali Strait, Indonesia. In: Ommer, R. E., I. Perry, P. Cury, and K. Cochrane, editors. World Fisheries: A social-ecological analysis. Wiley-Blackwell Publication, Victoria, Canada.

Christensen V., C. J. Walters, R. Ahrens, J. Alder, J. Buszowski, L. B. Christensen, W. W. L. Cheung, J. Dunne, R. Froese, and V. Karpouzi. 2009. Database-driven models of the world's Large Marine Ecosystems. Ecological Modelling **220**(17):1984-1996.

Crawford B. R., A. Siahainenia, C. Rotinsulu, and A. Sukmara. 2004. Compliance and enforcement of community-based coastal resource management regulations in North Sulawesi, Indonesia. Coastal Management **32**(1):39-50.

Dahlgren C. P., J. Sobel. 2000. Designing a Dry Tortugas Ecological Reserve: How big is big enough? To do what? Bulletin of Marine Science **66**(3):707-719.

Dixon J. A., L. F. Scura, and T. van'tHof. 1993. Meeting Ecological and Economic Goals: Marine Parks in the Caribbean. Ambio: A Journal of the Human Environment **22**(2-3):117-125.

Elliott G., B. Mitchell, B. Wiltshire, I. A. Manan, and S. Wismer. 2001. Community participation in marine protected area management: Wakatobi National Park, Sulawesi, Indonesia. Coastal Management 29(4):295-316.

Erdmann M. V., P. R. Merrill, and M. Mongdong. 2004. Building effective co-management systems for decentralized protected areas management in Indonesia:

Bunaken National Park case study. Natural Resources Management Program. NRM III Program Secretariat. Jakarta. 149p.

Fernandes L., J. Day, A. Lewis, S. Slegers, B. Kerrigan, D. Breen, D. Cameron, B. Jago, J. Hall, and D. Lowe. 2005. Establishing representative no-take areas in the Great Barrier Reef: large-scale implementation of theory on marine protected areas. Conservation Biology **19**(6):1733-1744.

Frank K. T., B. Petrie, J. S. Choi, and W. C. Leggett. 2005. Trophic cascades in a formerly cod-dominated ecosystem. Science **308**(5728):1621-1623.

Gallopín G. C., S. Funtowicz, M. O'Connor, and J. Ravetz. 2001. Science for the twenty-first century: from social contract to the scientific core. International Social Science Journal **53**(168):219-229.

Halim A., P.J. Mous. 2006. Community perceptions of marine protected area management in Indonesia. Bali, Indonesia: The Nature Conservancy - Coral Triangle Center. Available: < www.coraltrianglecenter.org/downloads/MPA_Perc_Monitoring.pdf> [Accessed on February 12, 2007]

Hamilton S. L., J. E. Caselle, D. P. Malone, and M. H. Carr. 2010. Incorporating biogeography into evaluations of the Channel Islands marine reserve network. Proceedings of the National Academy of Sciences **107**(43):18272-18277.

Harriott V. J. 2002. Marine tourism impacts and their management on the Great Barrier Reef. CRC Reef Research Centre Technical Report **46**:1-41.

Hawkins J. P., C. M. Roberts, T. V. T. Hof, K. De Meyer, J. Tratalos, and C. Aldam. 2002. Effects of recreational scuba diving on Caribbean coral and fish communities. Conservation Biology **13**(4):888-897.

Hilborn R., K. Stokes, J. Maguire, T. Smith, L. W. Botsford, M. Mangel, J. Orensanz, A. Parma, J. Rice, J. Bell, K. L. Cochrane, S. Garcia, S. J. Hall, G. P. Kirkwood, K. Sainsbury, G. Stefansson, and C. Walters. 2004. When can marine reserves improve fisheries management? Ocean and Coastal Management **47**(3-4):197-205.

Holm P. 2002. History of marine animal populations: a global research program of the Census of marine life. Oceanologica Acta **25**(5):207-211.

Jackson J. B. C. 2008. Ecological extinction and evolution in the brave new ocean. Proceedings of the National Academy of Sciences 105(Supplement 1):11458-11465.

Jackson J. B. C., M. X. Kirby, W. H. Berger, K. A. Bjorndal, L. W. Botsford, B. J. Bourque, R. H. Bradbury, R. Cooke, Jon Erlandson, J. A. Estes, T. P. Hughes, S. Kidwell, C. B. Lange, H. S. Lenihan, J. M. Pandolfi, C. H. Peterson, R. S. Steneck, M. J.

Tegner, and R. R. Warner. 2001. Historical overfishing and the recent collapse of coastal ecosystems. Science **293**(5530):629-638.

Lauck T., C. W. Clark, M. Mangel, and G. R. Munro. 1998. Implementing the precautionary principle in fisheries management through marine reserves. Ecological Applications **8**(1) Supplement:S72-S78.

Lotze H. K., I. Milewski. 2004. Two centuries of multiple human impacts and successive changes in a North Atlantic food web. Ecological Applications **14**(5):1428-1447.

Lozano-Montes H. M., T. J. Pitcher, and N. Haggan. 2008. Shifting environmental and cognitive baselines in the upper Gulf of California. Frontiers in Ecology and the Environment **6**(2):75-80.

Lynam C. P., M. J. Gibbons, B. E. Axelsen, C. A. J. Sparks, J. Coetzee, B. G. Heywood, and A. S. Brierley. 2006. Jellyfish overtake fish in a heavily fished ecosystem. Current Biology **16**(13):492.

MacKenzie B. R., J. Alheit, D. J. Conley, P. Holm, and C. C. Kinze. 2002. Ecological hypotheses for a historical reconstruction of upper trophic level biomass in the Baltic Sea and Skagerrak. Canadian Journal of Fisheries and Aquatic Sciences **59**(1):173-190.

Mangel M. 1998. No-take areas for sustainability of harvested species and a conservation invariant for marine reserves. Ecology Letters **1**(2):87-90.

McLeod E., B. Szuster, and R. Salm. 2009. Sasi and marine conservation in Raja Ampat, Indonesia. Coastal Management **37**(6):656-676.

Mills C. E. 2001. Jellyfish blooms: are populations increasing globally in response to changing ocean conditions? Hydrobiologia **451**(1):55-68.

Molloy P. P., I. B. McLean, and I. M. Côté. 2009. Effects of marine reserve age on fish populations: a global meta-analysis. Journal of Applied Ecology **46**(4):743-751.

Olsen S., P. Christie. 2000. What are we learning from tropical coastal management experiences? Coastal Management **28**(1):5-18.

Olsson P., C. Folke, and T. P. Hughes. 2008. Navigating the transition to ecosystem-based management of the Great Barrier Reef, Australia. Proceedings of the National Academy of Sciences **105**(28):9489-9494.

Pandolfi J. M., R. H. Bradbury, E. Sala, T. P. Hughes, K. A. Bjorndal, R. G. Cooke, D. McArdle, L. McClenachan, M. J. H. Newman, G. Paredes, R. R. Warner, and J. B. C. Jackson. 2003. Global Trajectories of the Long-Term Decline of Coral Reef Ecosystems. Science **301**:955-958.

Parnell P. E., P. K. Dayton, C. E. Lennert-Cody, L. L. Rasmussen, and J. J. Leichter. 2006. Marine reserve design: optimal size, habitats, species affinities, diversity, and ocean microclimate. Ecological Applications **16**(3):945-962.

Pitcher T. J., D. Kalikoski, K. Short, D. Varkey, and G. Pramod. 2009. An evaluation of progress in implementing ecosystem-based management of fisheries in 33 countries. Marine Policy **33**(2):223-232.

Pollnac R. B., B. R. Crawford, and M. L. G. Gorospe. 2001. Discovering factors that influence the success of community-based marine protected areas in the Visayas, Philippines. Ocean and Coastal Management **44**(11-12):683-710.

Roberts C. M. 2007. The unnatural history of the sea. Island Press, Washington DC. 435p.

Roberts C. M., G. Branch, R. H. Bustamante, J. C. Castilla, J. Dugan, B. S. Halpern, K. D. Lafferty, H. Leslie, J. Lubchenco, and D. McArdle. 2003. Application of ecological criteria in selecting marine reserves and developing reserve networks. Ecological Applications **13**(1) Supplement:S215-S228.

Rose G. A. 2007. Cod: the ecological history of the North Atlantic fisheries. Breakwater Books, Newfoundland, Canada. 592p.

Sáenz-Arroyo A., C. M. Roberts, J. Torre, M. Cariño-Olvera, and R. R. Enríquez-Andrade. 2005. Rapidly shifting environmental baselines among fishers of the Gulf of California. Proceedings of the Royal Society of London. Series B: Biological Sciences **272**(1575):1957-1962.

Sandin S. A., J. E. Smith, E. E. DeMartini, E. A. Dinsdale, S. D. Donner, A. M. Friedlander, T. Konotchick, M. Malay, J. E. Maragos, and D. Obura. 2008. Baselines and degradation of coral reefs in the northern Line Islands. PLoS One **3**(2):e1548.

Schleyer M. H., B. J. Tomalin. 2000. Damage on South African coral reefs and an assessment of their sustainable diving capacity using a fisheries approach. Bulletin of Marine Science **67**(3):1025-1042.

Shanks A. L., B. A. Grantham, and M. H. Carr. 2003. Propagule dispersal distance and the size and spacing of marine reserves. Ecological Applications **13**(1) Supplement:S159-S169.

Smith T. D. 1994. Scaling fisheries: the science of measuring the effects of fishing, 1855-1955. Cambridge University Press, Cambridge. 392p.

Stewart R. R., I. Ball, and H. P. Possingham. 2007. The effect of incremental reserve design and changing reservation goals on the long-term efficiency of reserve systems. Conservation Biology **21**(2):346-354.

Sumaila U. R., J. Alder, and H. Keith. 2006. Global scope and economics of illegal fishing. Marine Policy **30**(6):696-703.

Townsley P. 1998. Social issues in fisheries. FAO Fisheries Technical Paper T375. Rome. 93p.

Walters C. 2000. Impacts of dispersal, ecological interactions, and fishing effort dynamics on efficacy of marine protected areas: how large should protected areas be? Bulletin of Marine Science **66**(3):745-757.

Walters C., R. Hilborn, and C. Costello. 2009. Comparison of marine protected area policies using a multispecies, multigear equilibrium optimization model (EDOM). Fisheries Centre Working Paper **4**:1-44.

Watson R., J. Alder, and C. Walters. 2000. A dynamic mass-balance model for marine protected areas. Fish and Fisheries **1**(1):94-98.

Worm B., R. A. Myers. 2003. Meta-analysis of cod-shrimp interactions reveals top-down control in oceanic food webs. Ecology **84**(1):162-173.

WWF/IUCN. 1996. Indonesian country study on integrated coastal and marine biodiversity management. Ministry of State for environment, Republic of Indonesia in cooperation with Directorate for Nature Management Kingdom of Norway. Appendix I Marine Conservation Areas **A**:121-125.

Appendix A Ecosystem Based Management: the Influence of a Project in Raja Ampat, Papua, Indonesia[36]

Abstract

The Birds Head Seascape Ecosystem Based Management (BHS EBM) project is a joint Packard-funded initiative between TNC, CI, WWF and UBC. The first two years of the project was based in Raja Ampat Regency in Indonesia, a region of incredible marine biodiversity. The project came into existence with the intentions of the partner NGOs and the Regency government to develop environmentally sound policies for the management of the marine resources. This paper evaluates the expected progress from the successful implementation of the project. The evaluation is based on previously-published criteria in implementing ecosystem-based fishery management (EBFM): overall principles (5 attributes); criteria for success (6 attributes); and implementation steps (12 attributes). The results show that a considerable improvement in management might be expected with the successful implementation of the BHS EBM project.

Introduction

There is now substantial interest in establishing ecosystem-based frameworks for fisheries management; in fact legislative requirements in some countries are beginning to demand the inclusion of principles of ecosystem based management (Hall and Mainprize

[36] A version of this analysis has been published. Varkey, D. A., C. H. Ainsworth, and T. J. Pitcher. 2008. Ecosystem based management: the influence of a project in Raja Ampat, Papua, Indonesia. Pages 169–175 in Bailey, M. and Pitcher, T. J (editors) (2007) Ecological and Economic Analyses of Marine Ecosystems in the Birds Head Seascape, Papua, Indonesia: II. Fisheries Centre Research Reports 16(1): 186 pp.

2004). The interest and faith leads to the increase in the number of projects designed according to principles of ecosystem based management. Before embarking on the project, or during a mid-term evaluation or after the completion of the project, it will be interesting to evaluate the change towards ecosystem-based management caused due to the project. This paper evaluates the marine management scenario before and after the implementation of the Birds Head Seascape Ecosystem Based Management (BHS EBM) project.

Raja Ampat Regency in Eastern Indonesia is an interesting and appropriate site for a case study for two reasons: Three environmental NGO partners (Conservation International, The Nature Conservancy's Southeast Asia Center for Marine Protected Areas, and WWF-Indonesia) are involved in a science-based initiative in partnership with local stakeholders to explore processes that contribute to management (Conservation International 2005). The second reason is that Indonesia scored below fail grade in all the three categories of the analysis, thus it was assumed that no factors external to the project contributed to the changes observed during the period. We evaluated the status of EBM in the area prior to the inception of the project and the status expected after successful implementation against the same three sets of the listed attributes used in this paper.

Method

We have chosen to base the analysis on Ward et al. (2002) framework which consists of three sets of attributes for ecosystem based management: overall principles (5 attributes; Table 2, page 19 in Ward et al. (2002); criteria for success (6 attributes; Table 3, pages 19-20 in Ward et al. (2002)); and implementation steps (12 attributes; Table 6, pages 50-51 in Ward et al. (2002)). Fishery management in Raja Ampat before and after the implementation were scored against the three main sets of the listed attributes.

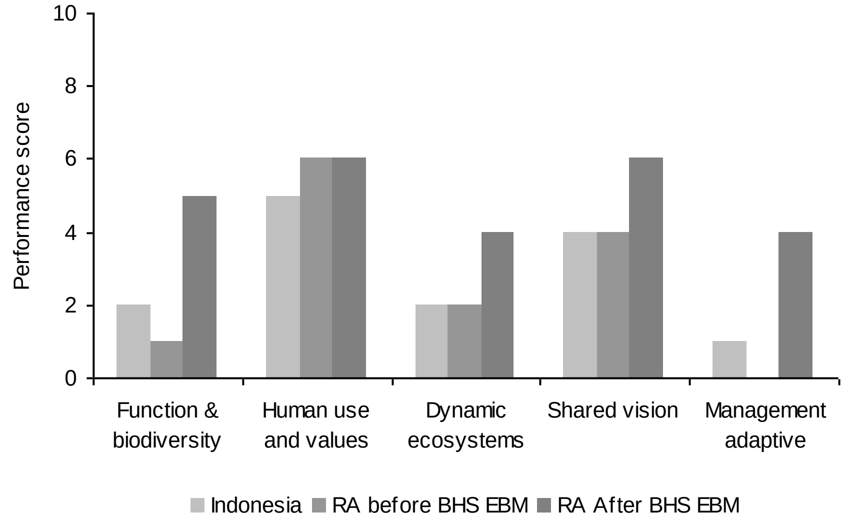
Results and Discussion

The scores for Indonesia were obtained from extensive material documenting Indonesia's compliance with the FAO (UN) Code of Conduct for Responsible fisheries (Pitcher et al. 2007; Pitcher et al. 2009). Scores, including the lower and upper bounds allocated to each attribute in Raja Ampat are shown in Table A1. Following the method outlined above, final ordination results are shown in Figures A1, A2 and A3.

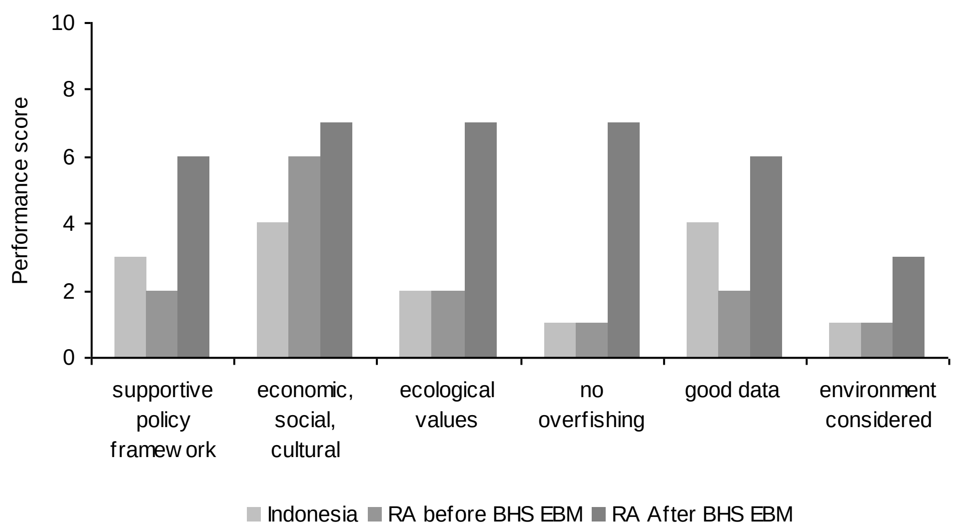
Appendix Table A-1 EBM Scores
Scores, lower, and upper bounds for Indonesia and Raja Ampat Regency before and after the implementation of the BHS EBM project. Scores for Indonesia are taken from Pitcher et al. (2006), and Pitcher et al. (2009).

Area	Indonesia			Raja Ampat			Raja Ampat		
	Score	min	max	Score	min	max	Score	min	max
Five Principles of EBM									
Function & biodiversity	2	0	2	1	1	3	5	4	7
Human use and values	5	4	7	6	5	8	6	5	8
Dynamic ecosystems	2	0	3	2	0	3	4	3	6
Shared vision	4	3	6	4	2	4	6	3	7
Management adaptive	1	0	2	0	0	1	4	2	6
Six indicators of EBM									
supportive policy framework	3	0	4	2	0	3	6	4	7
economic, social, cultural	4	3	6	6	4	7	7	4	8
ecological values	2	1	3	2	1	3	7	4	7
no overfishing	1	0	2	1	0	2	7	4	7
good data	4	3	5	2	0	2	6	3	7
environment considered	1	0	2	1	0	1	3	2	4
Twelve Steps Implementing									
stakeholders identified	2	1	6	2	1	4	7	5	8
Eco-regions map	2	1	6	1	1	2	8	8	10
stakeholders interests	4	3	7	3	1	3	6	5	8
ecosystem values	3	0	5	1	0	2	6	4	6
hazards	2	0	4	3	2	4	6	3	6
ecological risk assessment	2	0	2	1	0	2	7	6	9
goals agreed	3	0	3	2	0	3	5	3	7

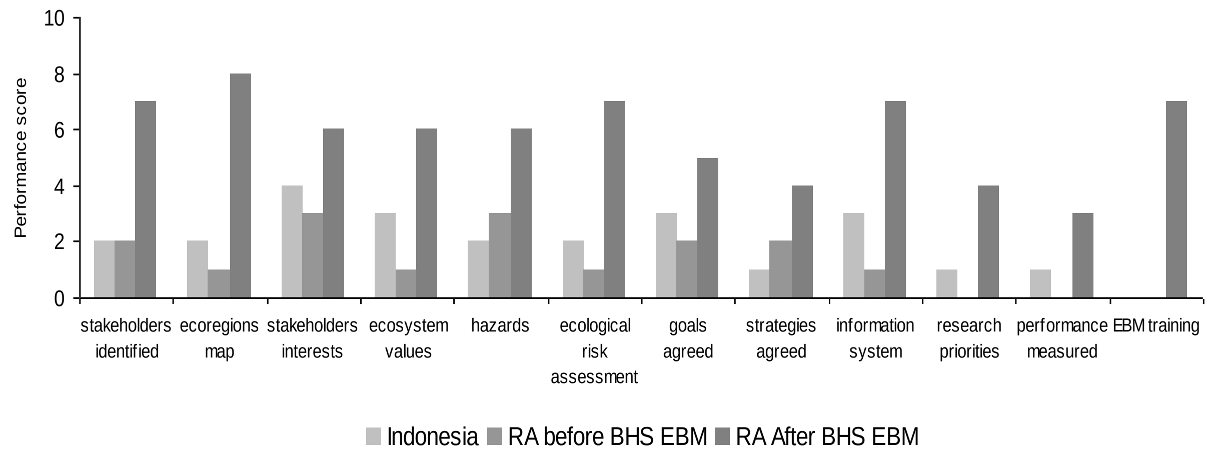
Area	Indonesia			Raja Ampat			Raja Ampat		
	Score	min	max	Score	min	max	Score	min	max
strategies agreed	1	0	1	2	1	2	4	3	6
information system	3	3	5	1	1	4	7	6	8
research priorities	1	0	2	0	0	3	4	3	7
performance measured	1	0	2	0	0	2	3	1	4
EBM training	0	0	1	0	0	0	7	5	8



Appendix Figure A-1 Scores for EBM principles
Scores for Indonesia in light grey, Scores for Raja Ampat before EBM project implementation in medium grey, and Scores for Raja Ampat after EBM project implementation in dark grey



Appendix Figure A-2 Scores for EBM indicators
Scores for Indonesia in light grey, Scores for Raja Ampat before EBM project implementation in medium grey, and Scores for Raja Ampat after EBM project implementation in dark grey



Appendix Figure A-3 Scores for EBM implementation
Scores for Indonesia in light grey, Scores for Raja Ampat before EBM project implementation in medium grey, and Scores for Raja Ampat after EBM project implementation in dark grey

Before the beginning of the project:

The ecosystem is in a better shape than in other parts of Indonesia, but no measures are in place to protect the system (McKenna et al. 2002). The people assume that the coral reefs will remain and support the population forever (Halim and Mous 2006). The role of habitat or a species in an ecosystem is not understood. Traditional rights were squashed during Suharto's regime. Management today exists as a conflict between the village head and the fisheries department, also there is no cooperation between different sectors (i.e. mining, fisheries, tourism etc). Local chiefs often receive payment and allow fishing in waters that traditionally belong to the village (Goram 2007). Recently there is recognition of damages from destructive fishing practices (Djuang, J. pers. comm.), but more fishers have adopted destructive fishing methods under the influence of fishermen from outside Raja Ampat. There is no assessment of the fish catches or the fish stocks, there is also a large amount of unreported catch, and hence it is impossible to ascertain the level of fishing for practice of adaptive management. No information system exists; however the government is planning an inventory of the fishing vessels in the area. The only maps that existed were the nautical charts made by the Dutch. A general idea exists about partners and stakeholders. Environment externalities are recognized but not a part of consideration in management. Human use values are recognized and the people connect deeply with the ocean. They also understand that fishermen from outside Raja Ampat engage in rampant use of destructive fishing methods as they have no respect for Raja Ampat waters (personal observation). The major management goal is to prevent entry of outside fishermen into Raja Ampat waters. Nobody understands or is involved in EBM.

Expected outcomes from the EBM project:

Many information lacunaes have been filled during the project. An aerial survey as conducted to find out the number of fisheries operations in the Regency (Barmawi 2006). A rapid appraisal was conducted of the demographics of the Regency for a deeper understanding of exploitation demand from the resource (Jacinta and Imbir 2007). Careful evaluation of the fishers and the other economic sectors has been done in the project (Dohar and Anggraeni 2007). The Atlas of Raja Ampat (Firman and Azhar 2006) is a clear inventory of habitats and eco-regions built during the project, future use of the information has been made easy by construction of GIS files.

The ecosystem model that built during the project integrated information from the different sources and quantified the interactions between the different ecosystem components, their habitat and the resource users. The model estimated the maximum sustainable yields of the important fish and invertebrate groups (Ainsworth et al. 2007). Study on anchovy fishery (Bailey et al. 2008) revealed unreported catches that were subsequently used to ascertain the actual fisheries extraction from the system (Varkey et al. 2010). Several scenarios were analyzed to study the direct and indirect effects of destructive fishing and overfishing. Risk assessment of fisheries was done using ecosystem model, the model can thus be used for adaptive management. Research questions were suggested by the participating NGOs were studied in detail (Ainsworth et al. 2008). Studies on the institutional roles and traditional marine tenure helps to identify people who wield power in fisheries management decisions. The CI, TNC, the Papuan council and the Navy are collaborating on a monitoring program for Raja Ampat (Rabu 2006). The findings will be communicated to the people via local newsletters like the CI tabloid (Rabu 2006), posters and booklets. Training manuals prepared by the University and the NGO teams will be used to give training and education. The information from the field surveys and the model will be used to design an EBM plan (Sumule and Boli 2006).

During the course of the project, residents in Kofiau stated that they had observed improvements in catch around their villages after following guidelines issued by the TNC. The NGOs conduct regular surveys for information update but the Regency lacks capacity for independent review. The NGOs plan and conduct review and performance assessment (Djuang pers. comm.) regularly. The project is making efforts to collaborate with the Local Papuan Council, a Council of local leaders on issues of marine management and design of policy framework. During the implementation of the project MPAs were declared to keep out fishermen from outside Raja Ampat. It is difficult to consider environment externalities for management even after the project has been implemented.

References

Ainsworth C. H., D. A. Varkey, and T. J. Pitcher. 2007. Ecosystem simulation models for the Bird's Head Seascape, Papua. In: Pitcher T.J., C.H. Ainsworth, and M. Bailey, editors, Ecological and economic analyses of marine ecosystems in the Birds Head Seascape, Papua, Indonesia. Fisheries Centre Research Reports **15**(5):6–172.

Ainsworth C. H., D. A. Varkey, and T. J. Pitcher. 2008. Ecosystem simulations supporting ecosystem-based fisheries management in the Coral Triangle, Indonesia. Ecological Modelling **214**(2-4):361-374.

Bailey M., C. Rotinsulu, and U. R. Sumaila. 2008. The migrant anchovy fishery in Kabui Bay, Raja Ampat, Indonesia: Catch, profitability, and income distribution. Marine Policy **32**(3):483-488.

Barmawi M. 2006. Aerial survey in Raja Ampat: An observation of fisheries activity, large marine fauna, and coastal habitat (January 9-13, 2006). Data Report. The Nature Conservancy Coral Triangle Center. Jl. Pengembak no. 2, Sanur, Denpasar. Available: joanne_wilson@tnc.org.

Conservation International. 2005. EBM factsheet. Ecosystem based management of the Bird's Head Seascape**.** A joint initiative of Conservation International, The Nature Conservancy, and WWF-Indonesia funded by the David and Lucille Packard Foundation. Jln. Arfak No. **45**, Sorong 98413, Indonesia.

Dohar A. G., D. Anggraeni. 2007. Valuasi Ekonomi Sumberdaya Alam di Kepulauan Raja Ampat. Conservation International, Bekerjasama dengan, Universitas Negeri Papua. (in Indonesian) Economic valuation report for Raja Ampat, Indonesia; Conservation International Indonesia.Available: Rotinsulu C. Gunung Arfak CI. Jl. 45. Sorong, Papua, Indonesia. chris@conservation.or.id.

Firman A., I. Azhar. 2006. Atlas sumberdaya wilayah pesisir Kabupaten Raja Ampat, Provinsi Irian Jaya Barat. Conservation International Indonesia, Sorong, Indonesia. Available: chris@conservation.or.id.

Goram Y. 2007. Interview (Sawen, Sawaki, J., Raubaba, S. and L. Metansan, 10-11 February, 2007).

Halim A., P.J. Mous. 2006. Community perceptions of marine protected area management in Indonesia. Bali, Indonesia: The Nature Conservancy - Coral Triangle Center. Available: < www.coraltrianglecenter.org/downloads/MPA_Perc_Monitoring.pdf> [Accessed on February 12, 2007]

Hall S. J., B. Mainprize. 2004. Towards ecosystem-based fisheries management. Fish and Fisheries **5**(1):1-20.

Jacinta D., C. Imbir. 2007. Analisis Tingkat Kesejehteraan Masyarakat Raja Ampat. Participatory Rural Appraisal of Raja Ampat community (in Indonesian). Draft Report. Indonesia: Conservation International Indonesia. Available: Rotinsulu C. CI. Jl.Gunung Arfak.45.Sorong, Papua, Indonesia. Available: chris@conservation.or.id.

McKenna S. A., P. Bolis, and G. R. Allen. 2002. Condition of coral reefs at the Raja Ampat Islands, Papua Province, Indonesia. In: McKenna, S.A., G.R. Allen, and S. Suryadi, editors. A marine rapid assessment of the Raja Ampat Islands, Papua Province, Indonesia. Conservation International, Washington, DC. RAP Bulletin of Biological Assessment **22**: 66-78.

Pitcher T. J., E. A. Buchary, and G. Pramod. 2007. An estimation of compliance of the fisheries of Indonesia with Article 7 (Fisheries Management) of the UN Code of Conduct for Responsible Fishing. Fisheries Centre Research Reports **14**(2):34.

Pitcher T. J., D. Kalikoski, K. Short, D. Varkey, and G. Pramod. 2009. An evaluation of progress in implementing ecosystem-based management of fisheries in 33 countries. Marine Policy **33**(2):223-232.

Rabu P. 2006. Tabloid Raja Ampat. Conservation International, Indonesia. **6**:1-4.

Sumule A., P. Boli. 2006. Collaborative Ecosystem-based management of Bird's Head seascape. Project in Progress, Conservation International, Raja Ampat, Indonesia.

Varkey D. A., C. H. Ainsworth, T. J. Pitcher, Y. Goram, and R. Sumaila. 2010. Illegal, unreported and unregulated fisheries catch in Raja Ampat Regency, Eastern Indonesia. Marine Policy **34**(2):228-236.

Ward T., D. Tarte, E. Hegerl, and K. Short. 2002. Policy proposals and operational guidance for ecosystem-based management of marine capture fisheries. World Wide Fund for Nature, Sydney, Australia. 80p.

Appendix B Ecopath Parametrization of Raja Ampat Model[37]

Ecopath

The idea of Ecopath was first given by Polovina (1984) and later adapted and developed by Christensen and Pauly (1992) and Christensen and Walters (2004). Ecopath functions under two master equations.The first equation explains that biological production within a functional group equals the sum of mortality caused by fisheries and predators, net migration, biomass accumulation and other unexplained mortality.

$$Production = Removals + Mortality + Biomass\ Accumulation + Migration\ .$$

This can be mathematically re-expressed as:

$$\boldsymbol{B_i * (P/B)_i = Y_i + \sum_{j=1}^{n} B_j * (Q/B)_j * DC_{ij} + E_i + BA_i + B_i * (P/B)_i * (1 - EE_i)}$$

where,

Subscript $\boldsymbol{i}$ and $\boldsymbol{j}$ represent prey and predator respectively;

B_i and B_j are biomasses of prey *(i)* and predator *(j)*, respectively;

P/B_i is the production/biomass ratio;

[37] This appendix is an excerpt from technical reports that describe in great detail the Ecopath and Ecosim models of the Raja Ampat coral reef ecosystem: Ainsworth et al. 2007 and Ainsworth et al. 2008.

Y_i is the total fishery catch rate of group *(i);*

Q/B_j is the consumption/biomass ratio;

DC_{ij} is the fraction of prey *(i)* in the average diet of predator *(j);*

E_i is the net migration rate (emigration – immigration); and

BA_i is the biomass accumulation rate for group *(i).*

EE_i is the ecotrophic efficiency; the fraction of group mortality explained in the model

The second assumption is that consumption within a group equals the sum of production, respiration and unassimilated food, as in eq 2.

$$B \cdot (Q/B) = B \cdot (P/B) + (1 - GS) \cdot Q - (1 - TM) \cdot P + B(Q/B) \cdot GS \qquad \text{Equation 2}$$

Where GS is the proportion of food unassimilated; and TM is the trophic mode expressing the degree of heterotrophy; 0 and 1 represent autotrophs and heterotrophs, respectively. Intermediate values represent facultative consumers.

Ecopath uses a set of algorithms (Mackay 1981) to simultaneously solve n linear equations of the form in eq.1, where n is the number of functional groups. Under the assumption of mass-balance, Ecopath can estimate missing parameters. This allows modellers to select their inputs. Ecopath uses the constraint of mass-balance to infer qualities of unsure ecosystem components based on our knowledge of well-understood groups. It places piecemeal information on a framework that allows us to analyze the compatibility of data, and it offers heuristic value by providing scientists a forum to summarize what is known about the ecosystem and to identify gaps in knowledge.

Raja Ampat EwE model

The Raja Ampat EwE model describes the region from 129° 12' E and 0° 12' N to 131° 30' E and 2° 42' S. This large-scale model includes all the waters of Raja Ampat. The functional groups represent reef-associated fish identified in Raja Ampat (McKenna et al. 2002a) as well as pelagic and deepwater fish occurring in Eastern Indonesia. In order to be included in the model, a fish species had to be listed both under the 'Indonesia' country code in FishBase (FB country code 360) and the 'Papua New Guinea' code (FB country code 598). The field data used to develop Ecopath, Ecosim and Ecospace models came from the integrated and diverse BHS-EBM research project.

Functional group designations

Ninety-eight functional groups are used to represent the marine ecosystem of Raja Ampat. These include mammals, birds, reptiles, fish, invertebrates, plants, zooplankton, phytoplankton, and non-living groups such as fishery discards and organic detritus (Table B1). The models have been designed to serve at various spatial scales. Ideally, smaller area models, such as the one representing Kofiau Island, would have a group structure especially suited to represent coral reef organisms and their interactions, while the larger area Raja Ampat model should consider pelagic and deep-water species in more detail. However, to keep the various models comparable, identical group structures are used. A compromise solution is therefore used that tends to emphasize reef communities, while providing the basic level of functionality necessary to assist management of pelagic and deep-water resources.

High-order food web dynamics are carefully represented in the BHS EBM models in order to provide reliable forecasts concerning the impacts of fisheries on coral reefs. Important predatory, herbivorous and commercial fish tend to be allotted into highly specialized functional groups, while basal organisms are generally aggregated. At 98 functional groups these are complex models; but we believe that this approach is necessary in order to provide sufficient resolution to capture important processes occurring on coral reefs.

Fish groups

Because of the enormous amount of differentiation in life-history, morphology and feeding guilds that appears within coral reef fish families, delineating functional groups by fish family or clad is impractical and may be unwise. Through evolutionary convergence, similar niche specializations can be present in unrelated taxa; or, a single fish family may include multiple functional niches. The specific group structure in a EwE model is largely subjective and should be tailored to satisfy specific requirements of the investigation. Therefore, most of the functional groups developed for the preliminary Raja Ampat ecosystem models are based on the functional role that the fishes play in the ecosystem, with additional groups configured to allow the representation of important commercial, social and ecological interests. The important specializations were determined based on the ecological literature available for coral reef ecosystems (Bellwood et al. 2004) and through expert communication.

There are 1203 fish species represented in the Raja Ampat model. The fish species are apportioned into 57 functional groups; of which 30 represent unique species or species groups. The remaining functional groups correspond to various juvenile, subadult and adult life history stages included in the model to represent ontogenetic feeding, mortality and behaviour.

Fish functional groups may be designed to represent specific functional roles (e.g., grooming by cleaner wrasse, algae mediation by herbivorous echinoids), to represent species of commercial interest (e.g., skipjack tuna, groupers) or to cover the wide diversity of fishes in aggregated species groups (e.g., large reef-associated fish). Fish have been allocated into functional groups based also on body size (e.g., small, medium and large groups), feeding guild (e.g., planktivorous and piscivorous) and habitat (e.g., pelagic, demersal, reef-associated).

Basic parameterization

The data needs of Ecopath can be summarized as follows. Four data points are required for each functional group: biomass (in t•km^{-2}), the ratio of production over biomass (P/B; in yr^{-1}), the ratio of consumption over biomass (Q/B; in yr^{-1}), and ecotrophic efficiency (EE; unitless). Ecopath also provides an input field representing the ratio of production over consumption (P/Q; unitless), which users may alternatively use to infer either P/B or Q/B based on the other. Each functional group requires 3 out of 4 of these input parameters and the remaining parameter is estimated using the mass-balance relationship in eq. 1.

A biomass accumulation rate may be entered optionally; the default setting assumes a zero-rate instantaneous biomass change. These Ecopath data points are referred to collectively in this appendix as the basic parameters. For a more thorough description of Ecopath data needs and parameter definitions please refer to Christensen and Walters (2004).

Appendix Table B-1 Basic parameters of the 2005 Raja Ampat Ecopath model

Group name	Trophic level	Biomass (t/km²)	P/B (yr^{-1})	Q/B (yr^{-1})	EE	P/Q
Mysticetae	3.49	0.033	0.055	4.850	0.024	0.011
Piscivorous odontocetae	4.22	0.051	0.035	6.105	0.024	0.006
Deepdiving odontocetae	4.05	0.090	0.020	3.599	0.024	0.006
Dugongs	2.00	0.054	0.025	11.012	0.000	0.002
Birds	3.59	0.366	0.381	63.949	0.019	0.006
Reef associated turtles	3.27	0.004	0.143	3.500	0.545	0.041
Green turtles	2.20	0.008	0.053	3.500	0.830	0.015
Oceanic turtles	3.44	0.008	0.053	3.500	0.830	0.015
Crocodiles	3.98	0.001	0.408	6.500	0.465	0.063
Adult groupers	3.65	0.500	0.225	9.086	0.950	0.025

Group name	Trophic level	Biomass (t/km²)	P/B (yr^{-1})	Q/B (yr^{-1})	EE	P/Q
Subadult groupers	3.70	0.156	0.400	13.110	0.950	0.031
Juvenile groupers	3.70	0.043	1.200	26.675	0.950	0.045
Adult snappers	3.72	0.345	0.400	7.105	0.989	0.056
Subadult snappers	3.66	0.178	1.100	11.085	0.743	0.099
Juvenile snappers	3.85	0.128	1.447	21.377	0.764	0.068
Adult Napoleon wrasse	3.85	0.049	0.450	8.900	0.950	0.051
Subadult Napoleon wrasse	3.62	0.083	0.500	12.952	0.950	0.039
Juvenile Napoleon wrasse	3.4	0.016	1.200	29.815	0.625	0.040
Skipjack tuna	4.09	0.693	2.000	6.644	0.600	0.301
Other tuna	4.05	0.541	1.408	4.693	0.411	0.300
Mackerel	3.79	0.086	2.913	9.712	0.960	0.300
Billfish	4.44	0.825	0.956	3.187	0.242	0.300
Adult coral trout	3.88	0.033	0.350	3.303	0.916	0.106
Juvenile coral trout	3.85	0.007	0.550	7.103	0.950	0.077
Adult large sharks	4.15	0.061	0.700	3.600	0.950	0.194
Juvenile large sharks	3.86	0.039	0.900	6.058	0.727	0.149
Adult small sharks	4.28	0.041	1.120	4.000	0.614	0.280
Juvenile small sharks	4.11	0.106	1.800	6.072	0.181	0.296
Whale shark	3.82	0.003	0.068	0.228	0.024	0.300
Manta ray	3.74	0.003	0.600	2.000	0.024	0.300
Adult rays	3.31	0.177	0.600	2.416	0.591	0.248
Juvenile rays	3.42	0.031	1.200	5.923	0.563	0.203
Adult butterflyfish	2.97	0.243	1.004	6.720	0.933	0.149
Juvenile butterflyfish	2.77	0.093	2.000	11.163	0.808	0.179
Cleaner wrasse	3.30	0.009	3.779	13.097	0.938	0.289
Adult large pelagic	3.89	0.074	0.800	2.667	0.950	0.300
Juvenile large pelagic	3.64	0.044	1.079	4.544	0.950	0.237
Adult medium pelagic	3.62	0.030	1.000	5.000	0.974	0.200

Group name	Trophic level	Biomass (t/km²)	P/B (yr^{-1})	Q/B (yr^{-1})	EE	P/Q
Juvenile medium pelagic	3.45	0.045	1.500	7.860	0.916	0.191
Adult small pelagic	3.59	0.071	2.000	13.266	0.965	0.151
Juvenile small pelagic	2.63	0.108	3.980	25.284	0.892	0.157
Adult large reef associated	2.95	5.000	0.250	4.000	0.994	0.063
Juvenile large reef associated	3.06	5.174	0.600	5.816	0.950	0.103
Adult medium reef associated	3.08	2.853	0.800	5.000	0.800	0.160
Juvenile medium reef	2.38	2.356	1.400	8.114	0.950	0.173
Adult small reef associated	2.76	0.259	3.000	15.000	0.953	0.200
Juvenile small reef asociated	2.70	0.135	4.000	30.345	0.967	0.132
Adult large demersal	3.21	0.127	0.600	3.100	0.902	0.194
Juvenile large demersal	3.47	0.135	0.920	5.140	0.990	0.179
Adult small demersal	3.61	0.192	2.000	8.600	0.970	0.233
Juvenile small demersal	3.22	0.135	2.568	15.718	0.936	0.163
Adult large planktivore	3.39	1.000	1.500	4.500	0.869	0.333
Juvenile large planktivore	3.48	0.887	2.000	7.511	0.972	0.266
Adult small planktivore	3.23	0.414	2.000	6.000	0.932	0.333
Juvenile small planktivore	2.52	0.565	2.000	7.349	0.980	0.272
Adult anchovy	3.31	1.500	2.700	14.625	0.950	0.185
Juvenile anchovy	2.14	1.855	3.200	27.329	0.631	0.117
Adult deepwater fish	3.83	0.500	1.100	3.667	0.800	0.300
Juvenile deepwater fish	3.37	0.661	1.000	5.316	0.950	0.188
Adult macro algal browsing	2.47	0.250	1.339	13.760	0.800	0.097
Juvenile macro algal browsing	2.16	0.500	1.400	18.888	0.950	0.074
Adult eroding grazers	2.45	0.526	0.435	1.451	0.858	0.300
Juvenile eroding grazers	2.71	0.256	1.000	2.200	0.803	0.455
Adult scraping grazers	2.16	0.348	2.339	12.740	0.942	0.184
Juvenile scraping grazers	2.33	1.656	3.000	22.729	0.773	0.132
Detritivore fish	2.24	0.016	2.339	8.333	0.946	0.281

Group name	Trophic level	Biomass (t/km²)	P/B (yr^{-1})	Q/B (yr^{-1})	EE	P/Q
Azooxanthellate corals	2.50	0.600	1.440	3.600	0.960	0.40
Hermatypic scleractinian corals	1.30	0.875	2.160	3.600	0.990	0.600
Non reef building scleractinian	1.30	0.600	1.400	2.330	0.990	0.601
Soft corals	1.75	0.600	0.917	1.913	0.950	0.480
Calcareous algae	1.00	0.100	0.475	-	0.996	-
Anemonies	3.15	0.500	0.050	0.069	0.920	0.726
Penaeid shrimps	2.51	2.000	3.824	37.900	0.948	0.101
Shrimps and prawns	2.02	2.000	2.228	20.000	0.960	0.111
Squid	3.49	0.237	4.348	14.792	0.943	0.294
Octopus	3.40	1.000	2.327	13.240	0.902	0.176
Sea cucumbers	2.00	0.971	0.740	8.248	0.923	0.090
Lobsters	3.23	0.500	0.800	15.207	0.950	0.053
Large crabs	2.95	0.286	0.953	14.558	0.960	0.065
Small crabs	2.51	0.286	2.610	20.208	0.927	0.129
Crown of thorns	2.47	0.219	0.920	9.423	0.920	0.098
Giant triton	3.34	0.050	1.224	4.080	0.992	0.300
Herbivorous echinoids	2.00	0.722	0.541	9.423	0.847	0.057
Bivalves	2.20	9.189	2.514	5.617	0.905	0.448
Sessile filter feeders	2.32	4.580	1.480	5.268	0.964	0.281
Epifaunal detritivorous	2.00	1.400	1.178	18.250	0.998	0.065
Epifaunal carnivorous	2.92	5.600	2.640	10.521	0.992	0.251
Infaunal invertebrates	2.01	27.422	4.014	19.267	0.924	0.208
Jellyfish and hydroids	3.10	0.100	10.230	26.462	0.913	0.387
Carnivorous zooplankton	3.18	1.000	63.875	177.777	0.950	0.359
Large herbivorous zooplankton	2.00	0.560	31.000	256.773	0.948	0.121
Small herbivorous zooplankton	2.00	2.430	91.250	265.810	0.883	0.343
Phytoplankton	1.00	26.100	109.119	-	0.313	-
Macro algae	1.00	39.389	10.225	-	0.375	-

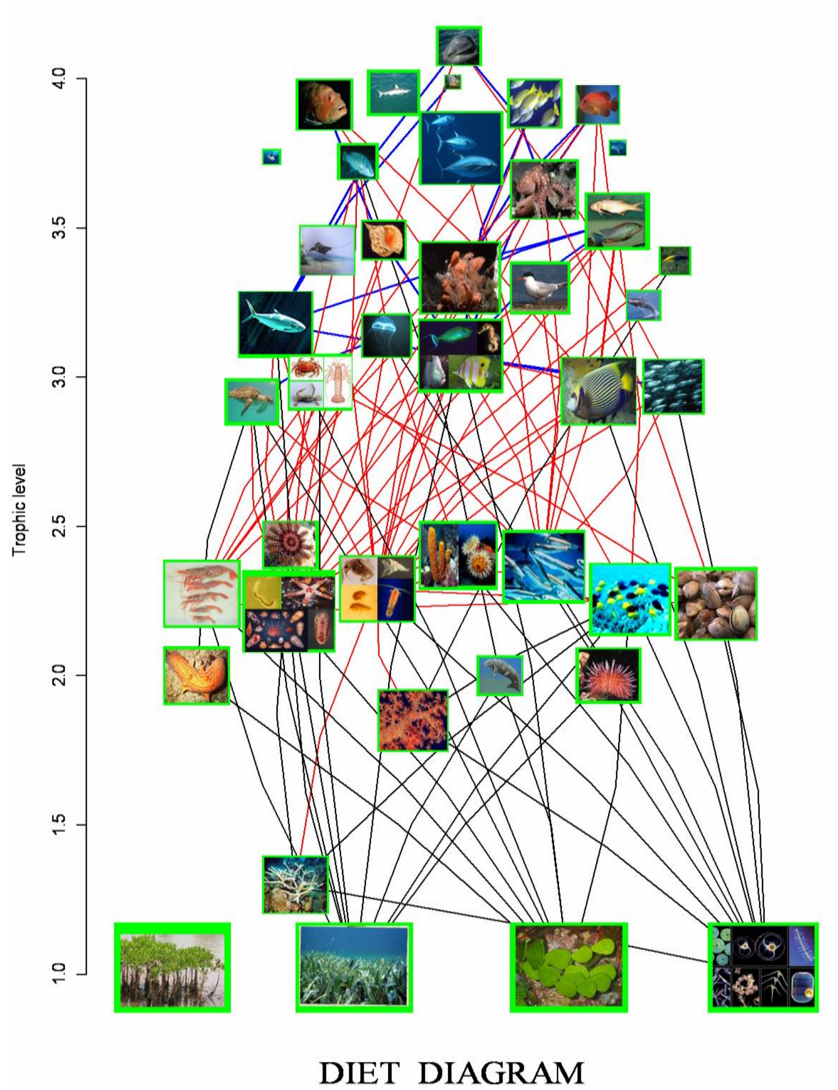
Group name	Trophic level	Biomass (t/km²)	P/B (yr^{-1})	Q/B (yr^{-1})	EE	P/Q
Sea grass	1.00	20.157	13.758	-	0.818	-
Mangroves	1.00	19.136	0.066	-	0.021	-
Fishery discards	1.00	20.000	-	-	0.916	-
Detritus	1.00	100.00	-	-	0.138	-

Summary of diet information 2005 Raja Ampat Ecopath

In order to produce a diet matrix for use in the Raja Ampat Ecopath with Ecosim (EwE) models, Ainsworth et al. (2007) developed a diet allocation algorithm to allocate prey fish composition into predator diets. The algorithm utilizes diet composition information stored in FishBase (FB) (Freose and Pauly 2010), an online data repository, and other literature sources. It considers habitat co-occupation and gape-size feeding restrictions, and predator and prey life stages. Using this method, Ainsworth et al. (2007) calculated a diet composition matrix at the resolution of species functional groups, and the diet matrix was subsequently adjusted for some species according to the results of a field stomach collection and gut content analysis (Ainsworth et al. 2008). We provide a summary of the diet estimation method here, and refer the reader to Ainsworth et al. (2007) for a more thorough description. Figure B1 shows a diagram of the major food web interactions in the model.

Diet allocation algorithm

Quantitative diet information was obtained from the FishBase (FB) (Freose and Pauly 2010) Diet table for 255 out of 1196 species in the Raja Ampat model. 26% of the reef fish and demersal fish species had available diet information, while 17% of the pelagic and deep water fish species had data. Of the 30 fish groups present in the model, 23 had information for at least one representative species. Categories of prey items listed in the FB Diet table are imprecise (e.g., ‘bony fish’, ‘benthic invertebrates’) and there are formatting and spelling variations. Some standardization was therefore required. FB prey items are sorted into their corresponding EwE functional groups, either in equal proportions for non-fish prey items, or in specific proportions for fish prey items calculated using a diet allocation algorithm.



Appendix Figure B-1 Trophic flows in the Raja Ampat marine ecosystem
Y-axis indicates functional group trophic level (TL); apex predators appear at the top, basal species are at the bottom. Boxes show model functional groups (simplified); box size is scaled logarithmically to represent relative group biomass. Lines show diet matrix connectances of 20% or greater. Coloured lines indicate predator TL (blue lines: TL 3+; red lines: TL 2-3; black lines: TL 1-2.

For each predator, the algorithm assigns appropriate EwE functional groups to each prey item listed in FB (Ainsworth et al. 2007). Where possible, prey functional groups were resolved into juvenile or adult life stages as determined from the data field 'SampleStage' in the FB Diet table. The algorithm then eliminates potential prey species from the predator's diet if they do not occur in the same habitat as the predator. Habitats were determined from the 'habitat' field of the FB Species table, and were resolved into the categories: reef-associated, demersal or pelagic.

A minimum and maximize prey size is then determined for each predator based on mouth gape size. These may be important parameters governing population dynamics (Claessen et al. 2002). Within this 'predation window', prey species are vulnerable. To determine the maximum gape size, family-specific gape-body length relationships (Karpouzi and Stergiou 2003) were utilized for seven fish families. For other families, maximum gape size was determined by assuming a similar gape-body length ratio as Labridae, in the case of mainly piscivorous predator families, or Mullidae, in the case of mainly planktivorous predator families. The window was shifted to allow larger prey items for predators that can tear or bite pieces off their prey such as the functional groups 'large sharks', 'small sharks', 'Manta ray' and 'rays' .

The smallest body dimension of the prey species, i.e., the dimension limiting consumption by a potential predator, is determined according to body morphology. For 'eel-like' or 'elongated' prey (FB categories), the smallest body dimension is assumed 12.5% of the maximum body length. For 'fusiform' fish or fish with no data, the smallest body dimension is assumed 25% of the maximum body length. For 'deep bodied' or 'flattened' fish, the smallest body dimension is assumed to equal 50%.

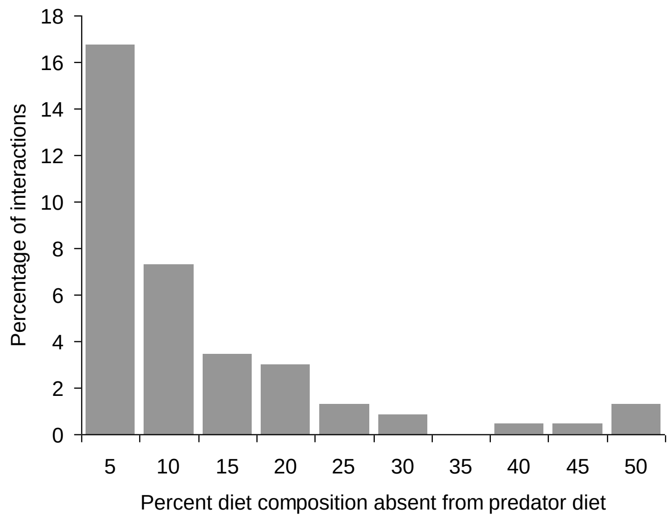
We assume that the predator-prey consumption rate follows a domed relationship that is dependant on the relative sizes of the species (Lundvall et al. 1999; Claessen et al. 2002).

The diet algorithm further assumes that the availability of prey species is affected by prey abundance, where consumption rate follows a step function relating to the categorical abundance of prey. See Ainsworth et al. (2007) for additional caveats and assumptions.

Gut content analysis

In November and December 2006 an analysis of fish gut contents was conducted in Raja Ampat by CI staff and two students from the State University of Papua (contact: Christovel Rotinsulu. CI. Jl.Gunung Arfak.45.Sorong, Papua, Indonesia. Email: chris@conservation.or.id). The protocol for obtaining samples, dissecting stomachs and analyzing the results is presented in Appendix C.2 of Ainsworth et al. (2007). Briefly, fish were purchased at markets and the stomachs removed, or else fishers were paid a fee in order to extract the stomachs. Stomachs were preserved in formalin and later dissected in the lab. The protocol was devised especially to support the current EwE models, so it was not important to identify prey species beyond the functional group level. Nevertheless, taxonomies were identified to a more precise level in order to make the data more valuable to future scientific studies. The diets of predator fish families were converted to percent composition values and scaled to total 100%.

The results of the gut content analysis related to 22 EwE functional groups. Once the predators and prey items were aggregated into EwE functional groups, 66% of feeding interactions identified by the stomach sampling program were successfully predicted by the diet allocation algorithm described above. The remaining 34% were mainly minor interactions (Figure B2). Of the predator-prey interactions that are absent from the diet allocation algorithm but identified by stomach sampling, only a small number (4.2 %) constitute major diet components (i.e., consisting of 25% or more of a predator's diet).



Appendix Figure B-2 Feeding interactions identified by stomach sampling that were not predicted by diet allocation algorithm.

Ainsworth et al. (2008) identified the critical interactions that were predicted to occur by the diet allocation algorithm but contradicted by the stomach sampling, and the interactions that were missed by the diet allocation algorithm but indicated by stomach sampling. The critical discrepancies, which were identified as the top 25 percentile of interactions based on rank importance, were modified in the EwE models to reflect the diet composition values calculated by the stomach content analysis. This process made the diet matrices more relevant and site-specific to Raja Ampat. All models (including 1990 and 2005 Raja Ampat models and sub-area models for Kofiau Is., Dampier St. and SE Misool Is.), use a similar diet matrix. Small changes were made to the matrices ad hoc during the process of tuning the models and establishing mass-balance.

References

Ainsworth C. H., D. A. Varkey, and T. J. Pitcher. 2008. Ecosystem simulation models of Raja Ampat, Indonesia, in support of ecosystem based fisheries management. In: Bailey M., T. J. Pitcher, editors. Ecological and Economic Analyses of Marine Ecosystems in the Birds Head Seascape, Papua, Indonesia: II. Fisheries Centre Research Reports **16**(1):3-122.

Ainsworth C. H., D. A. Varkey, and T. J. Pitcher. 2007. Ecosystem simulation models for the Bird's Head Seascape, Papua. In: Pitcher T.J., C.H. Ainsworth, and M. Bailey, editors, Ecological and economic analyses of marine ecosystems in the Birds Head Seascape, Papua, Indonesia. Fisheries Centre Research Reports **15**(5):6–172.

Bellwood D. R., T. P. Hughes, C. Folke, and M. Nyström. 2004. Confronting the coral reef crisis. Nature **429**(6994):827-833.

Christensen V. 1992. ECOPATH II--A software for balancing steady-state ecosystem models and calculating network characteristics. Ecological Modelling **61**(3):169-185.

Christensen V., C. J. Walters. 2004. Ecopath with Ecosim: methods, capabilities and limitations. Ecological Modelling **172**(2-4):109-139.

Claessen D., C. Van Oss, A. M. de Roos, and L. Persson. 2002. The impact of size-dependent predation on population dynamics and individual life history. Ecology **83**(6):1660-1675.

Freose R. and D. Pauly. 2010. FishBase. World Wide Web electronic publication. Available:www.fishbase.org version (05/2010).

Karpouzi V., K. Stergiou. 2003. The relationships between mouth size and shape and body length for 18 species of marine fishes and their trophic implications. Journal of Fish Biology **62**(6):1353-1365.

Lundvall D., R. Svanbäck, L. Persson, and P. Byström. 1999. Size-dependent predation in piscivores: interactions between predator foraging and prey avoidance abilities. Canadian Journal of Fisheries and Aquatic Sciences **56**(7):1285-1292.

Mackay A. 1981. The generalized inverse. Practical computing **9**:108-110.

Polovina J. J. 1984. Model of a Coral Reef Ecosystem I. The Ecopath Model and its Application to French Frigate Shoals. Coral reefs **3**(1):1-11.

Appendix C Estimation of IUU Fishing in Raja Ampat[38]

Influence table of IUU fishing in Raja Ampat

Appendix Table C-1 Influence table
Timeline of the Raja Ampat fishery constituted by 150 influences. The code represents type of influence (1-Policy, 2-technology, 3-Socio/Political, 4-Supply/Market changes)

Year	Event summary	Code	Reference
1960	Licensing of fishing boats started for large-scale commercial fisheries	1	(BRKP 2005a)
1963	Papua integrated into Indonesia and migration of people from other areas in Indonesia to Papua	3	(Chauvel 2005)
1963	Papuans introduced to use nets to catch fish	2	(Goram 2007)
1966	Market supply of salty fish to Java and Sulawesi	4	(Goram 2007)
1967	First long term development plan PJP I 1967/68 to 1993/94	3	(MOEROI-UNDP 1997)
1968	Bombing practices introduced to Papua	2	(Goram 2007)

[38] A version of this appendix is published. Pitcher T. 2010. Assessing illegal, unreported and unregulated fishery catches: magnitude and influences from case studies. Fisheries Centre Research Reports 18: in press.

Year	Event summary	Code	Reference
	by the Butonese		
1969	People from South Sulawesi and military officials arrive in large numbers	3	(Goram 2007)
1969	Two Tuna boats (Injeros and Cakalang) started by local government company (PD. Irian Bakti) for fish supply to residents in Sorong. Tuna catch started in limitation and under control by Fishery Department in Sorong.	2	(Goram 2007)
1970	Dominant purse seine fishery	2	(APFIC 2007)
1970	Usaha Mina (Fish Company owned by Suharto's Family) conducted its exploration in Raja Ampat and two companies owned by Japanese Government, WIF and IMPD, also conducted survey for shrimps and tuna.	2	(Goram 2007)
1973	WIF and IMPD were open and established in Sorong (Klademak I). Tuna and shrimps catches began in large numbers.	2	(Goram 2007)
1973	Usaha Mina was also established and officially inaugurated by the then president Suharto.	2	(Goram 2007)
1973	Fishermen from South Sulawesi start using bagans for anchovy fishery.	3	(Goram 2007)
1973	Act I - continental shelf of Indonesia	3	(MOEROI-UNDP 1997)
1974	Australia and Indonesia entered into a MoU which recognised the rights of	3	(DAFF 2007)

Year	Event summary	Code	Reference
	access for traditional Indonesian fishers in shared waters to the north of Australia.		
1974	Act 5 - on the devolution of central government authority to regional and local government	3	(MOEROI-UNDP 1997)
1975	Agricultural Ministerial Decree No 23/1975 restricted the use of purse seine with mesh size > 60 mm to catch pelagic	1	(BRKP 2005a)
1975	Usaha Mina was also established and officially inaugurated by the then president Suharto. 15 boats increased to 60 boats with the capacity of 50 tonnage for each boat.	2	(Goram 2007)
1976	First transmigration from Java to Sorong.	3	(Goram 2007)
1977	Shark fin became popular. Anchored shark bodies in coastal areas a common sight.	4	(Goram 2007)
1977	Permits from Fishery Department in Sorong were easily processed and issued	3	(Goram 2007)
1978	The catch of sea cucumber by South Sulawesi people began in large scale. Local Papuans not yet involved.	3	(Goram 2007)
1978	WIF and IMPD increased its catching. Around 2000-4000 tonnes exported in a month. More trawls used.	2	(Goram 2007)
1978	Outside fishermen came in large number from Madura – East Java. Bagan fisheries	2	(Goram 2007)

Year	Event summary	Code	Reference
	increase.		
1978	People from South Sulawesi introduce gillnets and longlines	2	(Goram 2007)
1979	Usaha Mina was also established and officially inaugurated by the then president Suharto. 15 boats increased to 80 boats. The capacity of 50 tonnage for each boat.	2	(Goram 2007)
1979	Bombing and Cyanide practices begin	2	(Kadarusman unpublished document)
1979	Second transmigration from Java to come to Sorong	3	(Goram 2007)
1979	Bombing and cyanide practices ran in large scales because of more competition while law enforcement was ineffective	2	(Goram 2007)
1979	More fishermen start catching fish, shrimps, octopus, tuna, sharks for market supply	4	(Goram 2007)
1979	Local Papuans start getting involved in catching sea cucumber .	3	(Goram 2007)
1979	Nets of more than 2 km used by Madura, Buton and Selayar boats for catching shark	2	(Goram 2007)
1980	Presidential Decree No 39/1980 banning trawling - The ban on trawl fishery has been implemented in 1980 along the Malacca Straits and off the North Coast of Java (President Decree No 39/1980).	1	(BRKP 2005a)

Year	Event summary	Code	Reference
	This ban was extended to all Indonesian waters in 1981		
1980	Supply of bomb materials from Buton on boat	2	(Kadarusman unpublished document)
1981	ADB - Second Irian Jaya Fisheries Development Project	3	(OAFIC 2007)
1982	Trawling for shrimp is restricted to the Arafura Sea (Eastern Indonesia)	1	(FAO 2000)
1982	Presidential Decree No. 085/1982 all units should be equipped with a Bycatch Efficiency Device (BED), which is a modified form of a Turtle Excluder Device (TED)	1	(FAO 2000)
1982	Act 4 - Declaration concerning conservation of living natural resources and their ecosystems	1	(MOEROI-UNDP 1997)
1982	Third transmigration from Java to come to Sorong	3	(Goram 2007)
1982	PT. Ramoi, Another tuna company, started its operation using 15 boats.	2	(Goram 2007)
1982	Bombing and cyanide practices became popular as fishermen try to compete with the bigger companies. Papuan people also engaged in destructive fishing	3	(Goram 2007)
1983	CIDA activities 1983 - 1995	3	(MOEROI-UNDP 1997)
1983	Act 5 - on EEZ of Indonesia	1	(MOEROI-UNDP 1997)

Year	Event summary	Code	Reference
1984	Government Regulation No. 15/1984 on resource utilization	1	(BRKP 2005a)
1985	Act 9 - concerning fisheries	1	(MOEROI-UNDP 1997)
1985	Blast fishing banned	1	(Djohani 1995)
1985	More Papuans engage in bombing, cyanide fishing. Coral reefs in East Salawati, East Bantanta and East Waigeo would be the targets since the sites were close to Sorong.	3	(Goram 2007)
1986	USAID – the ASEAN US cooperative program on marine sciences (1986-1992) aimed to increase capabilities in the ASEAN region to develop and implement comprehensive multidisciplinary and environmentally sustainable coastal resources management strategies.	3	(BRKP 2005b)
1987	Conservation areas declared in Kofiau, Misool, Salawati, Waigeo and Batanta with tentative recommended area (WWF 1987) 20, 1119 , 678 and 1137 and 100 km^{-2} respectively	1	(Mitchell 1987)
1987	Lobsters and grouper become target species	4	(Goram 2007)
1988	Shark fin fishing tended to decrease due to decline in shark population.	4	(Goram 2007)
1988	Irian Jaya canning started its survey to establish its factory of fish canning in Sorong	2	(Goram 2007)

Year	Event summary	Code	Reference
1988	The operation of WIF and IMPD tend to decrease because of its low catch.	2	(Goram 2007)
1988	More bombing, cyanide, compressors used by the smaller fishermen	2	(Goram 2007)
1988	Trawls of small size used by the smaller fishermen	2	(Goram 2007)
1988	Live reef fish export to Hong Kong begins from Indonesia	4	(Chan 2000)
1988	High foreign investment in fisheries	2	(PCI 2001a)
1988	Lobsters and grouper became popular among the traders and the local community	4	(Goram 2007)
1989	New guidelines under the MoU (Aus-Ind) were agreed in 1989, in order to clarify access boundaries for traditional fishers	3	(DAFF 2007)
1989	Office of the State Minister for population and environment compiled the national strategy for management of biological diversity	1	(PCI 2001b)
1990	Concerning conservation of living natural resources and their ecosystems - important in the effort to manage all designated protected areas in Indonesia	1	(PCI 2001b)
1990	LRFT has been growing steadily since early 1990s and has become a big business worth over a billion US dollars annually.	4	(Chan 2000)

Year	Event summary	Code	Reference
1990	Big ferry called the (Kapal Putih) brought bombing material	2	(Kadarusman unpublished document)
1990	High domestic investment in fisheries	2	(PCI 2001a)
1990	Decrease in coral and sponge trade (worldwide)	4	(PCI 2001a)
1990	Ministerial Decree 8 - development of institutes for community self help (Pembinaan Lembaga Swadaya Masyarakat)	3	(PCI 2001b)
1991	Shark availability for fishing reduced drastically	2	(Goram 2007)
1991	The natural resource management project (NRMP) (1991-1997) begins to improve resource management in Indonesia through policy analysis of production forests and protected areas	3	(PCI 2001b)
1991	Lobsters, grouper, Napoleon trading in large scale.	4	(Goram 2007)
1992	Cooperation on illegal fishing between Australia and Indonesia	3	(OceanLaw 2007)
1992	Country study on biodiversity in 1992 – the latter document was presented at UNCED in 1992	1	(PCI 2001b)
1992	Act 24 use of comprehensive and integrated approaches to the spatial management to support integrated coastal management of resources and permits	1	(BRKP 2005b)

Year	Event summary	Code	Reference
	designation of protected areas		
1992	Papuan traditional council (Dewan Adat Papua) by the Papuan congress		Goram pers. comm.
1992	Increase in operations in Sulawesi sea and Arafura sea	3	(PCI 2001a)
1992	Steep increase in longlining by national vessels (however the inc is from 1989 also for pukat ikan)	3	(PCI 2001a)
1993	Indonesia signed CBD	1	(MOEROI-UNDP 1997)
1993	TNC starts functioning in Indonesia	3	(PCI 2001b)
1993	Increase of inboard and outboard motors in Indonesia	2	(PCI 2001a)
1993	Increase in fishing crew - almost double	3	(PCI 2001a)
1993	Sea cucumber prices rise	4	(PCI 2001a)
1993	In 1990 the national development planning agency (BAPPENAS) formed a team to compile a NBAP which was published in 1993	1	(Gunarso and Davie 2000)
1993	The marine resource evaluation and planning project (MREP) started in 1993 for a duration of 5 years	3	(Dahuri and Dutton 2000)
1994	Act 5 - concerning ratification of the United Nations Convention on Biodiversity	3	(PCI 2001b)

Year	Event summary	Code	Reference
1994	Collapse of aquaculture	3	(PCI 2001a)
1994	Increase in coral and sponge trade (worldwide)	4	(PCI 2001a)
1994	The coastal environmental management planning project (CEMP) (1994-present)	1	(PCI 2001b)
1994	Indonesians working on foreign flag vessels decrease and a decrease in foreign longliners	3	(PCI 2001a)
1994	Rencana Umum Tata ruang Propinsi Irian Jaya Paduserai, Pengembangan Kawasan Konservasi di Irian Jaya	1	(Conservation International 2002)
1995	Revision of national conservation plan	1	(PCI 2001b)
1995	Number of national licensed fishing vessels increase but foreign vessels decrease	3	(PCI 2001a)
1995	'Ban on the Napoleon Wrasse Fish Haul' and 'Ban on Export of Napoleon Wrasse Fish' 'Decree of the Director General of Fisheries Regarding Size, Location, and Manners of Hauling Napoleon Wrasse'	1	(Lowe 2002)
1995	Ornamental fish trade increases (worldwide)	4	(PCI 2001a)
1995	Max Amer operting dive resort in Raja Ampat – Krie island – Dampier strait	3	Goram, pers. comm.
1996	Agricultural Ministerial Decree No. 197/1996 limited the maximum total	1	(BRKP 2005a)

Year	Event summary	Code	Reference
	length of gillnet to 2.5 km		
1996	Raja Ampat Marine WildLife Reserve established	1	(WWF/IUCN 1996)
1996	Napoleon wrasse and giant grouper listed as vulnerable on IUCN	3	(Mous et al. 2000)
1996	Fish landing starts at Sorong fishing port	4	(PCI 2001a)
1996	Teluk cenderawasih-kepulauan auri marine national park established	1	(WWF/IUCN 1996)
1997	Coremap 97-2001	3	(PCI 2001b)
1997	Increased use of bombing materials, cyanide and compressors (bombing material mostly made of fertilizers dropped from Wanci-Buton and Madura). Kofiau and Batanta Islands would be the anchoring and unloading sites.	2	(Goram 2007)
1997	Fishermen began to find difficulty in searching for sea cucumber, • Lobsters, Grouper, Napoleon tend to decrease	2	(Goram 2007)
1997	CI and Bappeda propinis apua, Universitas Cendrawasih, and Litbang Biologi – LIPI decided the priority areas for conservation to be about 24770660 ha.	1	(Conservation International 2002)
1997	Bagan tend to decrease	2	(Goram 2007)
1998	190 million requested for marine management for all Indonesia	3	(PCI 2001b)

Year	Event summary	Code	Reference
1998	Jump in tuna and shrimp and lobster export	4	(PCI 2001a)
1999	Balancing the number of licenses with available resources is implemented based on Agricultural Ministry Decree No. 995/1999.	1	(BRKP 2005a)
1999	Several stipulations made by the Sorong fisheries office	1	(Farid and Anggraeni 2003)
1999	International grouper/wrasse species survival group formed	3	(Sadovy 2000)
1999	Liveaboards operated in Raja Ampat about 14 to 17 in number	3	Goram pers. comm.
1999	The fishing waters are divided into the following fishing belts (Agricultural Ministerial Decree 392/1999)	3	(BRKP 2005a)
1999	Live reef food fish trade based in Hong Kong several difficulties	4	(Chan 2000)
2000	Usaha Mina collapsed	2	(Goram 2007)
2000	Fishermen find that Mouse grouper and Napoleon wrasse have become scarce	2	(Goram 2007)
2000	Ornamental fish and turtle trade increase	4	(Goram 2007)
2000	Political Condition in Papua changing under Special Autonomy Law (UU. 21. 2001).	1	(Satria 2006)

Year	Event summary	Code	Reference
2000	One of the Consensuses reached in the Congress is “Fundamental Rights Manifesto, which stated that The land in Papua and every thing containing in it belongs to the Customary Community People and not the Indonesian State. Dewan Adat Papua established.	3	(Goram 2007)
2001	MMAF has implemented the re-registration of fishing vessel (September 2001)	1	(BRKP 2005a)
2001	The Indonesian National Workshop to deal with IUU fishing was held in Jakarta on 30 April–1 May 2001.	3	(BRKP 2005b)
2001	CI scientific survey in Raja Ampat	3	(Diamond 1986)
2001	Community based surveillance - Ministerial Decree of Fisheries No Kep 59/Men/2001	1	(Farid and Anggraeni 2003)
2001	Downturn of seafood business in Hong Kong after 9/11 (attack on WTC)	4	(McGilvray and Chan 2003)
2001	Joint ASEANSEAFDEC conference, entitled The ASEANSEAFDEC Conference on Sustainable Fisheries for Food Security in the New Millennium: “Fish for the People”, was held in Bangkok	3	(Heazle 2005)
2001	Australia-Indonesia Working Group on Marine Affairs and Fisheries established in June 2001, under the auspices of the AIMF.	3	(DAFF 2007)

Year	Event summary	Code	Reference
2002	The International Commission for the Conservation of Atlantic Tuna (ICCAT) had threatened to restrict export of Indonesian fish products in 2002. This measure was based on the allegation that tuna longline vessels flying the flag of Indonesia have been co	3	(BRKP 2005b)
2002	Indonesia Biodiversity Strategy and Action Plan Region Plan for Papua decided	1	()
2002	Bupati decree on Tourism	1	(Rabu 2006)
2002	Raja Ampat created as a new regency based on No. regulation No. 26 in 2002. In 2003 it was declared its own regency.	1	(Rabu 2006)
2002	Management of fisheries-related businesses is currently regulated by Government Regulation No. 54/2002	1	(BRKP 2005a)
2002	TNC scientific survey in Raja Ampat	3	(Donnelly et al. 2003)
2002	Shark finning in Kapadiri area in Waigeo (by a fishing company from Philippines)	2	(Farid and Anggraeni 2003)
2003	Draft national plan on IUU	1	(ACIAR 2008)
2003	MMAF Decree No 10/2003	1	(BRKP 2005a)
2003	Management and Policy Frameworks for Illegal, Unreported and Unregulated (IUU) Fishing in Indonesian and Philippine Waters” (ACIAR ProjectNo. FIS/2002/019)	3	(ACIAR 2008)

Year	Event summary	Code	Reference
2003	Tomolol declaration	1	(Conservation International 2005)
2003	CI appealed to print and radio in Sorong municipality to cover natural resources management issues in Raja Ampat	3	(Rabu 2006)
2003	Prices of live reef fish fall as a result of SARS outbreak	4	(Muldoon et al. 2005)
2003	Raja Ampat declared a maritime regency	3	(Conservation International 2008.)
2004	National IUU fishing workshops 2004-05	1	(ACIAR 2008)
2004	Law of the Republic of Indonesia No. 31 of 2004	1	(MMAF 2004)
2004	Indonesian and foreign fishing vessels have been apprehended for conducting fishing activities outside the terms and conditions of their licenses	1	(BRKP 2005a)
2004	139 fishing licenses have been revoked for failure to submit deletion certificate requirements	1	(BRKP 2005a)
2004	Andi - set up a resort in Batbetem island in Misool	3	Goram pers. comm.
2005	Ttraining of observers in turtle de-hooking and resuscitation techniques in Indonesia and providing training to beach monitoring projects	2	(NOAA 2005)
2005	Another 28 licenses have been revoked for the first quarter of 2005, composing of	1	(BRKP 2005a)

Year	Event summary	Code	Reference
	26 fishing vessel licenses and two fisheries business licenses		
2005	The National Workshop on Illegal, Unreported and Unregulated Fishing was held in Jakarta	3	(BRKP 2005b)
2005	CI - Tabloid launched	3	(Rabu 2006)
2005	Humphead wrasse listed on CITES	3	(Sadovy 2005)
2006	Local adat for Waigeo and Misool	1	Goram pers. comm.
2006	Seismic survey for oil mining	3	(Rabu 2006)
2006	Declaration of MPA network	1	(Rabu 2006)
2006	Election of custom leader to take care of illegal fishery in each village	1	Goram pers. comm.
2007	Local adat for Kofiau - 2007	1	Goram pers. comm.
2007	CI and TNC trying to develop a patrol system with the new patrol boat Imbekwan	1	(Rabu 2006)
2007	Development of entry system to Raja Ampat	1	(Rabu 2006)

Appendix Table C-2 Predicted incentives for each year for each fishery
The incentives are categorized into High, Medium High, Medium, Medium Low and Low.

Year	Illegal Reef fish	Reef fish	Tuna	Anchovy	Sharks	Sea cucumber	Lobster
1960		L	L	L	L		L
1961		L	L	L	L		L
1962		L	L	L	L		L
1963		L	L	ML	L		L
1964		L	L	ML	L		L
1965		L	L	ML	L		L
1966		ML	L	ML	L		L
1967		ML	L	ML	L		L
1968		ML	L	ML	L		L
1969		ML	L	ML	L		L
1970		ML	ML	ML	L		ML
1971		ML	ML	ML	L		ML
1972		ML	ML	ML	L		ML
1973		ML	ML	ML	L		ML
1974		ML	ML	ML	L		L
1975		ML	M	ML	ML		L
1976		ML	M	ML	ML	L	ML
1977		ML	M	M	ML	L	L
1978		ML	MH	M	ML	L	L
1979		ML	H	MH	ML	ML	ML
1980		M	H	MH	ML	ML	ML
1981		M	H	MH	MH	ML	M
1982		M	H	MH	MH	ML	M
1983		M	H	MH	MH	M	M
1984	L	M	H	MH	H	M	M
1985	L	M	H	H	H	M	M
1986	L	M	H	H	H	MH	M
1987	L	MH	H	H	H	MH	MH
1988	M	MH	H	H	H	MH	MH
1989	M	MH	H	H	H	MH	MH
1990	MH	H	H	H	H	H	H
1991	MH	H	H	H	H	H	H
1992	MH	H	H	H	H	H	H
1993	MH	H	H	H	H	H	H
1994	MH	H	H	H	H	H	H
1995	H	H	H	H	H	H	H
1996	H	H	H	H	H	H	H
1997	H	H	H	H	H	H	H
1998	H	H	H	H	H	H	H
1999	H	H	H	H	H	H	H
2000	H	H	H	H	H	H	H
2001	H	H	MH	H	H	H	H
2002	H	H	MH	H	MH	MH	H
2003	MH	MH	MH	H	MH	MH	MH
2004	MH	MH	MH	H	MH	MH	MH
2005	MH	MH	MH	H	MH	MH	MH
2006	MH	MH	M	H	MH	MH	MH
2007	MH	MH	M	H	MH	MH	MH

Anchor points for the fisheries

Illegal reef fishery

Blast fishing and cyanide fishing for reef fishes started in 1979 by communities from Buton and Biak who came to fish in western Raja Ampat (Kadarusman unpublished document). Butonese brought bombing material to Raja Ampat in wooden boats. During 1981 to 1985, bomb fishing spread to the islands of Salawati, Batanta, Kofiau, and Misool in Raja Ampat. By the year 1990, bombing became so popular that public transport ferries (kapal putih) were used to bring bombing material from as far as East Borneo, Sulawesi, and Java (Goram 2007; Kadarusman unpublished document). There was a decrease in bomb fishing after 2004 due to increased awareness among fishers about the destructive effect of bomb fishing. The illegal reef fish catch ranged between 660 and 1100 tonnes in 2003 and reduced to a range of 400 to 600 tonnes in 2004 (Kadarusman unpublished document).

From the 1980s, fishers from Wakatobi in South-East Sulawesi often came to Boo Island (Kofiau) to fish for reef fish. In 2001-2002 fishers from Wakatobi earned about Rp 10 to 15 million (10000-15000 $USD) for 5 tonnes of fish caught every week. They fished about 4 times a week with high speed engine boats. After 2002, the catches declined, and by 2007, most of these fishers moved to Malukku jurisdiction, west of Raja Ampat islands. On average, of the boats engaged in illegal fishing for reef fishes in Raja Ampat, about 75% were engaged in bomb fishing, and 25% were engaged in cyanide fishing (Suebo,A. pers. comm.).

The city of Seram is located in Sorong Regency, east of Misool islands. The number of fishers from Seram fishing in Misool waters went up during 1995 to 2000; the fishers carried guns and fished illegally using chemicals. Consequently, the government sent the Navy to control the illegal fishers. Since 1996 the security employed by pearl farms has controlled illegal fisheries activities (Sumule and Donnelly 2003) and the entry of fishers from Seram into the waters adjacent to Misool (Suebo, A. pers. comm.). Recently about 10 sampans (flat bottomed wooden boats of size 3.5 to 4.5 m)—sampans may be propelled by oars or fitted with outboard motors—arrive on a mothership from Flores

Islands to Kedo Kedo (Misool) to fish for live groupers (Erdmann, pers. comm.). About 10 to 20 large boats from Seram also fish on reefs in South-East Misool.

About 30 fishing boats originally from South-East Sulawesi are based in Sorong and fish on reefs using bomb and cyanide. They distribute the chemicals for bombing and cyanide fishing to local fishers in Sorong and Halmahera (Suebo,A. pers. comm.). Almost 90% of the grouper and Napoleon wrasse are caught by fishers from outside Raja Ampat while the local fishers catch about 90% of the other reef fishes (Erdmann, pers. comm.).

The estimates above add up to a total of about 61 boats. Each boat is estimated to catch 40 baskets of fish in each boat; each basket has a capacity of 4 to 8 kg (Kadarusman unpublished document); this value is raised to the number of boats to estimate the total catch.

Karambas are floating net cages in which live reef fish are held. The karambas are privately owned, and local fishers sell their live fish catch to the local karamba owners. The resource evaluation assessment survey (Donnelly et al. 2003) shows several areas around Raja Ampat prone to destructive fishing practices. Karambas from Gam to Mansuar in Waigeo were reported to produce about 12 to 24 tonnes per year (Erdmann and Pet 2002). 25 karambas were reported from a survey of fishing villages in SW Waigeo Island (Farid and Anggraeni 2003); they reported that most of the live fish observed in the karambas were collected using cyanide. A resource use survey in Kofiau Island (Muljadi unpublished data) reported 8 karambas around Kofiau Island. Production rate of each karamba was about 400 kg per month (Erdmann and Pet 2002) totaling to about 38.4 tonnes from only around Kofiau. Farid and Anggraeni (2003) surveyed four villages in South Waigeo and estimated the Napoleon wrasse and grouper catch; they found that the estimate exceeded the catch estimated by the fisheries office in Sorong. The aerial survey (Barmawi 2006) reported a large number of fish cages in Raja Ampat; however, it was not evident how many of these were karambas. Hence an estimate of the illegal reef catch in karambas could not be made.

Unreported catch

Estimates for artisanal catch of the reef fish, tuna, anchovy, shark, sea cucumber, and lobster were obtained from the valuation report (Dohar and Anggraeni 2007). Anchor points for unreported catches of the reef fishes were based on a comprehensive rural appraisal (CRA) (Muljadi 2004) conducted by TNC field staff in the Misool and Kofiau. The CRA report estimated that only 31.21% of the fishers marketed their catch in Sorong. We have assumed that the remainder was used for subsistence or was sold locally or to foreigners and therefore never made it to the Sorong statistics. The grouper and Napoleon wrasse captured by the industrial fisheries in Raja Ampat was about 3% of the total Raja Ampat reef fish catch (Palomares and Heymans 2006).

Farid and Anggraeni (2003) estimated reef fish catch from 12 villages in South Waigeo for year 2002. The estimates calculated were increased proportionately to represent the total fisher population in Waigeo. Population estimates from Department of Statistics (BPS 2001) and the Fisheries Office (DKP 2007) were used to estimate reef fish catch from Waigeo. For Misool and Kofiau Islands, the CRA report (Muljadi 2004) was used to calculate the fraction of fishers engaged in fishing for different species. The catchability rates in fisheries in Waigeo were assumed to be held same in Kofiau and Misool Islands. The same ratio was used for fishers from Batanta and Salawati.

There was anonymous information that the tuna industries based in Sorong highly under-report their tuna catch. We used an anchor point from a survey conducted by Dohar and Anggraeni (2007), but the estimate was based only on the information from 2 tuna companies in the region. The catch from the 2 companies (819 tonnes) exceeded the catch reported by DKP (369 tonnes).

The average estimate for anchovy catch by a bagan (lift net) was reported to be 49 to 76 tonnes annually (Bailey et al. 2008). The number was raised to represent the total number of bagans that were seen during the aerial survey in Raja Ampat (Barmawi 2006) to arrive at an anchor point for the anchovy fishery. The total anchovy catch ranged from 15400 to 24000 tonnes; of this amount about 10% was reported.

More than 100 boats (about 7 m long) from Halmahera fish for sharks in Raja Ampat. Bigger vessels collect shark fin from the small vessels fishing in Raja Ampat and ship it to Halmahera or Makassar. These operations have financial support from outside Indonesia (Suebo, A. pers. comm.). Shark fin collectors in Sorong gave 8 to 10 million Rp (8000-10000 $USD) per trip to fishers from outside Raja Ampat to catch sharks. Each trip lasted 2 to 3 weeks. The shark fin price in 2002 was 1.2 million Rp per kg (Farid and Anggraeni 2003). Assuming 20 trips in a year, the catch would be 130 to 160 kg per boat. The number was increased to account for the number of shark fishing boats in Raja Ampat (Suebo, A. pers. comm.) artisanal catches of shark (Dohar and Anggraeni 2007) were added to calculate an anchor point for the shark fisheries (634 tonnes).

The sea cucumber catch from 12 villages in South Waigeo Island (Farid and Anggraeni 2003) was raised to account for the fisher population fishing for sea cucumber in Raja Ampat to arrive at an anchor point for sea cucumbers (76 tonnes per year). The proportion of fishers fishing for sea cucumber in Raja Ampat was based on the CRA report (Muljadi 2004). Another sea cucumber anchor point (26 tonnes per year) was arrived at using the estimates for commercial catch and gleaned catch from the report by Dohar and Anggraeni (2007).

The unreported catch for lobster was obtained from the valuation report by Dohar and Anggraeni (2007). The number of active boats from the areas adjoining Raja Ampat was estimated to be about 60, of which 25% were engaged in cyanide fishing and the rest in blast fishing. We assumed the boats using cyanide were fishing for lobster among other species. The lobster catch from 12 villages in South Waigeo (Farid and Anggraeni 2003) was raised proportionately to account for the fisher population fishing for lobster in Raja Ampat to calculate an anchor point for lobsters (600 tonnes). The proportion of fishers fishing for lobster in Raja Ampat was based on the CRA report (Muljadi 2004).

References

ACIAR. 2008. Management and policy frameworks for illegal, unreported and unregulated (IUU) Fishing in Indonesian and Philippine waters. Australian Government Australian Center for International Agricultural Research. 2002. Project ID: FIS/2002/019. Available: <www.aciar.gov.au/project/FIS/2002/019> [Accessed on May 15, 2008].

ACIAR. 2008. Management and policy frameworks for illegal, unreported and unregulated (IUU) Fishing in Indonesian and Philippine waters. Australian Government Australian Center for International Agricultural Research. 2002. Project ID: FIS/2002/019. Available: <www.aciar.gov.au/project/FIS/2002/019> [Accessed on May 15, 2008].

APFIC. 2007. The history of industrial marine fisheries in South East Asia, Industrial fisheries. Statistics and information. Asia Pacific Fishery Commission. Available: <www.apfic.org/modules/xfsection/article.php?articleid=20> [Accessed on May 20, 2007].

Bailey M., C. Rotinsulu, and U. R. Sumaila. 2008. The migrant anchovy fishery in Kabui Bay, Raja Ampat, Indonesia: Catch, profitability, and income distribution. Marine Policy **32**(3):483-488.

Barmawi M. 2006. Aerial survey in Raja Ampat: An observation of fisheries activity, large marine fauna, and coastal habitat (January 9-13, 2006). Data Report. The Nature Conservancy Coral Triangle Center. Jl. Pengembak no. 2, Sanur, Denpasar. Available: joanne_wilson@tnc.org.

BPS. 2001. Dalam angka. Census (in Indonesian). Badan Pusat Statistik Kabupaten Sorong. Department of Statistics, Sorong Regency. 238p.

BRKP. 2005a. Fishing vessel registration and licensing system in Indonesia. Badan Riset Kelautan dan Perikanan (Research Body of Marine Affairs and Fisheries) Directorate of Fisheries Services and Enterprises. Directorate General of Capture Fisheries. Paper presented at the National Workshop on Illegal Unreported and Unregulated (IUU) Fishing in Indonesia. Jakarta, Indonesia. Funded by the Centre for Maritime Policy. University of Wollongong. Research Centre for capture fisheries. Agency for Marine and Fisheries Research Department of Marine Affairs and Fisheries. Organized by Australian Centre for International Agricultural Research. 28 April 2005 (12pp).

BRKP. 2005b. Proceedings of the National Workshop on Illegal, Unreportedand Unregulated (IUU) Fishing Jakarta, Indonesia. Badan Riset Kelautan dan Perikanan. Research Body of Marine Affairs and Fisheries. 28 April 2005. Funded by the Centre for Maritime Policy. University of Wollongong. Research Centre for capture fisheries.

Agency for Marine and Fisheries Research Department of Marine Affairs and Fisheries. Organized by Australian Centre for International Agricultural Research.

Chan P. S. W. 2000. Current status of the live reef fish trade based in Hong Kong. SPC Live Reef Fish Information Bulletin **7**:8-9.

Chauvel R. 2005. Constructing Papuan nationalism: history, ethnicity, and adaptation. Washington DC: East-West Center Washington. Policy Studies **14** 140p.

Conservation International. 2008. Environmental Friendly Development in Raja Ampat. Available: <http://www.conservation.or.id/home.php?catid=30&tcatid=178&page=g_peluang.detail > [Accessed on July 30, 2008].

Conservation International. 2005. Sebuah masa depan kehidupan, Visi, Aksi, Hasil: Strategi CI 2005-2010. Translated from "A future for life: Vision, Action, Results: CI's Strategy for 2005-2010. Conservation International. Washington, DC. Available: C.Rotinsulu. CI. Jl.Gunung Arfak.45.Sorong, Papua, Indonesia. chris@conservation.or.id.

Conservation International. 2002. Indonesia Biodiversity Strategy and Action Plan Region Plan – Laporan Akhir (Final Report) Jayapura, October 2002 – Koordinator Pelaksanan Penyusunan. Indonesia Biodiversity Strategy and Action Plan Regional Papua Kantor Conservation International Papua Program Jl. Bhayangkara I/No. 33, Jayapura 99112. Available: chris@conservation.or.id.

DAFF. 2007. Indonesia-Australia fisheries cooperation. Australian Government Department of Agriculture, Fisheries and Forestry. Available: <http://www.daffa.gov.au/fisheries/international/regional/indonesia> [Accessed on May 20, 2007].

Dahuri R., I. M. Dutton. 2000. Integrated coastal and marine management enters a new era in Indonesia. Integrated Coastal Zone Management **1**:11-16.

Diamond J. 1986. The design of nature reserve system for Indonesian New Guinea. In: Soule ME. Conservation Biology: the science of scarcity and diversity. Sinauer and Associates, Massachussetts. 485-503p.

Djohani R. 1995. The combat of dynamite and cyanide fishing in Indonesia. The Nature Conservancy, Jakarta, Indonesia. 47pp.

DKP. 2007. Laporan tahun 1994, 1998, 1999, 2002. Annual Reports – 1994.; 1998.; 1999. Departemen Kelautan dan Perikanan. Department of Marine Affairs and Fisheries. Sorong, Indonesia [in Indonesian].

Dohar A. G., D. Anggraeni. 2007. Valuasi Ekonomi Sumberdaya Alam di Kepulauan Raja Ampat. Conservation International, Bekerjasama dengan, Universitas Negeri Papua. (in Indonesian) Economic valuation report for Raja Ampat, Indonesia; Conservation International Indonesia.Available: Rotinsulu C. Gunung Arfak CI. Jl. 45. Sorong, Papua, Indonesia. Available: chris@conservation.or.id.

Donnelly R., D. Neville, and P. J. Mous. 2003. Report on a rapid ecological assessment of the Raja Ampat Islands, Papua, Eastern Indonesia. The Nature Conservancy Coral Triangle Center. Indonesia. Final Draft. 250p.

Erdmann M. V., J. S. Pet. 2002. A rapid marine survey of the northern Raja Ampat islands. Henry Foundation/The Nature Conservancy/NRM/EPIQ. Indonesia: The Nature Conservancy Coral Triangle Center. Available: <www.coraltriangle.org/downloads/Raja4%20trip%20report.pdf> [Accessed on July 15, 2007].

FAO. 2000. Report of the four GEF/UNEP/FAO regional workshops on reducing the impact of tropical shrimp trawl fisheries (Lagos, Nigeria, 15-17 December 1999. Puntareñas, Costa Rica, 15-17 January 2000. Teheran, Iran, 28 February-1 March 2000. Denpasar, Bali, Indonesia, 6-8 March 2000). Food and Agriculture Organization of the United Nations. Rome Fisheries Report No. 627. 40p.

Farid M., D. Anggraeni. 2003. Preliminary assessment of community-based natural resource management and conservation options in South Waigeo. Indonesia: Conservation International Indonesia. Available: chris@conservation.or.id.

Goram Y. 2007. Interview (Sawen, Sawaki, J., Raubaba, S. and L. Metansan, 10-11 February, 2007).

Gunarso P., J. Davie. 2000. Biodiversity conservation in Indonesia: Policy and politics. In: Tacconi L., editor. Biodiversity and ecological economics: participation, values, and resource management. Earthscan. 254p.

Heazle M. 2005. Regional fisheries regulation as a solution to Southeast Asia's emerging fisheries crisis. Panel 2 - Maritime risk society in East Asia: Is the state the answer or the problem? People and the Seas III: New Directions in Coastal and Maritime Studies. Centre for Maritime Research Biannual Conference July 2005. University of Amsterdam, Amsterdam.

Kadarusman K. unpublished document. Studi Awal: Kegiatan Pengeboman Ikan di Pulau Buaya. Akademi Perikanan Sorong. Departemen Kelautan dan Perikanan. The study of early fish bombing activities in Crocodile Island Sorong Fisheries Academy. Department of Marine affairs and Fisheries. Indonesia: Dewan Adat Suku Maya, Kepulauan Raja Ampat (The Maya Council, Raja Ampat islands). [in Indonesian].

Lowe C. 2002. Who is to blame? Logics of responsibility in the live reef food fish trade in Sulawesi, Indonesia. SPC Live Reef Fish Information Bulletin **10**(6):7-16.

McGilvray F., T. Chan. 2003. Market and industry demand issues in the live reef food fish trade. SPC Live Reef Fish Information Bulletin **11**(4):36-39.

Mitchell A. H. 1987. Implementation of conservation in Irian Jaya, Indonesia. Progress Report Irian Jaya Conservation project. WWF/IUCN No 3770.

MMAF. 2004. Law of the Republic of Indonesia No. 31 of 2004 concerning fisheries published by Arafura and Timor Sea Expert Forum (ATSEF) Indonesia Secretariat, Cooperation between the Ministry of Marine Affairs and Fisheries and UNDP, Jl. MT. Haryono Kav. 52-53. Jakarta 12770 111pp English translation Prof. Dr. Etty R. Agoes, Sh, LLM. Padjadjaran University, Indonesia.

MOEROI-UNDP. 1997. Agenda 21 – Indonesia - A national strategy for sustainable development. State Ministry for Environment, Republic of Indonesia, United Nations Development Program, March 1997. 601p.

Mous P. J., L. Pet-Soede, M. Erdmann, H. S. J. Cesar, Y. Sadovy, and J. S. Pet. 2000. Cyanide fishing on Indonesian coral reefs for the live food fish market–What is the problem? Collected Essays on the Economics of Coral Reefs **7**(5):69-76.

Muldoon G., L. Peterson, and B. Johnston. 2005. Economic and market analysis of the live reef food fish trade in the Asia-Pacific region. SPC Live Reef Fish Information Bulletin **13**(1):35-41.

Muljadi A. unpublished data. The Nature Conservancy Coral Triangle Center. Jl Gunung Merapi no. 38, Kampung Baru, Sorong, Papua, Indonesia 98413. Available: amuljadi@tnc.org.

Muljadi A. 2004. Coastal Rural Appraisal report in Kofiau and Misool area of Raja Ampat Islands. Sorong, Indonesia: The Nature Conservancy, South East Asia Centre for Marine Protected Areas. 12p.

NOAA. 2005. NOAA fisheries reducing bycatch 2005 report. National Oceanic and Atmospheric Administration. Washington, DC. 10p. Available: <www.nmfs.noaa.gov/by_catch/2005BycatchReportJune9v2.pdf > [Accessed on January 4, 2008].

OAFIC. 2007. Projects East Asia and the Pacific. Overseas Agro Fisheries Consultants Co. Ltd. Available: <www.oafic.co.jp/projects-eastasia.html> [Accessed on July 7, 2007].

OceanLaw. 2007. Bilateral Treaties Compendium. Internet Guide to International Fisheries Law, Law of the Sea Information and Consultancy Services. Available:

<http://www.intfish.net/treaties/bilaterals/texts/aus-ins/1993.htm> [Accessed on March 4, 2007].

Palomares M. L. D., J. J. Heymans. 2006. Historical Ecology of the Raja Ampat Archipelago, Papua Province, Indonesia. Fisheries Centre Research Reports **14**(6):1-114.

PCI. 2001a. Study on fisheries development policy formulation. Vol III. Database for Analysis of Study. Pacific Consultants International under Jakarta Fishing Port/Market Development Project (Phase IV: JBIC Loan No. IP-403). Jakarta. 138p.

PCI. 2001b. Study on fisheries development policy formulation, Vol. I. White Paper. Pacific Consultants International under Jakarta Fishing Port/Market Development Project (Phase IV: JBIC Loan No. IP-403). Jakarta. 234 pp. + Annexes.

Rabu P. 2006. Tabloid Raja Ampat. Conservation International, Indonesia. **6**:1-4.

Sadovy Y. 2005. Humphead wrasse listed on CITES. Live reef fish information bulletin **13**(1):42-43.

Sadovy Y. 2000. Grouper/wrasse species survival group formed. SPC Live Reef Fish Information Bulletin **7**(5):13.

Satria A. 2006. Decentralization of property right in marine fisheries: Indonesia perspective. Center for Coastal and Marine Resources Studies. Bogor Agricultural University, Indonesia. Conference of the International Association for the Study of Common Property (IASCP) June 19-June 23, 2006. 27p. Available: <www.indiana.edu/~iascp/bali/papers/Satria_Arif.pdf > [Accessed on March 16, 2008].

Sumule A., R. Donnelly. 2003. Socio-economic conditions in the Raja Ampat islands. Pages 21-41 In: Donnelly R., D. Neville, and P.J. Mous, editors. Report on a rapid ecological assessment of the Raja Ampat Islands, Papua, Eastern Indonesia. The Nature Conservancy Coral Triangle Center, Indonesia. Final Draft.

WWF/IUCN. 1996. Indonesian country study on integrated coastal and marine biodiversity management. Ministry of State for environment, Republic of Indonesia in cooperation with Directorate for Nature Management Kingdom of Norway. Appendix I Marine Conservation Areas A:121-125.

Appendix D Dispersal Rates in Raja Ampat Ecospace Model[39]

The ‘dispersal rate’ is the rate (km.yr^{-1}) at which organisms will disperse in an ecosystem as a result of random movements. The value is used to calculate the fraction of biomass of the functional group in the cell that would move into the adjacent cell at the next time step and hence is important in generating the spatial distribution of organisms in the ecosystem. The default value for all groups is 300 km·yr^{-1}, except for detritus group for which the rate is 10 km·yr^{-1}. Where necessary, the dispersal rates were adjusted according to the movement patterns of functional groups in the model based on published literature and expert comments made by Dr Neil Gribble Contact: Queensland Dept of Primary Industries & Fisheries, Northern Fisheries Center, Cairns. Email: Neil.gribble@dpi.qld.gov.au). The dispersal rates used in the model and the sources for the values are presented in Table 1. The following is a brief description of dispersal rates used for corals and reef associated fish in the model.

Dispersal rates for coral species range from 1-3 km·yr^{-1} based on observations that coral reefs primarily self-seeded with highest settlement density at 500m distance from the adult population (Shanks et al. 2003). The value also considers the conclusions of Sammarco and Andrews (1988) coral recruitment declined logarithmically with distance from the reef. However calculations based on stored energy and settlement time have shown that coral larvae are capable of long-distance dispersal (Richmond 1987). The dispersal rates for adult reef fish range from 5-50 km·yr^{-1}. Studies have concluded that butterfly fish “spent their entire lives associated with a small portion of the reef” (Bardach 1958). Similar results were observed for angelfish and surgeon fish (Bardach 1958; Ormond and Gore 2005). Groupers range within 20 km (Bardach 1958); Lutjanids

[39] This material in the appendix has been submitted for publication as an online appendix with the manuscript paper based on Chapter 3 of this thesis.

range within 20 km (Roberts and Polunin 1991). However, several other studies have indicated higher dispersion ranges (Barlow 1981; Ormond and Gore 2005; Jue 2006) but the values chosen (grouper 30 km·yr^{-1}, snapper 30 km·yr^{-1}) were based on expert consultation with Dr Neil Gribble. Other groups of adult reef fish in the model were assigned dispersal rates of 50 km·yr^{-1} based on studies that concluded that fishes of family Siganidae, Caesionidae and Pomacentridae are highly mobile (Barlow 1981; Roberts et al. 2001), though other families like Haemulidae do not disperse long distances (Bardach 1958). Dispersal rates for juvenile reef fish range from 100-150 km·yr^{-1} (Fisher 2005). Other studies suggest that larvae are retained "at natal reefs", in a range of 30 km (Mora and Sale 2002; Ormond and Gore 2005); however, maximum suggested dispersal ranges to 219 km (Roberts 1997). The lengthy spawning migrations were not considered because the dispersal rate in Ecospace does not represent directed migration pattern. The high dispersal rates chosen for the juveniles essentially capture the wide dispersal of the larvae and juveniles from an adult population.

Appendix Table D-1 Dispersal rates in Ecospace model

Functional Groups	Dispersal rate (km.yr^{-1})	Functional Groups	Dispersal rate (kmyr^{-1})
Mysticetae	10000	Adult small demersal	100
Piscivorous odontocetae	10000	Juvenile small demersal	100
Deepdiving odontocetae	10000	Adult large planktivore	200
Dugongs	300	Juvenile large planktivore	1000
Birds	300	Adult small planktivore	200
Reef associated turtles	1000	Juvenile small planktivore	1000

Functional Groups	Dispersal rate (km.yr^{-1})	Functional Groups	Dispersal rate (kmyr^{-1})
Green turtles	1000	Adult anchovy	500
Oceanic turtles	10000	Juvenile anchovy	500
Crocodiles	300	Adult deepwater fish	300
Adult groupers	30	Juvenile deepwater fish	300
Subadult groupers	220	Adult macro algal browsing	50
Juvenile groupers	100	Juvenile macro algal browsing	100
Adult snappers	30	Adult eroding grazers	50
Subadult snappers	350	Juvenile eroding grazers	100
Juvenile snappers	150	Adult scraping grazers	5
Adult Napoleon wrasse	30	Juvenile scraping grazers	100
Subadult Napoleon wrasse	100	Detritivore fish	50
Juvenile Napoleon wrasse	150	Azooxanthellate corals	2
Skipjack tuna	1000	Hermatypic scleractinian corals	1
Other tuna	1000	Non reef building scleractinian corals	2

Functional Groups	Dispersal rate ($km.yr^{-1}$)	Functional Groups	Dispersal rate ($kmyr^{-1}$)
Mackerel	1000	Soft corals	2
Billfish	1000	Calcareous algae	2
Adult coral trout	6	Anemonies	5
Juvenile coral trout	150	Penaeid shrimps	100
Adult large sharks	300	Shrimps and prawns	30
Juvenile large sharks	300	Squid	300
Adult small sharks	100	Octopus	50
Juvenile small sharks	100	Sea cucumbers	20
Whale shark	1000	Lobsters	20
Manta ray	1000	Large crabs	20
Adult rays	1000	Small crabs	20
Juvenile rays	1000	Crown of thorns	20
Adult butterflyfish	5	Giant triton	20
Juvenile butterflyfish	10	Herbivorous echinoids	20

Functional Groups	Dispersal rate ($km.yr^{-1}$)	Functional Groups	Dispersal rate ($kmyr^{-1}$)
Cleaner wrasse	3	Bivalves	20
Adult large pelagic	500	Sessile filter feeders	20
Juvenile large pelagic	500	Epifaunal detritivorous invertebrates	20
Adult medium pelagic	300	Epifaunal carnivorous invertebrates	20
Juvenile medium pelagic	300	Infaunal invertebrates	20
Adult small pelagic	200	Jellyfish and hydroids	300
Juvenile small pelagic	200	Carnivorous zooplankton	300
Adult large reef associated	50	Large herbivorous zooplankton	300
Juvenile large reef associated	150	Small herbivorous zooplankton	300
Adult medium reef associated	50	Phytoplankton	300
Juvenile medium reef associated	150	Macro algae	20
Adult small reef associated	30	Sea grass	5

Functional Groups	Dispersal rate ($km.yr^{-1}$)	Functional Groups	Dispersal rate ($kmyr^{-1}$)
Juvenile small reef associated	100	Mangroves	5
Adult large demersal	200	Fishery discards	10
Juvenile large demersal	200	Detritus	10

References

Bardach J. E. 1958. On the movements of certain Bermuda reef fishes. Ecology **39**(1):139-146.

Barlow G. W. 1981. Patterns of parental investment, dispersal and size among coral-reef fishes. Environmental Biology of Fishes **6**(1):65-85.

Fisher R. 2005. Swimming speeds of larval coral reef fishes: impacts on self-recruitment and dispersal. Marine Ecology Progress Series **285**:223-232.

Jue N. 2006. Exploring the structure of genetic variation and the influences of demography on effective population size in the gag grouper *Mycteroperca microlepi* (Goode and Bean). Journal of fish biology **69**:217–224.

Mora C., P. F. Sale. 2002. Are populations of coral reef fish open or closed? Trends in Ecology & Evolution **17**(9):422-428.

Ormond, R., M Gore 2005. No-take zones: does behaviour matter? Aquatic telemetry. Advances and applications. Proceedings of the fifth Conference on Fish Telemetry held in Europe, Ustica, Italy, 9-13 June 2003. FAO/COISPA, Rome. 71-89.

Richmond R. 1987. Energetics, competency, and long-distance dispersal of planula larvae of the coral Pocillopora damicornis. Marine Biology **93**(4):527-533.

Roberts C. M., N. V. C. Polunin. 1991. Are marine reserves effective in management of reef fisheries? Reviews in Fish Biology and Fisheries **1**(1):65-91.

Roberts C. M. 1997. Ecological advice for the global fisher crisis. Trends in Ecology & Evolution **12**(1):35-38.

Roberts C. M., J. A. Bohnsack, F. Gell, J. P. Hawkins, and R. Goodridge. 2001. Effects of marine reserves on adjacent fisheries. Science **294**(5548):1920-1923.

Sammarco P. W., J. C. Andrews. 1988. Localized dispersal and recruitment in Great Barrier Reef corals: the Helix experiment. Science **239**(4846):1422.

Shanks A. L., B. A. Grantham, and M. H. Carr. 2003. Propagule dispersal distance and the size and spacing of marine reserves. Ecological Applications **13**(1) Supplement:S159-S169.

Appendix E Creation of Sub-Area Models[40]

Sub area model locations

The sub-area models were built for 3 of the 7 areas where the Regency had declared marine protected areas. The three areas are the same as the ones analyzed for ecological change using the Raja Ampat Ecospace model. The Kofiau Island Ecospace model extends from 129^{o} 14' E and 1^{o} 5' S in the north-west corner to 130^{o} 1' E and 1^{o} 20' S in the south east corner. The Kofiau Island Ecospace model extends from 129^{o} 14' E and 1^{o} 5' S in the north-west corner to 130o 1' E and 1o 20' S in the south east corner. The Dampier Strait model extends from 130^{o} 25' 12" E and 0^{o} 18' S at the northwest corner to 131^{o} 21' 36" E and 0^{o} 50' S at the southeast corner. The sub-area models representing Kofiau Island and South-East Misool are primarily coral and reef-fish models that have been expanded to include important pelagic elements. Dampier Strait is an important and productive area in Raja Ampat that sustains a major artisanal fishery for anchovy due to a region of strong upwelling.

Parameter estimations: Biomass

Area ratio conversions

For most functional groups, biomasses are determined for the sub-area models based on master biomass values used in the Raja Ampat (2005) model scaled according to appropriate physical ratios representing biogeographic features. The physical ratios are based on reef area, shelf area, coastline length and other values. The ratios were assembled by the BHS EBM project from field studies, such as the coastal rural appraisal

[40] This appendix is an excerpt from technical reports that describe in great detail the Ecopath and Ecosim models of the Raja Ampat coral reef ecosystem: Ainsworth et al. 2007 and Ainsworth et al. 2008.

surveys (Muljadi, 2004), and from data collections such as nautical charts (TNI AL – Dishidros, 2002). Biomasses are determined for certain piscivorous and herbivorous reef fish species based on the results from dive and snorkel transects conducted on Kofiau and Misool Islands by the reef health monitoring program.

Reef area ratio

Biomass for coral groups (azooxanthellate corals, hermatypic scleractinian corals, non-reef building scleractinian corals and soft corals) were assumed to vary between Kofiau, Weigeo and Misool Island study areas in direct proportion to the relative areas covered by hard coral. The area of hard coral coverage is calculated from various sources, including recent BHS EBM reef health monitoring data (Table E-1). The biomass density of these coral groups is therefore based on the larger Raja Ampat model, and modified for each sub-area by a weighting factor that adjusts for the relative coverage. The coverage of hard coral in Raja Ampat by area (32.8 %) is relatively greater than Kofiau Island (27.7 %) and relatively less than Waigeo (37.5%) and Misool (37.9%) Islands. Biomass density of coral groups is therefore adjusted down for Kofiau (i.e., by a factor of 27.7 / 32.8) and up for Waigeo and Misool. Reef health monitoring data was assembled by Andreas Muljadi (Kofiau Is.), Mohammad Syakir (SE Misool Is.) (TNC-CTC. Jl Gunung Merapi No. 38, Kampung Baru, Sorong, Papua, Indonesia 98413. Email: amuljadi@tng.org and msyakir@tnc.org. *Unpublished data*). Reef health monitoring data was collected for Waigeo Is. by M. Erdi Lazuardi (CI. Jl.Gunung Arfak.45.Sorong, Papua, Indonesia. Email: erdi@conservation.or.id) but was not available at the time of this report.

Appendix Table E-1 Hard coral coverage reported for Raja Ampat

Area	Source	Average (%)	SD	# sites
Weigeo Is.	McKenna et al. (2002)	28.5	14.8	44
Weigeo Is.	COREMAP (2001)	45.2	11.9	8
Weigeo Is.	COREMAP (2005)	38.9	32.5	35
Weigeo Is.	Donnelly et al. (2003)	37.2	21.6	25
Weigeo average		37.5		
Kofiau Is.	A. Muljadi (unpublished data) BHS EBM reef health monitoring	25.3	16.4	450
Kofiau Is.	Donelly et al. (2003)	30.0	22.4	35
Kofiau average		27.7		
Misool Is.	M. Syakir (unpublished data) BHS EBM reef health monitoring	45.9	14.4	53
Misool Is.	Donelly et al. (2003)	30.0	22.4	11
Misool average		37.9		
Ave. Raja Ampat	Donnelly et al. (2003)	32.8	22.9	94
Indonesia	Spalding et al. (2001)	1.8	-	-

Mangrove area ratio

The source of the mangrove area data is from LandSat imagry (2000-2002) (NASA Landsat Program, 2006), and it was summarized into GIS format by Mohammad Barmawi (TNC-CTC. Jl Pengembak 2, Sanur, Bali, Indonesia. *Unpublished data.* Contact: joanne_wilson@tnc.org). The relative area is presented in Table E-2.

Appendix Table E-2 Area occupied by mangroves

Area	Mangrove area (km^2)	Total area (km^2)	Relative mangrove coverage (%)	Source
Weigeo	46.6	6101	0.76	Barmawi, M. (unpublished data)
Kofiau	31.5	2391	1.32	Barmawi, M. (unpublished data)
Misool	35.1	4273	0.82	Barmawi, M. (unpublished data)
Raja Ampat	455.2	45000	1.01	Barmawi, M. (unpublished data)
Indonesia	42550.0	2915000	1.46	Spalding et al. 2001

Coastline ratio

Coastline was calculated by Mohammad Barmawi (TNC-CTC. Jl Pengembak 2, Sanur, Bali, Indonesia. *Unpublished data.* Contact: joanne_wilson@tnc.org) from LandSat images (NASA Landsat Program, 2006). The coastline for the areas in Raja Ampat are provided in Table E-3.

Appendix Table E-3 Perimeter of coastline

Area	Coastline (km)	Source
Weigeo	673	Barmawi, M. (unpublished data)
Kofiau	393	Barmawi, M. (unpublished data)
Misool	655	Barmawi, M. (unpublished data)
Raja Ampat	4261	Barmawi, M. (unpublished data)
Indonesia	95181	Spalding et al. 2001

Shelf area ratio

Bathymetry was determined using nautical charts held by the Indonesian Navy (TNI AL, 2002) and summarized into GIS format by Mohammad Barmawi (TNC-CTC. Jl Pengembak 2, Sanur, Bali, Indonesia. *Unpublished data.* Contact: joanne_wilson@tnc.org). The relative area is presented in Table E-4.

Appendix Table E-4 Area < 200 m depth

Area	Shallow area <200 m (%)	Deep area < 200 m (%)	Source
Weigeo	38.9	61.1	Barmawi, M. (unpublished data)
Kofiau	16.6	83.4	Barmawi, M. (unpublished data)
Misool	70.8	29.2	Barmawi, M. (unpublished data)
Raja Ampat	58.2	41.8	Barmawi, M. (unpublished data)
Indonesia	63.4	36.6	Spalding et al. (2001)

Parameter estimation: Catch matrices

The catch matrices for the three sub-area models were calculated based on three assumptions: The three areas Kofiau, Misool and Dampier St. contribute 70% of the catch from Raja Ampat. The catch in each sub-area model is proportional to the biomass density of species groups, the fishermen population density and the area of the models. The population density can be used to approximate fishermen density.

A value of 70% was assumed based on the fact that the sub area model for Koifiau accounts for all the Kofiau and nearby areas, the model for Misool is located in SE Misool, almost all the fishery is also concentrated in SE Misool. The catch that is not included is the catch from all parts of Waigeo other than Dampier strait and the catch by fishermen from Sorong. The biomass density of the species was calculated based on the results from the reef health monitoring and the area of the habitats available in the sub area models for the different species groups.

The population density was used to approximate the fishermen density. This population density was obtained from Jacinta and Imbir (2007). The population density estimates are as follows: Kofiau 0.9, Dampier St. 1.11 and Misool 1.08 persons·km^{-2} of model area. The fishermen density estimates from the same source were: Kofiau 0.005, Dampier St. 0.22 and Misool 0.08 persons·km^{-2} of model area. Firman and Azhar (2006) give the following estimates for the three areas respectively: 0.9, 0.88 and 1.88 persons·km^{-2} and 0.49, 0.46 and 0.98 men·km^{-2}. The statistics bureau (BPS) provides: Kofiau 0.005, Dampier St. 0.22 and Misool 0.08 persons·km^{-2} of model area. Thus there were several population estimates that we could use, we chose to use the population density from Djuang and Imbir (2007) as this seemed to be most recent and reasonable.

References

Ainsworth C. H., D. A. Varkey, and T. J. Pitcher. 2007. Ecosystem simulation models for the Bird's Head Seascape, Papua. In: Pitcher T.J., C.H. Ainsworth, and M. Bailey, editors, Ecological and economic analyses of marine ecosystems in the Birds Head Seascape, Papua, Indonesia. Fisheries Centre Research Reports **15**(5):6–172.

Ainsworth C. H., D. A. Varkey, and T. J. Pitcher. 2008. Ecosystem simulation models of Raja Ampat, Indonesia, in support of ecosystem based fisheries management. In: Bailey M., T. J. Pitcher, editors. Ecological and Economic Analyses of Marine Ecosystems in the Birds Head Seascape, Papua, Indonesia: II. Fisheries Centre Research Reports **16**(1):3-122.

COREMAP, 2001. Base Line Study Raja Ampat Islands, Papua. COREMAP Secretariate. Jl. Raden Saleh 43. Jakarta 10330. December 2001. CRITC Nasional Coremap.

COREMAP, 2005. Penelitian Terumbu Karang Tingkat Lokal, Kabupaten Raja Ampat, Provinsi Irian Jaya Barat. (In Indonesian). Sekretariat COREMAP Tahap II WB, Dinas Perikanan dan Kelautan, Kabupaten Raja Ampat, Indonesia. 75 p.

Djuang J., C. Imbir. 2007. Analisis Tingkat Kesejehteraan Masyarakat Raja Ampat. Participatory Rural Appraisal of Raja Ampat community (in Indonesian). Draft Report. Indonesia: Conservation International Indonesia. Available: Rotinsulu C. CI. Jl.Gunung Arfak.45.Sorong, Papua, Indonesia. Available: chris@conservation.or.id.

Donnelly R., D. Neville, and P. J. Mous. 2003. Report on a rapid ecological assessment of the Raja Ampat Islands, Papua, Eastern Indonesia. The Nature Conservancy Coral Triangle Center. Indonesia. Final Draft. 250p.

Firman A., I. Azhar. 2006. Atlas sumberdaya wilayah pesisir Kabupaten Raja Ampat, Provinsi Irian Jaya Barat. Conservation International Indonesia, Sorong, Indonesia. Available: chris@conservation.or.id.

McKenna S. A., G. R. Allen, and S. Suryadi (editors). 2002a. A marine rapid assessment of the Raja Ampat Islands, Papua Province, Indonesia. Conservation International, Washington, DC. RAP Bulletin of Biological Assessment **22**. 193p.

Muljadi A. 2004. Coastal rural appraisal report in Kofiau and Misool area of Raja Ampat Islands. The Nature Conservancy, South East Asia Centre for Marine Protected Areas. Sorong, Indonesia, 28p. Available: amuljadi@tng.org.

Spalding, M.D., C. Ravilious and E.P. Green, 2001, World Atlas of Coral Reefs. Prepared at the UNEP World Conservation Monitoring Centre. University of California Press, Berkeley, USA. 421p.

TNI AL – Dishidros, 2002. Tentara Nasional Indonesia Angkatan Laut - Dinas hidrologi dan Oceanografi. TNI Office: Jl. Pasir Putih. Ancol, Jakarta Indonesia.

Appendix F Probabilities Tables of the Bayesian Influence Model

Appendix Table F-1 Probability table for node restored ecosystem state
The probabilities of restored ecosystem state under different restoration scenarios and starting ecosystem states are shown. The abbreviations for each bar describe the restoration scenario (SQ Status Quo, ND No destructive fishing, NL No live fish fishing, NN No Net fishing, NS No shark fishing, C25 25% closure, C50 50% closure, C75 75% closure)

Starting Ecosystem State	Restored Ecosystem State	Ecosystem Restoration Scenarios							
		SQ	ND	NL	NN	NS	C25	C50	C75
Highly Exploited	Highly Exploited	1.000	0.999	1.000	0.890	1.000	0.824	0.650	0.028
	Medium Exploited	0.000	0.001	0.000	0.110	0.000	0.176	0.270	0.815
	Medium Restored	0.000	0.000	0.000	0.000	0.000	0.000	0.080	0.157
	Fully Restored	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Medium Exploited	Highly Exploited	0.022	0.013	0.000	0.022	0.019	0.006	0.000	0.000
	Medium Exploited	0.959	0.965	0.980	0.835	0.969	0.822	0.533	0.200
	Medium Restored	0.020	0.023	0.021	0.143	0.013	0.171	0.467	0.800
	Fully Restored	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Partially Restored	Highly Exploited	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
	Medium Exploited	0.043	0.028	0.010	0.025	0.169	0.000	0.000	0.000
	Medium Restored	0.953	0.944	0.956	0.914	0.828	0.911	0.702	0.505
	Fully Restored	0.004	0.028	0.034	0.060	0.004	0.089	0.298	0.495

Appendix Table F-2 Probability table for node fisheries catch

The probabilities of fisheries catch under different restoration scenarios and starting ecosystem states are shown. The abbreviations for each bar describe the restoration scenario (SQ Status Quo, ND No destructive fishing, NL No live fish fishing, NN No Net fishing, NS No shark fishing, C25 25% closure, C50 50% closure, C75 75% closure)

Starting Ecosystem State	Fisheries Catch	Ecosystem Restoration Scenarios							
		SQ	ND	NL	NN	NS	C25	C50	C75
Highly Exploited	50	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000
	100	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000
	200	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.8655
	300	0.0005	0.0006	0.0094	0.0167	0.0005	0.1128	0.8990	0.1337
	400	0.6300	0.6473	0.8153	0.8360	0.6394	0.8148	0.0987	0.0008
	500	0.3695	0.3521	0.1752	0.1473	0.3600	0.0724	0.0022	0.0000
Medium Exploited	50	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000
	100	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.9043
	200	0.0000	0.0000	0.0001	0.0184	0.0000	0.0380	0.9838	0.0957
	300	0.8827	0.8959	0.9249	0.9614	0.8903	0.9452	0.0161	0.0000
	400	0.1145	0.1018	0.0735	0.0199	0.1072	0.0165	0.0001	0.0000
	500	0.0027	0.0023	0.0015	0.0003	0.0025	0.0002	0.0000	0.0000
Partially Restored	50	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0499	0.9989
	100	0.9998	0.9999	1.0000	1.0000	0.9999	1.0000	0.9501	0.0011
	200	0.0002	0.0001	0.0000	0.0000	0.0001	0.0000	0.0000	0.0000
	300	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000
	400	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000
	500	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000

Appendix Table F-3 Probability table for node average price

The probabilities of average revenue under different restoration scenarios and catch levels in the ecosystem are shown. The abbreviations for each bar describe the restoration scenario (SQ Status Quo, ND No destructive fishing, NL No live fish fishing, NN No Net fishing, NS No shark fishing, C25 25% closure, C50 50% closure, C75 75% closure)

Catch	Average price	Ecosystem Restoration Scenarios							
		SQ	ND	NL	NN	NS	C25	C50	C75
50	12	0.0007	0.0007	0.0009	0.0002	0.0006	0.0016	0.0251	0.1177
	16	0.8214	0.8203	0.8369	0.7322	0.8156	0.8601	0.9189	0.8528
	20	0.1724	0.1735	0.1574	0.2575	0.1780	0.1345	0.0549	0.0290
	24	0.0053	0.0054	0.0047	0.0098	0.0056	0.0037	0.0010	0.0004
	28	0.0001	0.0001	0.0001	0.0003	0.0001	0.0001	0.0000	0.0000
	32	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000
	36	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000
100	12	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000
	16	0.0001	0.0001	0.0001	0.0000	0.0001	0.0000	0.0000	0.0000
	20	0.2633	0.2431	0.2771	0.0195	0.2516	0.0332	0.0024	0.0005
	24	0.6113	0.6225	0.6032	0.5465	0.6178	0.5969	0.3318	0.2081
	28	0.1152	0.1233	0.1101	0.3683	0.1198	0.3199	0.5162	0.5637
	32	0.0095	0.0104	0.0089	0.0598	0.0100	0.0458	0.1328	0.1975
	36	0.0006	0.0007	0.0006	0.0059	0.0007	0.0042	0.0169	0.0300
200	12	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000
	16	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0014
	20	0.0001	0.0001	0.0002	0.0000	0.0001	0.0002	0.0104	0.5251
	24	0.1084	0.1086	0.1443	0.0037	0.0937	0.1398	0.4813	0.4232
	28	0.5580	0.5581	0.5679	0.2408	0.5499	0.5673	0.4211	0.0472
	32	0.2808	0.2805	0.2455	0.5219	0.2980	0.2495	0.0788	0.0030
	36	0.0527	0.0526	0.0422	0.2337	0.0584	0.0433	0.0084	0.0002

Appendix Table F-4 Probability table for node fisheries revenue
The probabilities of fisheries revenue for combinations of catch and average revenue are shown.

Catch	Scaled Revenue	Average price						
		12	16	20	24	28	32	36
50	500	0.999	0.000	0.000	0.000	0.000	0.000	0.000
	1000	0.001	1.000	0.999	0.830	0.001	0.000	0.000
	1600	0.000	0.000	0.001	0.169	0.980	0.902	0.485
	2000	0.000	0.000	0.000	0.001	0.019	0.098	0.514
	3000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
	4000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
	5000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
	6000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
	7000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
	8000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
100	500	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000
	1000	0.8305	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000
	1600	0.1685	0.9023	0.0519	0.0000	0.0000	0.0000	0.0000
	2000	0.0010	0.0977	0.9456	0.5995	0.0006	0.0000	0.0000
	3000	0.0000	0.0000	0.0024	0.3997	0.9890	0.8866	0.2247
	4000	0.0000	0.0000	0.0000	0.0007	0.0103	0.1124	0.7553
	5000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0010	0.0198
	6000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0003
	7000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000
	8000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000
200	500	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000
	1000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000
	1600	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000
	2000	0.5995	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000
	3000	0.3997	0.8866	0.0046	0.0000	0.0000	0.0000	0.0000
	4000	0.0007	0.1124	0.8960	0.1398	0.0003	0.0000	0.0000
	5000	0.0000	0.0010	0.0970	0.7627	0.3869	0.0157	0.0001
	6000	0.0000	0.0000	0.0023	0.0932	0.5304	0.5269	0.0958
	7000	0.0000	0.0000	0.0001	0.0042	0.0768	0.3906	0.5823
	8000	0.0000	0.0000	0.0000	0.0002	0.0055	0.0668	0.3219
300	500	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000
	1000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000
	1600	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000
	2000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000
	3000	0.2247	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000
	4000	0.7553	0.1398	0.0000	0.0000	0.0000	0.0000	0.0000
	5000	0.0198	0.7627	0.1080	0.0001	0.0001	0.0001	0.0000
	6000	0.0003	0.0932	0.6648	0.0958	0.0958	0.0958	0.0000
	7000	0.0000	0.0042	0.2054	0.5823	0.5823	0.5823	0.0002
	8000	0.0000	0.0002	0.0219	0.3219	0.3219	0.3219	0.9998
400	500	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000
	1000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000
	1600	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000
	2000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000
	3000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000
	4000	0.1398	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000
	5000	0.7627	0.0157	0.0000	0.0000	0.0000	0.0000	0.0000
	6000	0.0932	0.5269	0.0036	0.0000	0.0000	0.0000	0.0000
	7000	0.0042	0.3906	0.2907	0.0085	0.0001	0.0000	0.0000
	8000	0.0002	0.0668	0.7057	0.9915	0.9999	1.0000	1.0000
500	500	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000
	1000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000
	1600	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000
	2000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000
	3000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000
	4000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000
	5000	0.1080	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000
	6000	0.6648	0.0036	0.0000	0.0000	0.0000	0.0000	0.0000
	7000	0.2054	0.2907	0.0027	0.0000	0.0000	0.0000	0.0000
	8000	0.0219	0.7057	0.9973	1.0000	1.0000	1.0000	1.0000

Appendix Table F-5 Tourism projection 'low' for restored ecosystem states

Revenue	Restored Ecosystem States			
	Highly Exploited	Medium Exploited	Medium Restored	Fully Restored
400	0.308	0.005	0.000	0.000
500	0.657	0.020	0.001	0.000
800	0.035	0.322	0.016	0.002
1200	0.000	0.616	0.165	0.021
1600	0.000	0.037	0.374	0.107
2000	0.000	0.000	0.182	0.228
2000	0.000	0.000	0.182	0.228
2200	0.000	0.000	0.071	0.241
2500	0.000	0.000	0.008	0.174

Appendix Table F-6 Ttourism projection 'high' for restored ecosystem states

Revenue	Restored Ecosystem States			
	Highly Exploited	Medium Exploited	Medium Restored	Fully Restored
1000	0.073	0.001	0.000	0.000
1500	0.503	0.010	0.001	0.000
2000	0.389	0.058	0.004	0.001
2500	0.034	0.199	0.016	0.003
3000	0.000	0.398	0.048	0.009
4000	0.000	0.308	0.222	0.051
5000	0.000	0.027	0.388	0.177
6000	0.000	0.000	0.257	0.352
7000	0.000	0.000	0.064	0.407

Appendix Table F-7 Benefits for conservation modeled as WTP

Revenue	Restored Ecosystem States			
	Highly Exploited	Medium Exploited	Medium Restored	Fully Restored
6	0.004	0.000	0.000	0.000
9	0.919	0.000	0.000	0.000
11	0.078	0.000	0.000	0.000
20	0.000	0.000	0.000	0.000
30	0.000	0.000	0.000	0.000
40	0.000	0.037	0.000	0.000
50	0.000	0.646	0.008	0.000
60	0.000	0.317	0.243	0.000
80	0.000	0.000	0.748	0.064
100	0.000	0.000	0.001	0.739
120	0.000	0.000	0.000	0.197

Appendix Table F-8 Benefits of conservation modeled as ES

Revenue	Restored Ecosystem States			
	Highly Exploited	Medium Exploited	Medium Restored	Fully Restored
2000	0.114	0.000	0.000	0.000
3000	0.886	0.000	0.000	0.000
4000	0.000	0.000	0.000	0.000
6000	0.000	0.000	0.000	0.000
8000	0.000	0.000	0.000	0.000
10000	0.000	0.003	0.000	0.000
12000	0.000	0.120	0.000	0.000
16000	0.000	0.871	0.060	0.000
20000	0.000	0.006	0.731	0.007
24000	0.000	0.000	0.209	0.186
30000	0.000	0.000	0.000	0.808

Appendix Table F-9 Probability tables for tourism revenue and conservation benefit combined

Revenue	Restored Ecosystem States			
	Highly Exploited	Medium Exploited	Medium Restored	Fully Restored
Conservation WTP Tourism Low				
300	0.032	0.000	0.000	0.000
400	0.172	0.002	0.000	0.000
500	0.390	0.007	0.001	0.000
600	0.378	0.023	0.001	0.000
800	0.028	0.135	0.009	0.002
1000	0.000	0.344	0.039	0.007
1200	0.000	0.387	0.114	0.022
1500	0.000	0.098	0.288	0.089
1800	0.000	0.004	0.318	0.226
2000	0.000	0.000	0.214	0.326
2500	0.000	0.000	0.016	0.328
Conservation WTP Tourism Low				
1000	0.058	0.001	0.000	0.000
1200	0.166	0.001	0.000	0.000
1500	0.413	0.006	0.001	0.000
2000	0.333	0.035	0.003	0.001
2500	0.030	0.125	0.012	0.002
3000	0.000	0.262	0.039	0.007
3500	0.000	0.319	0.096	0.020
4000	0.000	0.228	0.187	0.046
5000	0.000	0.023	0.347	0.164
6000	0.000	0.000	0.248	0.342
7000	0.000	0.000	0.068	0.417

Revenue	Restored Ecosystem States			
	Highly Exploited	Medium Exploited	Medium Restored	Fully Restored
Conservation ES Tourism Low				
3000	1.000	0.000	0.000	0.000
5000	0.000	0.000	0.000	0.000
8000	0.000	0.000	0.000	0.000
10000	0.000	0.002	0.000	0.000
14000	0.000	0.428	0.001	0.000
18000	0.000	0.567	0.117	0.000
22000	0.000	0.004	0.576	0.013
25000	0.000	0.000	0.302	0.104
30000	0.000	0.000	0.003	0.553
35000	0.000	0.000	0.000	0.311
40000	0.000	0.000	0.000	0.019
Conservation ES Tourism Low				
3000	0.137	0.000	0.000	0.000
5000	0.863	0.000	0.000	0.000
8000	0.000	0.000	0.000	0.000
10000	0.000	0.000	0.000	0.000
14000	0.000	0.086	0.000	0.000
18000	0.000	0.701	0.023	0.000
22000	0.000	0.204	0.254	0.003
25000	0.000	0.009	0.514	0.019
30000	0.000	0.000	0.203	0.192
35000	0.000	0.000	0.006	0.482
40000	0.000	0.000	0.000	0.304

Appendix G   Previously Published Work Related to the Thesis

List of technical reports

1. Ainsworth, C. H., **D. A. Varkey,** and T. J. Pitcher. 2008. Ecosystem simulation models of Raja Ampat, Indonesia, in support of ecosystem based fisheries management. Pages 3–122 in Bailey, M., T. J. Pitcher, (eds) (2007) Ecological and economic analyses of marine ecosystems in the Birds Head Seascape, Papua, Indonesia: II. Fisheries Centre Research Reports **16**(1): 186 p. Available: < http://www2.fisheries.com/archive/publications/reports/report16_1.php >.
2. Ainsworth, C. H., **D. A. Varkey,** and T. J. Pitcher. 2007. Ecosystem simulation models for the Bird's Head Seascape, Papua. Pages 6–172 in Pitcher T.J., C.H. Ainsworth, and M. Bailey, (eds) (2007) Ecological and economic analyses of marine ecosystems in the Birds Head Seascape, Papua, Indonesia: I. Fisheries Centre Research Reports **15**(5): 182 p. Available: <http://www2.fisheries.com/archive/publications/reports/report15_5.php>
3. **Varkey, D. A.**, G. Pramod, and T. J. Pitcher. 2006. Estimating compliance with FAO Code of Conduct Article 7, fisheries management in New Zealand. 30 p. In: Pitcher T.J., D. Kalikoski, and G. Pramod. Evaluations of compliance with the FAO (UN) Code of Conduct for Responsible Fisheries. Fisheries Centre Research Reports **14**(2) 1191 p. Available: <http://www2.fisheries.com/archive/publications/reports/report14_2.php>
4. Kumar, R., **D. A. Varkey.** 2009. Ecosystem food web using SAS graph software. SAS Conference Proceedings: Pacific North-West SAS Users Group Conference. Portland, Oregon, 28-29th September, 2009. Available: <http://www.lexjansen.com/pnwsug/2009/Kumar,%20Rajeev%20and%20Divya

%20Varkey%20-
%20Ecosystem%20Food%20Web%20Constellation%20Diagram.pdf>

Reprints of published papers

1. Ainsworth, C.H., **D. A. Varkey,** and T. J Pitcher. 2008. Ecosystem simulations supporting ecosystem-based management in the coral triangle, Indonesia. Ecological Modeling **214**: 361-374.
2. Pitcher. T. J., D. Kalikoski, K. Short, **D. A. Varkey,** G. Pramod. 2008. An evaluation of progress in implementing ecosystem-based management of fisheries in 33 countries. Marine Policy **33**(2): 223-232.

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Ecosystem simulations supporting ecosystem-based fisheries management in the Coral Triangle, Indonesia

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ABSTRACT

A comprehensive field study ongoing in Eastern Indonesia has provided data for a trophodynamic Ecosim ecosystem model of the Raja Ampat archipelago on the west coast of New Guinea. Model dynamics have been tuned to agree with local catch and relative biomass time series data developed for this project, and validated by experts. The model is used in this article to investigate five high priority research questions related to ecosystem-based fisheries management (EBFM) in the region. Regency fisheries managers and scientific partners working in Indonesia posed the questions. Here, we analyze the ecosystem impacts of blast fishing, including trophic effects and removal of refuge space. Removal of refuge space is as harmful to juvenile reef fish populations as direct mortality from the fishery itself; reef damage is cumulative due to the slow re-growth rate of corals. We quantify the likely ecological and economic impacts of limiting commercial fisheries for groupers. Artisanal fisheries benefit slightly and system biodiversity is improved, but the improvement is lost if artisanal fisheries increase effort to compensate for missing catch. We forecast the effects of limiting commercial net fisheries for reef fish. There is a marginal increase in reef fish biomass and an unexpected benefit to large pelagic species due to reduced interception of their anchovy prey. We evaluate the exploitation status of anchovy and tuna and report on the ecosystem effects of these fisheries. All fisheries appear fully exploited in Raja Ampat or nearly so. There is an indication that anchovies provide an ecosystem service: a large population may act to buffer fluctuations in large pelagic fish biomass under climate variation.

1. Introduction

1.1. *Raja Ampat*

Raja Ampat is located in the centre of the Southeast Asia 'coral triangle'; it contains some of the most biodiverse coral reef ecosystems ever recorded (Donnelly et al., 2003; McKenna et al., 2002) and possesses over 75% of the world's known coral species (Halim and Mous, 2006). This remote island group in Eastern Indonesia faces challenges, however, like other coral reef areas in the world, relating to overexploitation of fishery resources (Pandolfi et al., 2003), destructive fishing practices (Erdmann and Pet-Soede, 1996; Pet-Soede and Erdmann, 1998), land-based pollution (Kaczmarsky et al., 2005) and outbreaks of coralivores such as the crown of thorns starfish (*Acanthaster planci*)—a source of mass mortality in corals (Chesher, 1969). There have been fishery depletions in Raja Ampat across a wide range of taxa but many examples of pristine coral reef wilderness remain thanks in part to a low, although increasing, human population (McKenna et al., 2002; Dohar and Anggraeni, 2007).

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1.2. BHS EBM Project

The Raja Ampat Regency government has developed initiatives to protect the marine environment and serve as many as 24,000 commercial and artisanal fishers who rely on it (Dohar and Anggraeni, 2007). For example, a decree by the Bupati (Regent) in 2003 declared Raja Ampat to be a Maritime Regency and helped to establish a new network of marine reserves in 2006 covering more than 650,000 ha of sea area and 44% of reef area. The fisheries office (Departemen Kelautan dan Perikanan, DKP) has further pledged to declare as much as 30% of the marine area protected in the Regency, exceeding the national goal of 20% (Rahawarin, personal communication, 2007).

Recently, the Regency government entered into a comprehensive program of field study and ecological modelling with The Nature Conservancy (TNC), Conservation International (CI), World Wildlife Fund (WWF) and the University of British Columbia (UBC) with the goals of increasing the body of scientific knowledge in this region, exploring marine protected area (MPA) zoning options and supporting ecosystem-based fishery management (EBFM). This article summarizes the results of a 2-year effort to integrate field data into a synthesis food-web model, built in the Ecopath with Ecosim modelling framework (EwE: Christensen and Pauly, 1992; Walters et al., 1997). We use the model to explore the trophodynamic impacts of management policies in an EBFM framework.

Trophic ecosystem models provide a valuable tool to help us understand the ecology of coral reefs. They offer a framework on which we can integrate data from a variety of sources, evaluate the commensurability of the data, find commonalities and identify gaps in knowledge. We can determine the controlling trophic linkages and predict the food web's major response to management plans; EwE has been applied throughout the world for this purpose (Christensen and Walters, 2005). The approach allows us to tease apart the interacting influences of fisheries and frame the activities of human beings in a whole-ecosystem context. The modelling lends itself to the use of quantitative ecosystem indicators that can provide a synoptic view of the ecosystem and so make model predictions more interpretable by scientists, stakeholders and the public.

1.3. EBFM research questions

We collaborated with researchers in partner institutes (TNC, CI, WWF, UBC and Packard Foundation), and we received input from DKP in order to develop five high priority research questions of relevance to the implementation of EBFM in Raja Ampat. They are listed here. Section 2 describes how we addressed these questions using the EwE model.

(1) What are the likely ecosystem effects of changes in the anchovy fishery under the following management scenarios?
  - Anchovy fishery is completely removed from Raja Ampat.
  - A limited anchovy fishery is allowed.
  - Anchovy fishery continues to increase in size.

(2) What are the likely effects of restricting the commercial exploitation of groupers?
(3) What are the likely effects of excluding all net fisheries for reef fish in Raja Ampat?
(4) What are the likely effects of blast fishing under the following scenarios?
  - Status quo.
  - Increase.

(5) What are the likely effects of an increase in the tuna fishery?

There is a wide range of interpretations used by fishery management bodies throughout the world as to what constitutes EBFM (e.g., Marasco et al., 2007), but all include the ecosystem-wide consequences of alterations in biomasses and trophic pathways. Accordingly, our analysis highlights changes in trophic interactions that may occur as a result of alternative management plans. This is acknowledged to be one of the strengths of the Ecopath with Ecosim (EwE) modelling approach (Plagányi, 2007).

2. Methods

2.1. Ecopath with Ecosim

We have used EwE ecosystem simulation software to represent the ecology of the coral reef environment (EwE: Christensen and Pauly, 1992; Walters et al., 1997). EwE is a mass-balance trophic simulator that acts as a thermodynamic accounting system. It summarizes biomass or energy flows in and out of functional groups, which are groups of similar species aggregated by life history and niche characteristics, and apportions the production of groups to predators and fisheries according to diet and catch matrices. The model can represent the impacts of human beings through fishery removals and incidental mortality (Christensen et al., 2004), habitat modification (this manuscript) and pollution (Okey et al., 2004). It can represent abiotic effects such as climate fluctuations through the use of production forcing patterns, and it can represent animal behaviour through interaction rate parameters and density-dependant mediation functions (Christensen et al., 2004). Reviews of EwE are found in Fulton et al. (2003), Christensen and Walters (2004, 2005), Plagányi and Butterworth (2004) and Plagányi (2007).

For parameterization of the ecosystem models we refer the readers to online technical publications by Ainsworth et al. (2007), who detailed the development of preliminary models for Raja Ampat, and Ainsworth et al. (2008), who updated the models, re-tuned dynamics to fishery catch histories and relative biomass, and included extensive field-gathered data. Data used in the models from the BHS EBM Project includes information from dive transects, fish stomach sampling, community interviews, coastal surveys, published literature and other sources; the models have also been reviewed by biologists in Indonesia. The technical reports describe a suite of ecosystem models representing Raja Ampat in the past and present (1990 and 2005), and smaller sub-areas in the archipelago. This article uses the present-day model of Raja Ampat for all predictions.

The Raja Ampat model describes the region from 129°12′E and 0°12′N to 131°30′E and 2°42′S off the west coast of New Guinea in the Indonesian province of Papua. The study area encompasses approximately 610 islands, including the 'four kings': Batanta, Misool, Salawati and Waigeo. The model uses 98 functional groups to represent the ecosystem, of which 31 are reef fish groups and 25 are pelagic or demersal fish groups (representing 1203 fish species altogether); these groups include juvenile and sub-adult age stanzas. A further 22 groups are macroinvertebrates, 9 are mammals, turtles or birds and the remainder are plankton, primary producers and detritus groups. There are more than 2400 diet linkages. The group design highlights commercially important species (e.g., groupers, Napoleon wrasse *Cheilinus undulatus* and tuna), and ecologically important species such as herbivores, bioeroders (e.g., parrotfish) and corallivores (e.g., crown-of-thorns starfish *Acanthaster planci*). Trophic vulnerabilities, which are critical parameters in the dynamic model Ecosim describing risk-adverse feeding behaviour, were tuned to observational data in the 1990 model and transferred to the 2005 model using a novel technique that assumes stationarity in density-dependent foraging tactics (Ainsworth et al., 2008).

The Ecosim model uses four mediation functions to represent the following non-trophic behavioural effects: (1) pelagic piscivores corralling small pelagics to the surface facilitating predation by sea birds, (2) hermatypic scleractinian corals conferring protection against predators to juvenile reef fish and some invertebrates, (3) mangroves and sea grass providing similar protection and (4) cleaner wrasse improving the health of large reef-associated fish reducing their vulnerability to predators. The first two mediation functions are relevant to experiments in this article, and are elaborated on later.

The following section will discuss the five research questions in more detail, and introduce modifications that have been designed in order to answer these questions.

2.2. *(Q1) Effects of the anchovy fishery*

Located mainly in coastal areas adjacent to the central upwelling region of Dampier St., a large anchovy fishery is conducted in Raja Ampat by a mobile lift net fleet (called bagans) in which paired vessels use lanterns at night to attract fish. The fish, called *ikan tri* locally, are dried on racks in large numbers. There are 17 Engaulids identified in Raja Ampat of the genera Stolephorus, Setipinna and Thryssa; Stolephorus is the dominant genus by biomass in shallow habitats and especially important is *S. indicus* (Erdmann, personal communication, 2007). All of these genera have been represented in a single aggregated EwE functional group called 'anchovies'. The functional group is divided into juvenile and adult stanzas using a multi-species equilibrium model (Christensen et al., 2004); ages are partitioned according to maturation and growth parameters assembled by Ainsworth et al. (2007).

Amarumollo and Farid (2002) suggested that there might have been a decline in anchovy abundance in Raja Ampat due to overfishing, although Erdmann and Pet (2002) pointed out uncertainties in their conclusion. Since most of the catch is trans-shipped at sea for sale in Java (Bailey, personal communication, 2007), catch figures are not recorded in Regency statistics or anywhere else. It is therefore difficult to estimate the tonnage being caught. However, Bailey et al. (2008) used a technique based on interviews and counts of drying racks to estimate annual catch of anchovy as 13,000 tonnes in Raja Ampat. We have used a higher value, 0.5 tonnes km^{-2} for the adult stanza in Raja Ampat, which equates to about 22,500 tonnes, based on the estimate of Ainsworth et al. (2007).

We use Ecosim simulations to determine the level of catch and biomass of anchovies at equilibrium under a range of fishing mortalities. This produces a multi-species surplus production curve upon which we can plot the current exploitation status (we discuss multi-species equilibrium curves in more detail under 'maximization of tuna fisheries').

As anchovy biomass increases or declines we can analyze biomass changes in other functional groups of the ecosystem, as may be affected through direct trophic effects like predation, or indirect effects like competition. The changing biomasses of other functional groups may have positive or negative net effects on other fishing sectors. We therefore perform a simple bioeconomic analysis, holding the fishing rate per functional group constant at 2005 levels for other fishing sectors and determining the impact on their fishery values that result from the changing ecosystem biomass densities.

To further investigate trophic effects of anchovy depletion we have employed a network analysis routine in Ecopath called mixed trophic impacts (MTI). MTI is a form of sensitivity analysis that summarizes the net impact of functional groups and fisheries on each other. It considers direct and indirect trophic interactions caused by predation and competition. The routine is based on the Leontief matrix (Leontief, 1951), and was applied to Ecosim by Ulanowicz and Puccia (1990). Christensen et al. (2004) provide more detail.

The anchovy population in Raja Ampat is fertilized by periodic wind-driven upwelling events in the central study region and throughout the archipelago (Erdmann, personal communication, 2007). Fluctuations in primary production may affect the anchovy population with secondary effects that cascade throughout the food web (Beamish, 1995; McFarlane et al., 2000; Chassot et al., 2007). We therefore introduce stochastic climate effects into the dynamic simulation in the form of a randomized annual primary production forcing function in order to test whether a depressed anchovy population might impact the variability of pelagic piscivore populations. This is similar to a population viability analysis conducted by Pitcher et al. (2005), but uses a novel technique to scale the projected primary production trend to historical data.

The randomized climate series is created by sampling with replacement of a primary production anomaly trend estimated by Ainsworth et al. (2008) using a historic 1990 EwE model. Those authors used an automated optimization routine in Ecosim to reconstruct the primary production pattern from 1990 to 2005. The pattern represents annual modifiers of the phytoplankton production rate (production/biomass, units: year^{-1}) that would best explain residuals between predicted and observed time series of catch and relative biomass when used as a forcing function in the 1990–2005 simulation. The residuals for 32 functional groups were minimized using a least-squares fitting criterion, assuming that the production pattern cascades up the food web to affect higher order groups.

The amplitude of the primary production anomaly pattern was subsequently scaled by Ainsworth et al. (2008) to recreate the observed phytoplankton variability, about 4.7% per year, as measured by SeaWifs satellite data (SAU, 2006). The scaling procedure ensures that the degree of primary production variability is representative of the past. In testing the effects of stochastic climate in this way we also assume that past variability in primary production might be similar to future variability.

There are several major sources of process and model uncertainty that influence the predicted dynamics of pelagic piscivores. The unsure biomass estimates of tuna, anchovy and other indirect players contribute to the uncertainty, but the greatest source of error may relate to the predation mortality functional response described by Ecosim's vulnerability parameters. Uncertainty in these terms reflects our imperfect knowledge of density-dependant predator and prey behaviour, and also the carrying capacity of the ecosystem. Ainsworth et al. (2008) document the process of tuning the model to observed time series in order to provide a plausible set of vulnerability parameters for this ecosystem.

2.3. *(Q2) Restricting commercial fisheries on groupers*

In order to determine the effects of restricting commercial fisheries for groupers (Serranidae), we have assumed that the following gear types are used primarily for the commercial exploitation of groupers (number in parenthesis indicates EwE fleet identification number in Ainsworth et al., 2007): diving with spear (7), diving for live fish (8), diving with cyanide (9) and blast fishing (10). The other gear types exploiting groupers are therefore assumed to be primarily artisanal: spear and harpoon (1), permanent trap (5) and set line (14). The distinction between artisanal and commercial catch is difficult to draw due to the unreported and unregulated nature of Raja Ampat reef fish fisheries and widespread casual local trade. A significant portion of the catch that we here call artisanal may be sold locally or traded within islands and villages, but we have chosen these gear types to highlight the distinction between fishing sectors that require low capital investment and/or whose products are destined for a small-scale local market, versus fishing sectors that require high capital investment and/or whose products are destined for regional or international market. For example, capital-intensive fishing methods such as compressor diving are assumed to be commercial; also fisheries that produce a high value product suitable for export such as cyanide fishing to capture groupers, Napoleon wrasse (*Cheilinus undulatus*) and other live reef fish (Erdmann and Pet-Soede, 1996; Pet-Soede and Erdmann, 1998). Blast fishing provides a high yield of low-value product, which is likely to be absorbed by a large regional market, and so is assumed to be commercial.

In the simulation model, we reduce the amount of commercial fishing on groupers and examine the effect on the artisanal gear types. We look at two scenarios, where artisanal fishing effort remains constant, and when it increases to keep the total amount of grouper catch constant. Effects of restricting commercial fisheries are framed in terms of the impact to artisanal fisheries production and assemblage changes caused by trophic dynamics.

2.4. *(Q3) Restrictions to reef fish net fisheries*

There are concerns that the livelihoods of artisanal fishers in Raja Ampat, particularly around the rural communities of Weigeo, Kofiau and Misool Islands, might be threatened by the commercial exploitation of reef-associated fish. Some commercial fishing is done by foreigners originating in urban centres like Sorong on the west coast of New Guinea. As a potential management tool, we evaluate the ecosystem impacts of restricting net fisheries that operate in inshore areas around Raja Ampat by varying the model net fisheries (shore gillnet (3) and driftnet (4)) and examine a range of scenarios including total closure. Non-net gear types that capture reef fish are spear and harpoon (1), reef gleaning (2), permanent trap (5), portable trap (6), diving with spear (7) and diving for live fish (8). We perform similar tests of the model as the ones described above for groupers.

2.5. *(Q4) Effects of blast fishing*

Blast fishing is a dangerous and damaging fishery that has long-term impacts on the health of coral reef communities (Pauly et al., 1989). It has been illegal in Indonesia since 1985 when Fisheries Law No. 9 came into effect legislating acceptable harvest areas and catching methods, and introducing some protection against pollution and other damages (Purwaka and Sunoto, 2002). Penalties for blast fishing are strict in Indonesia, and can include up to 10 years in prison and/or a 100 million Rupiah fine (more than $10,000) but enforcement and monitoring are seriously lacking (Donnelly et al., 2003).

Blast fishing is known to occur in Raja Ampat, however, estimates of the frequency vary widely (McKenna et al., 2002; Erdmann and Pet, 2002; Donnelly et al., 2003). Aerial surveys in 2006 for the BHS-EBM Project did not detect any active operations (Barmawi, personal communication, 2007), nevertheless it is perceived by locals to be a serious threat to ecosystem health (Muljadi, 2004; Halim, 2007). The blasting operations provide a high quantity of low-value fish. Fish caught with this method are easily identified in markets by their pulverized flesh and low value. Nevertheless, there is a strong economic incentive for fishers to continue the practice (Bailey, 2007) despite the fact that the cumulative effect of blast fishing greatly reduces the profitability of the reef system in the long term (Pet-Soede et al., 1999).

Simulating the effects of removals from blast fishing is straightforward in the EwE models; we can capture the impacts on both targeted and bycatch species as we can with most gear types. However, to accurately simulate non-trophic effects caused by the removal of complex substrate requires the use of special mediation functions. With these, we can attempt to capture the refuge benefit that juvenile fish and invertebrates gain from high coral biomass. The Raja Ampat model uses mediation functions to emulate this benefit, where the vulnerability of juvenile fish and invertebrates to their predators increases as the biomass of coral is reduced.

We used a simple linear mediation function. As the biomass of corals decreases, the trophic vulnerabilities of the mediated groups increase to a maximum of two times the baseline Ecopath vulnerability value. The vulnerability value represents the maximum allowable increase in the predation mortality for a given predator–prey interaction when predator biomass is high. Therefore, as modelled, if coral biomass were to decrease to zero, predation mortality on the mediated groups would have the potential to double under conditions of high predator biomass. Using a higher maximum than two has almost no effect under the conditions of the present test. The mediation functions affect all predators in the same way, regardless of attack mode. Mediation functions were applied to 27 reef fish functional groups, including all juvenile age stanzas in the model, and two invertebrate groups: small crabs and octopus. For the purpose of this analysis, we are only concerned with juvenile stanzas of the following 12 reef fish groups: groupers, snappers, Napoleon wrasse, coral trout, large/medium/small reef-associated, large/small planktivores, macro algal browsing, eroding grazers and scraping grazers. For functional group descriptions consult Ainsworth et al. (2008).

Some other applications of mediation functions in EwE are described by Okey et al. (2004) (sea floor shading by plankton blooms) and Cox et al. (2002) (tunas mediating forage fish mortality caused by birds). We frame ecosystem results from blast fishing in terms of the assemblage change at various levels of fishing effort, and the effect on juvenile fish biomass resulting from a loss in coral reef habitat.

2.6. (Q5) Maximization of tuna fisheries

In order to determine the resource potential of tuna stocks we have constructed multi-species catch and biomass equilibrium curves for skipjack tuna (*Katsuwonus pelamis*) and 'other tuna' (including 10 other species of Scombridae). The equilibrium analysis produces outputs analogous to biomass dynamic models commonly used in single-species fisheries management. The biomass of an exploited group will usually be highest under zero fishing effort (the catch then will also be zero); this biomass level is referred as B_0, or virgin biomass. As fishing intensity increases, catch on the subject functional group will increase to a maximum called maximum sustainable yield (MSY: Russell, 1931; Graham, 1935). When fisheries take exactly this amount, the biomass at maximum sustainable yield (B_{MSY}) can be maintained indefinitely (in principle, with caveats). However, when catches exceed this amount, recruitment overfishing occurs and catches will be sub-optimal.

As with analogous single-species methods, the equilibrium analysis relies on the assumption of deterministic population behaviour in growth, recruitment and mortality, and so is subject to similar criticisms as biomass dynamic models (see Larkin, 1977; Punt and Smith, 2001). Climate variation, for example, can only reduce the estimate of safe harvest limits. However, an equilibrium analysis using an ecosystem model offers a major advantage over single-species methods because it accounts for species interactions. Even though an ecosystem model represents a greatly simplified and abstract version of the ecosystem, the number of trophic and non-trophic interactions increases exponentially with the number of functional groups. These interactions can combine in unexpected ways to affect stock dynamics. Multispecies production curves can potentially differ drastically compared to a single-species assessment, and it is prudent to analyze sources of the discrepancy in an EBFM framework.

We also develop an optimal tuna fishing solution by use of the policy search routine in Ecosim (Christensen and Walters, 2004). The routine uses a non-linear optimization procedure to iteratively step towards the vector of fishing fleet effort that maximizes an objective function. The solution we present maximizes the sustainable amount of catch available from tuna groups by maneuvering the ecosystem into a hyper-productive form, eliminating predators and competitors, while enhancing prey and supporting groups. By engineering the ecosystem to support high tuna yields we can exceed the single-species sustainable catch limits predicted by MSY. It is important to note that this does not represent a reasonable policy option. Under this extreme scenario, the ecosystem is simplified, biodiversity is reduced and the risk of catastrophic stock failure may also be increased. Such a policy would also be socially undesirable: the increased catch rate could only be achieved in principle through the cooperation of fishing sectors, some of which must sacrifice earnings to service the tuna fleet. This solution can be viewed as a theoretical 'upper-limit' on ecosystem production as set by thermodynamics.

Among other modelling uncertainties, Ecosim has an inherent limitation in representing migratory functional groups like tuna (Martell, 2004). The default assumption we make here is that predation and fishing mortality trends elsewhere it its range (i.e., outside of Raja Ampat) mirror the changes detailed in the model.

3. Results

3.1. (Q1) Effects of the anchovy fishery

Fig. 1 shows the ecosystem effects of Raja Ampat anchovy fisheries at levels of fishing mortality above and below the baseline 2005 year ($F_{2005} \approx 0.5\,\text{year}^{-1}$). All values represent the equilibrium ecosystem condition reached after a 20-year simulation from 2005 to 2025. Each data series represents changes in the aggregate biomass of EwE functional groups shown in the figure. As the anchovy fishery increases to four times its current level of fishing mortality, anchovy biomass is reduced to approximately 60% of its current value (see Fig. 2) and many of their predators decline. The biomass of large, medium and small reef-associated fish increase slightly, no more than 3%, due to a release of predation mortality from pelagic piscivores. However, the biomass of large, medium and small pelagic fish, which feed on anchovy, is decreased by as much as 80%. The large pelagic group includes skipjack tuna and other tuna, mackerel and billfish: these are reduced to 53%, 36%, 66% and 19% of their current biomass levels, respectively, when the anchovy fishing rate is increased to four times the 2005 level. Seabirds are among the predators that suffer the most from a loss of anchovy biomass; a 30% decline in the biomass of birds seems likely under these conditions.

The multi-species equilibrium analysis presented in Fig. 2 suggests that anchovies are not currently overexploited by humans in Raja Ampat. Although there are large uncertainties associated with the estimate, catch from fisheries accounts for

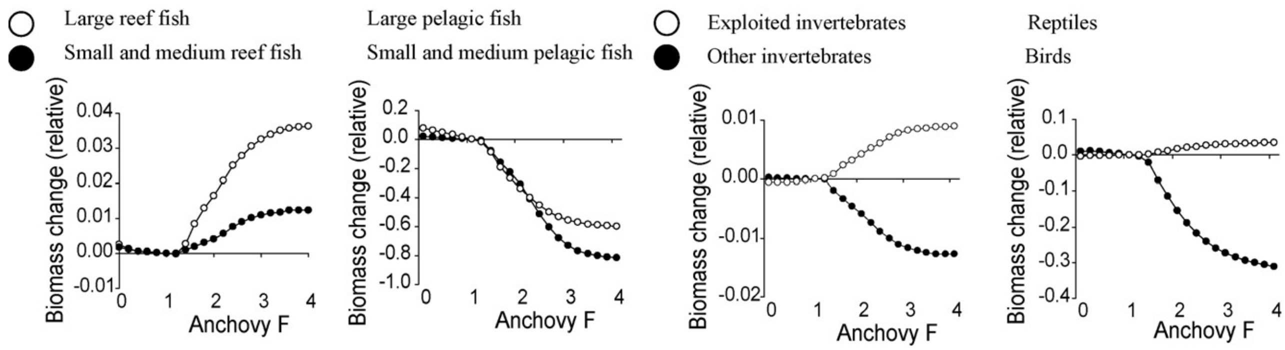

Fig. 1 – Ecosystem biomass changes with increasing anchovy exploitation. The biomasses of all relevant EwE functional groups have been combined into these generic categories. *X*-axis shows anchovy fishing mortality relative to 2005 levels.

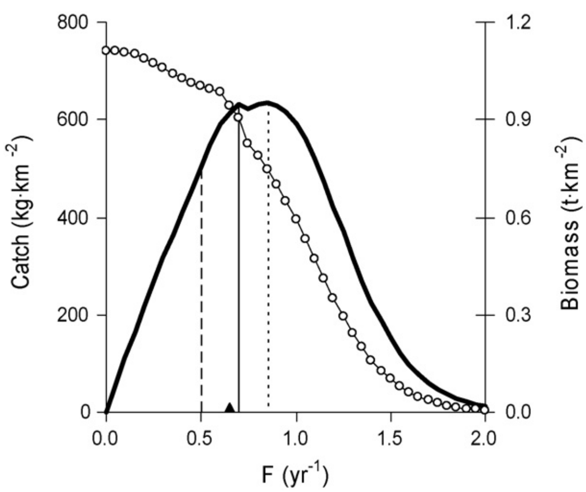

Fig. 2 – Multispecies catch and biomass equilibriums for Raja Ampat anchovy at various levels of fishing mortality. Solid curve is catch; open circles are biomass. Vertical lines: dashed (F_{2005}); solid ($F_{0.1}$); dotted (F_{MSY}). Triangle marker on *X*-axis shows multi-species optimum *F* (F_{OPT}). Exploitation is currently at 60% of F_{MSY}.

approximately 19% of the total mortality of this group. Recall that we used a relatively large anchovy catch estimate from Ainsworth et al. (2007), 70% larger than the estimate made by Bailey et al. (2008), while at the same time this model uses a revised anchovy biomass of 1.0 tonnes km^{-2} down from 1.5 tonnes km^{-2} of the preliminary model of Ainsworth et al. (2007). It is worth noting therefore that even under these conditions there is still no evidence of recruitment overfishing by the bagan fishery.

The current fishing mortality on anchovy (F_{2005}) is calculated to be 0.5 year^{-1}, which is approximately 60% of F_{MSY} (0.85 year^{-1}). The precautionary fisheries management target $F_{0.1}$ is lower than F_{MSY}, at 0.7 year^{-1}. We also present the multi-species optimum fishing mortality (F_{OPT}), 0.65 year^{-1}, which maximizes the combined fishery value of anchovy, skipjack tuna, other tuna, billfish and mackerel. It is very similar to the $F_{0.1}$ exploitation level considering the range of error of the model. When anchovy fishing mortality is at F_{OPT}, the combined value of these fisheries is about 7% higher than under the current conditions (F_{2005}). When anchovy fishing mortality is at F_{MSY}, the combined value of these fisheries is only 3.6% higher than current conditions.

Fig. 3 shows the results of an MTI sensitivity analysis. Of all anchovy predators, birds have the greatest effect on anchovy populations. The relationship in Fig. 3 will reflect a strong direct predator–prey interaction, but it is also affected by the mediation function mentioned earlier, in which tuna facilitate the consumption of anchovies by birds. Losses in anchovy biomass will not only have a direct negative impact on the bird population as a loss of prey; it will also reduce the biomass of tuna and other large pelagics (see Fig. 2) thereby removing a facilitation effect that worked in the birds' favour.

In Fig. 4, we model an anchovy population collapse. Economic impacts extend throughout the fleet. Without increasing their fishing effort, trolling operations for tuna loose more than 50% of their catch value. The bagan fishery follows then other commercial fisheries for large pelagics. Due to a small increase in the biomass of reef fish and exploited inver-

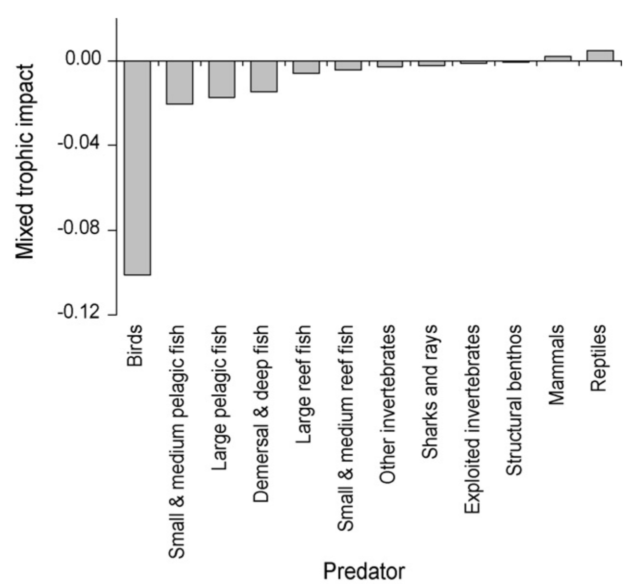

Fig. 3 – Mixed trophic impacts on anchovy as impacted group. These functional groups benefit from high anchovy biomass through predation or indirect relationships. Birds have the strongest negative impact on anchovy biomass.

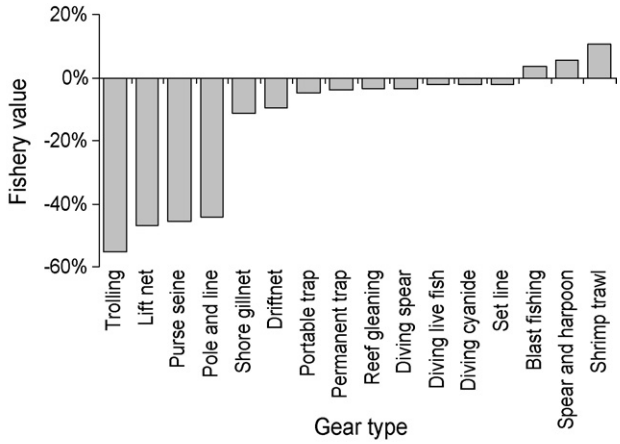

Fig. 4 – Incidental economic impacts of a major anchovy collapse. Fishery values assume status quo levels of fishing pressure and reflect the post-collapse ecosystem biomass equilibrium. Anchovy biomass is near zero.

Fig. 5 – The effect of anchovy biomass on the variability of large pelagic piscivore populations. Based on 20-year Monte Carlo simulations (2005–2025) using stochastic climate forcing (*n* = 10). Error bars show 1S.D. Y-axis shows average coefficient of variation for the biomass of large pelagic groups.

tebrates from predation release, there is an incidental increase in the value of some reef and invertebrate fisheries.

Fig. 5 considers an ecosystem service performed by anchovy. High anchovy biomass may have a stabilizing effect on large pelagic piscivore populations under stochastic climate variation. The average coefficient of variation for the large pelagic groups (skipjack, other tuna, mackerel and billfish) is highest when anchovy biomass is low. When anchovy is at B_{MSY}, there is a greater amount of variation in the biomass of its pelagic predators. When anchovy is at B_0 there is less variability. There are uncertainties relating to the fact that the wide-ranging nature of the large pelagic species cannot be fully represented in the model (see Section 2). It is also unclear from these results whether a large anchovy population would forestall a major perturbation in piscivore populations.

3.2. *(Q2) Restricting commercial fisheries on groupers*

When commercial fishing for groupers is eliminated, the catch of artisanal gear types increases by about 3% (Fig. 6A). This assumes no increase in the artisanal harvest rate; the improvement is a result of the additional grouper biomass that becomes available for artisanal fishers. The artisanal harvest rate could otherwise be increased by 17.5% to maintain the same amount of grouper catch overall (right—Fig. 6B).

In terms of system biodiversity, there is no advantage to shifting fishing effort from commercial gear types to artisanal gear types unless there is a concurrent reduction in grouper catch. When commercial catch is completely replaced by artisanal catch, ecosystem biodiversity remains the same to within 0.01% using the Q90 ecosystem biodiversity index (Ainsworth and Pitcher, 2005). For comparison, there is a 2.5% increase in Q90 biodiversity if artisanal fishing effort does not replace the missing commercial effort. The Shannon–Weaver biodiversity statistic did not detect any notable differences between the scenarios (Shannon and Weaver, 1949; adapted for EwE by Ainsworth, 2006).

Halting commercial fisheries for groupers results in an increase in the biomass of large reef fish, structural benthos (from the cessation of blast fishing), small and medium reef fish (Fig. 7). It also results in the decrease of some demersal and deepwater fish, which are more heavily exploited by artisanal gear types.

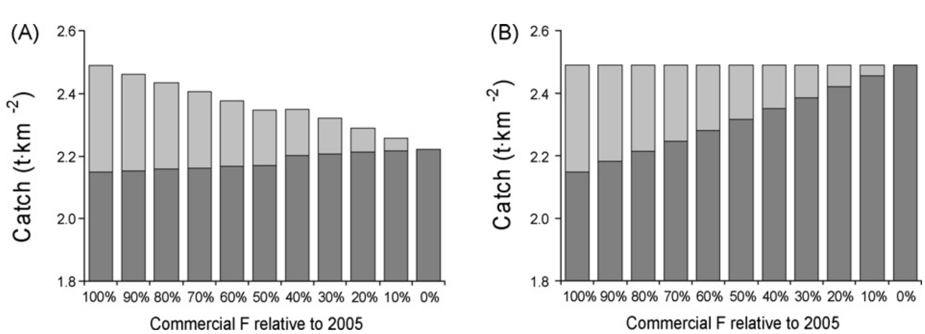

Fig. 6 – Artisanal benefits of restricting commercial grouper fisheries. When commercial fisheries are restricted, higher grouper biomasses are available for the artisanal fleet (artisanal fisheries: dark grey; commercial fisheries: light grey). (A) Assumes constant artisanal harvest rate at 2005 level. (B) Assumes increasing artisanal effort to maintain steady catch levels.

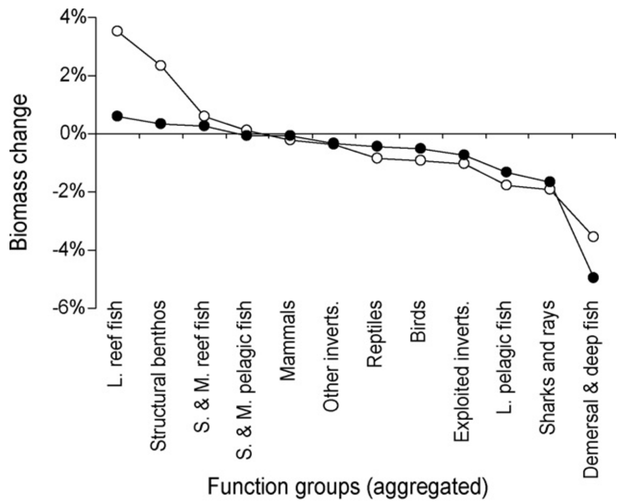

Fig. 7 – Ecosystem biomass change when commercial fishing for groupers is eliminated. When commercial fisheries for groupers are eliminated, the left most groups increase in biomass and the right most groups decrease. *Open circles*: Commercial fishing is eliminated and artisanal fishing does not replace the missing effort. *Closed circles*: Artisanal fishing replaces missing commercial effort. The biomasses of all relevant EwE functional groups have been combined into these categories.

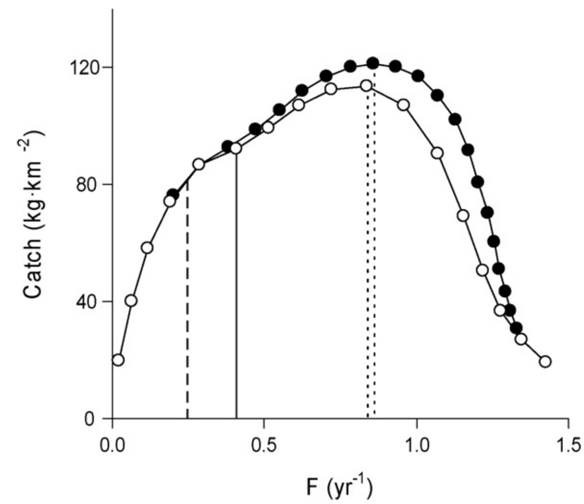

Fig. 8 – Multispecies catch equilibrium curve for groupers. *X*-axis shows grouper *F*; *Y*-axis shows catch (adult stanza only). Surplus production curve is calculated by incrementing the commercial fleet (solid circles) or artisanal fleet (open circles). Commercial fisheries achieve a slightly more efficient use of the stock (though not necessarily of the assemblage); multi-species MSY is about 7% higher using the commercial suite of fishing gears due to trophic effects. Vertical lines: dashed (F_{2005}); solid ($F_{0.1}$); dotted (F_{MSY}).

The commercial gear types we have outlined tend to catch a large proportion of grouper relative to other species; 26% of their total catch comes from the grouper functional groups. The artisanal fisheries are broader. Only 4% of their total catch is composed of groupers. At high levels of grouper fishing mortality, there are greater disturbances to the ecosystem made by artisanal fisheries. These result in lower grouper biomass and catch (Fig. 8) compared to commercial exploitation. The difference in the predicted MSYs is not much (~7%) considering the error range of the model. Although, the current exploitation status of groupers (F_{2005}) is about 30% of F_{MSY}, catch is already at 88% of MSY. A small sustainable increase in catch might be achievable if we accept an increased risk of stock collapse and a reduction in the average body weight of fish.

3.3. (Q3) Restrictions to reef fish net fisheries

Fig. 9 presents some of the ecosystem effects predicted by the model of restricting net fisheries. Complete closures of shore gillnet and drift net fisheries would result in an increase of large reef-associated fish biomass, on the order of 5%. A smaller increase is predicted for small and medium reef fish. Net fisheries have a relatively large impact on the biomass of pelagic fish. They intercept anchovies, and so we see a similar surplus production curve in the small and medium pelagic fish category as we see in Fig. 2. Large pelagic fish could increase by as much as 8% with a closure of commercial net fisheries. However, net fisheries are synergistic with some invertebrate fisheries—when we remove net fisheries,

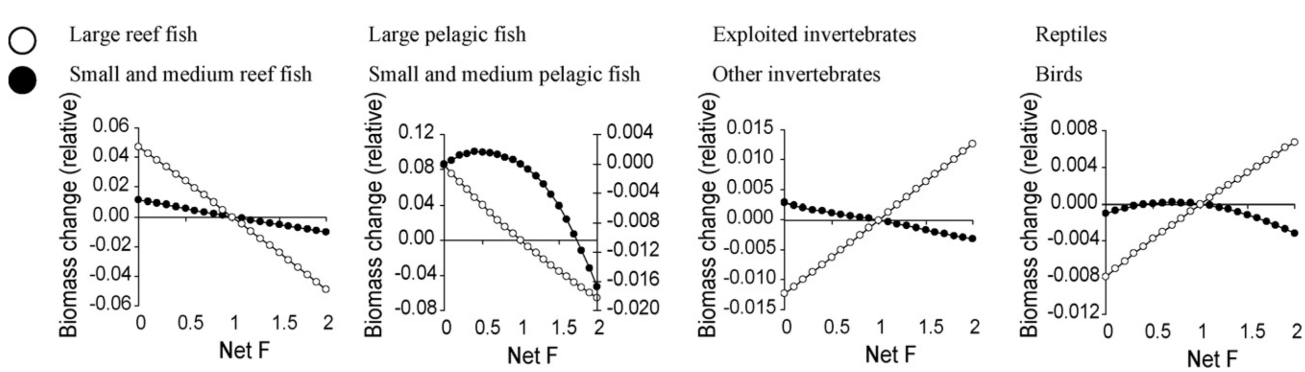

Fig. 9 – Ecosystem biomass changes at various levels of fishing mortality due to net fisheries. The biomasses of all relevant EwE functional groups have been combined into these generic categories. *X*-axis shows net fishing mortality relative to 2005 levels.

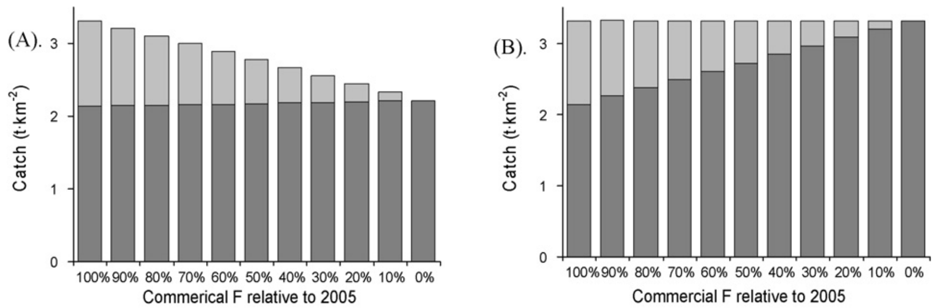

Fig. 10 – Effect of restricting net fisheries for reef fish. When net fisheries for reef fish are restricted, higher reef fish biomasses are available for non-net gear types (non-net fisheries: dark grey; net fisheries: light grey). (A) Assumes constant non-net gear effort at 2005 level. (B) Assumes increasing non-net effort to maintain steady catch levels overall. When net fishing is completely eliminated, the non-net fishery effort can increase by 65.5% to maintain reef catches (far right, B).

there is a small negative impact on the biomass of exploited invertebrates, a decrease of about 1%. It is not much considering the error range of the model. There is little effect on birds or reptiles.

When net fisheries are restricted, other fisheries that target reef fish may become more profitable. Without increasing the fishing rate of non-net fisheries, there is a 4% increase in catch in non-net gear types due to an increase in reef fish biomass. The fishing rate of non-net fisheries can increase by 65.5% in order to maintain the total catch of reef fish (but not the relative species mix) (Fig. 10). Removing all net fisheries results in an increase in ecosystem biodiversity by a small amount, <1%, using either the Q90 statistic or Shannon–Weaver index. However, if we increase effort in non-net fisheries to compensate, biodiversity is actually worse off than the status quo condition with nets allowed: there is a 1.7% reduction in Q90 biodiversity (Shannon–Weaver index is not sensitive to the changes, <0.1% difference).

3.4. *(Q4) Effects of blast fishing*

In Fig. 11, we predict ecosystem impacts at various intensities of blast fishing. At 10 times the current blast fishing rate,

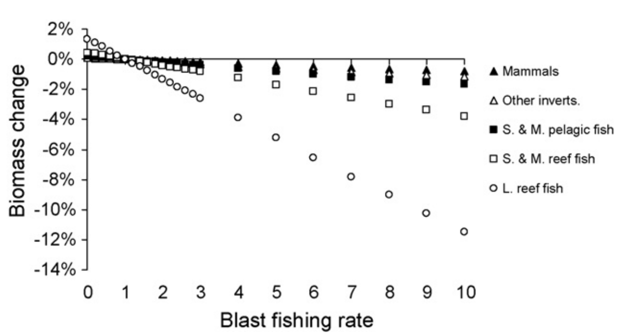

Fig. 11 – Ecosystem biomass change with increasing blast fishing. Large reef fish and structural benthos are depleted by increased levels of blast fishing. The biomasses of all relevant EwE functional groups have been combined into these generic categories. Baseline fishing rate (2005) is equal to 1.

the population of large reef fish decreases to a new biomass equilibrium about 12% lower than today. Small and medium reef fish decline 4%. Most other functional groups change in biomass by only a small amount, but demersal and deep water fish and large pelagic fish increase by 9% and 7%, respectively.

As estimated in the models, blast fishing currently accounts for 2.7% of the total catch of reef fish. The amount of blast fishing would therefore have to increase by many times before the capture rate became large compared to alternative reef fishing methods. However, even the current amount of blast fishing is probably outpacing the rate of coral re-growth. Pet-Soede et al. (1999) estimated a current loss of 3.75% coral cover every year due to blast-fishing in SW Sulawesi, near Raja Ampat. This is high compared to the natural recovery rate of coral species: Saila et al. (1993) considered a moderate estimate of coral recovery to be 1% per year. Empirical evidence from Fox et al. (2003) and Fox and Caldwell (2006) confirmed a similarly slow rate of coral recovery following blast fishing events. It is concerning that there is the potential for a phase shift to occur in affected areas, if succeeding species inhibit re-colonization by corals (Hughes et al., 2003).

Fig. 12 shows the impacts of an increase in the blast fishing harvest rate on juvenile reef fish. Through direct catch and bycatch we expect a 1.6% decline in the biomass of juvenile reef fish when blast fishing is increased ten times. However, when we factor in the loss of refuge, there is a 3.2% decline in the biomass of juvenile reef fish. With this, the catch value of other reef fisheries drops between 5% and 10%. A decrease in the rate of blast fishing from the current 2005 level will produce only a small return on the biomass of juvenile reef fish.

It is important to note that there are large uncertainties in the results. The mediation functions were not parameterized empirically by Ainsworth et al. (2007), although testing suggests that the results are robust against various forms of the mediation function. When we increase the maximum refuge benefit offered by corals from 2× (used by Ainsworth et al., 2007) to 15×, the resulting biomass of juvenile reef fish predicted by the model varies by only 0.01% the predator–prey vulnerability term in Ecosim is highly non-linear and the populations of juvenile reef fish are strongly influenced by bottom-up trophic control in the model.

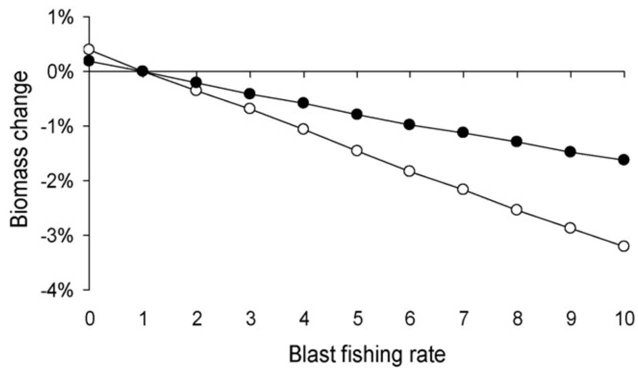

Fig. 12 – Biomass decline in juvenile reef fish with increased blast fishing. The biomass of juvenile reef fish declines with increased blast fishing due to direct fishing impacts and loss of refuge space. *Closed circles*: Biomass decline from fishing (no mediation functions included). *Open circles*: Biomass decline from fishing and loss of refuge (mediation functions included). Blast fishing rate is presented relative to 2005.

3.5. *(Q5) Maximization of tuna fisheries*

Modelling suggests that the current catch levels for skipjack tuna and other tuna are both very close to MSY (Fig. 13). Any increase in the harvest rate for these species would therefore be unadvisable, and could lead to recruitment overfishing.

There are caveats associated with Fig. 13 that could further reduce the estimate of the safe extraction rate for skipjack tuna. Between the years 1998–2001, biomass of the western and central Pacific Ocean stock was thought to be at the highest levels in 30 years thanks to an upward shift in recruitment rates occurring during the mid-1980s (Langley et al., 2003), and El Ninõ events in the 1990s may have benefited Skipjack tuna recruitment as well (SCTB, 2004). Skipjack biomass is now thought to lie above the level that produces MSY (SCTB, 2004) and our model concurs with this. However, if the current oceanic regime is favourable for tuna then future fluctuations in climate may yet demand a lower harvest rate for sustainable use.

The current biomass of bigeye tuna (*Thunnus obesus*) is thought to lie above the MSY level (SCTB, 2004). The biomass of albacore (*Thunnus alalunga*) in the south Pacific may be at approximately 60% of B_0, while the biomass of yellowfin (*Thunnus albacares*) in the western central Pacific Ocean may be 65–80% of B_0 (SCTB, 2004). Results from our 'other tuna' group agree well with those assessments. Fig. 13 indicates that the other tuna group is currently at 58.4% of the B_0, and that current biomass lies just over B_{MSY}. There is little margin to safely increase the current harvest rate of this functional group.

Fig. 14 compares the current catch estimate for the tuna groups with the precautionary $F_{0.1}$ limit and the multi-species optimal F_{MSY}. It also considers a theoretical maximum catch limit that was set using the policy optimization routine in Ecosim (Christensen and Walters, 2004). The maximum sustainable catch rate for skipjack tuna under this extreme fishing regime is only about 14% larger than the multi-species MSY. The catch increase for the 'other tuna' functional group is greater; the optimal fishing solution increases catch over the current MSY by two times, although many solutions failed to achieve this best-case scenario (see error bars in Fig. 14B). However, it is unlikely that this tuna policy could be achieved in reality because of the highly coordinated and selfless cooperation of fishing sectors that is required. Management error would also prohibit this level of improvement, whereas Ecosim executed the fishing pattern perfectly. If it were realized, the ecosystem would become more like a monoculture: Q90 biodiversity is predicted to decrease by 8% before equilibrium is reached. For comparison, the status quo fishing regime is predicted to cause a 4% decline in biodiversity within the next 25 years without any further expansion of the fishery.

4. Discussion

4.1. *(Q1) Effects of the anchovy fishery*

Results from the modelling suggest that the anchovy stock in Raja Ampat is not overexploited, so any decline in their pop-

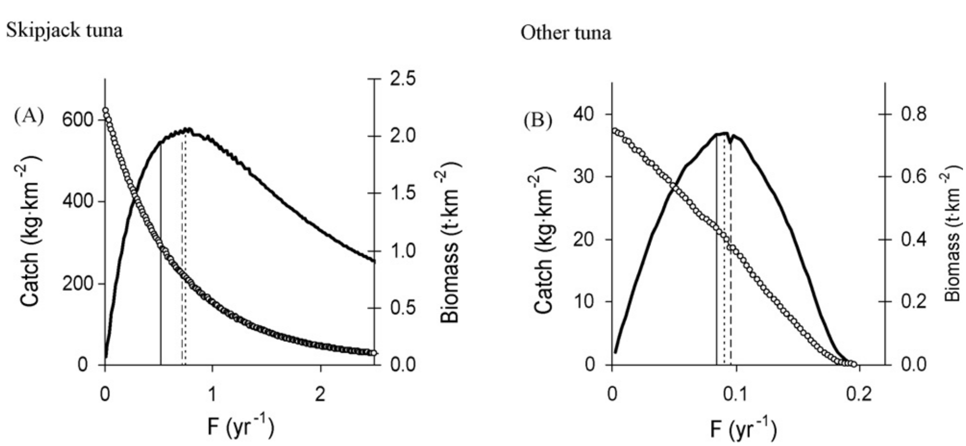

Fig. 13 – Multispecies catch and biomass equilibriums for tuna at various levels of fishing mortality. Solid curve is catch; open circles are biomass. Vertical lines: dashed (F_{2005}); solid ($F_{0.1}$); dotted (F_{MSY}). Exploitation is currently close to FMSY for both groups.

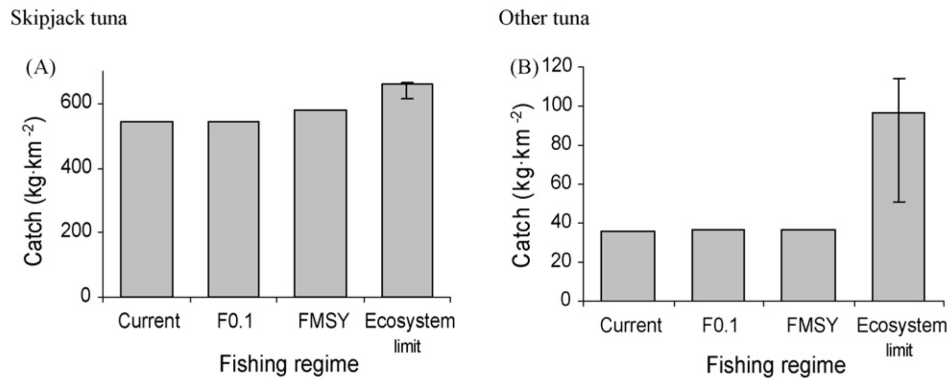

Fig. 14 – Tuna catch expected under various fishing regimes. Current catch represents estimated total extractions for year 2005. $F_{0.1}$ is the level of fishing mortality at which the slope of the yield-per-recruit curve is 10% of the slope at the origin, F_0. F_{MSY} is the fishing mortality that will maximize sustainable yield. Ecosystem limit refers to a hypothetical ecosystem optimized to deliver maximum sustainable tuna catch (e.g., competitors and predators removed; prey groups increased). Error bars show the range of solutions determined by the optimization facility in Ecosim (95% confidence intervals); the upper error bar represents the best solution, however, most iterations find a lower value (bar graph shows average).

ulation noted by fishers or researchers could be short-term or localized to inshore areas. However, the equilibrium analysis that we performed assumes constant ecosystem carrying capacity and climate conditions, so the estimates of exploitation status are most appropriately viewed as a best-case scenario. Any further increase in the anchovy exploitation rate could be cause for concern because of the variable nature of small pelagic stocks, for example, due to temperature (Fréon et al., 2005), salinity (Goarant et al., 2007) or current patterns (Lett et al., 2007), and the fact that (more valuable) piscivorous species directly rely on the anchovy population.

Anchovies may provide an important conduit through which primary and secondary production is channelled to higher trophic levels. We know this to be true of forage fish in other ecosystems (e.g., Cury et al., 2000; Hunt and McKinnell, 2006) so anchovies can have a controlling influence on system dynamics (Libralato et al., 2006; Coll et al., 2007). Anchovy also dominate the forage fish assemblage in Raja Ampat (Erdmann, personal communication, 2006). The results of this study suggest they are an important link, especially to large pelagic piscivores and bird populations. From a single-species perspective, we may wish to pursue a fishing policy for anchovy that would increase the catch rate closer to MSY, but such a policy would ignore potential ecosystem services offered by this functional group.

Anchovy are only moderately exploited in Raja Ampat. Eliminating the fishery for them would cause their biomass to increase close to the unexploited level, B_0, which is 11% higher than the current stock size (Fig. 2). There will be negligible changes in the biomass of predators like birds and pelagic fish, but there may be a beneficial stabilizing effect on fisheries for more valuable large pelagic fish. If a limited fishery is allowed, at or below current exploitation rates, the stock will remain safe from recruitment overfishing and it will be more resilient to climate variation than under a F_{MSY} policy. The multi-species optimal fishing effort lies near $F_{0.1}$. An increase in fishing effort beyond this amount will reduce total income from pelagic resources; a large increase (e.g., 2×) would cause a noticeable decline in bird and piscivore populations.

4.2. *(Q2) Restricting commercial fisheries on groupers*

If commercial fisheries on groupers were restricted, more resource would be available for artisanal fishers. However, artisanal fisheries are greater in scale; they catch almost six times as much as commercial fisheries so the relative increase in catch for the artisanal fleet would be modest. With commercial fisheries removed, groupers increase in biomass and system biodiversity improves. However, any ecological benefit from closing commercial fisheries is lost if artisanal fisheries were to increase to compensate for the missing effort. There are differences in the way that commercial and artisanal gear types interact with the ecosystem in terms of trophic effects, but the differences only become noticeable at high (and unsafe) exploitation rates.

If we completely removed commercial fisheries for groupers, and did not replace the missing effort, total catch of groupers would be reduced about 14%, and the stocks would assume an equilibrium biomass 37% higher than the status quo prediction. We did not detect any significant difference in the food-web or biodiversity impacts of commercial versus artisanal gear types. We did not consider the toxicological effects of cyanide fishing.

4.3. *(Q3) Restrictions to reef fish net fisheries*

Restricting net fisheries on coral reefs might be a viable management option to protect inshore reef fish stocks, especially in areas near villages where locals could participate in monitoring and enforcement. Results from our study suggest there would be an economic incentive for them to do so. Restricting net fisheries has the potential to increase catch in both pelagic fisheries and artisanal reef fisheries. Biodiversity might improve if we restrict net fisheries, but if effort is increased in other sectors to compensate for missing catch, biodiversity could be worse off than under status quo conditions.

4.4. (Q4) Effects of blast fishing

Blast fishing is an indiscriminate catching method, but unlike the hooks, nets and traps used in artisanal fisheries, much of the catch goes unutilized. Other commercial reef fisheries, which are pursued mainly by divers, are relatively clean unless cyanide is involved. They focus on a small number of species and produce little or no bycatch. Blast fishing has more serious potential to impact the ecosystem, and in unpredictable ways through the compounding effects of removals, untargeted mortality and destruction of refuge space. Injured fish may also suffer increased predation mortality; this effect was not captured in the model.

Nevertheless, the results of this study highlight the indirect ecological and economic value of preserving coral reefs. As blast fishing increases, the loss of refuge space can hinder juvenile reef fish populations as severely as direct mortality from the fishery itself. It is difficult to judge how widespread the fishery is in Raja Ampat, but the level of fishing effort in the model is not sufficient to provoke a major response in the food web even when increased by several times. The structural damage to corals, however, is more concerning than trophic effects because the slow re-growth rate of corals makes any damage cumulative. Even under status quo conditions, the destruction rate of coral habitat could be greater than the replacement rate. In terms of ecosystem functioning, the loss of refuge space incurred so far in Raja Ampat from blast fishing has probably had a minor effect on reef fish populations compared to exploitation by legitimate fisheries. Nevertheless, even a small amount of blast fishing will eventually hinder alternative development options like ecotourism in Raja Ampat.

4.5. (Q5) Maximization of tuna fisheries

Our results suggest that it is not advisable to permit any further increases in the catch for skipjack tuna or other tuna. The models predict that both groups are fully exploited, so any increase would result in recruitment overfishing and reduce stock production. Unfortunately, our model has a limited ability to represent stock dynamics of these highly migratory species. Mortality sources from fishing or environmental stressors occurring outside of the Raja Ampat system are not explicitly accounted for in the model. However, it is unwarranted and possibly dangerous to assume that Raja Ampat's tuna populations are subject to fewer pressures in other parts of their range (Myers and Worm, 2003).

5. Conclusions

Despite being seriously data poor by developed-nations standards, the targeted acquisition of key field data for a tuned ecosystem simulation model has enabled Raja Ampat stakeholders to evaluate the likely ecosystem-wide effects of five high priority issues that are part of an implementation strategy for ecosystem-based fishery management.

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REFERENCES

Ainsworth, C., 2006. Strategic ecosystem restoration in Northern British Columbia. Ph.D. Thesis. Resource Management and Environmental Studies. University of British Columbia, p. 422.

Ainsworth, C., Pitcher, T.J., 2005. Modifying Kempton's species diversity index for use with ecosystem simulation models. Ecol. Indication 6, 623–630.

Ainsworth, C.H., Varkey, D., Pitcher, T.J., 2007. Ecosystem simulation models for the Bird's Head Seascape, Papua, fitted to data. In: Pitcher, T.J., Ainsworth C.H., Bailey, M. (Eds.), Ecological and Economic Analyses of Marine Ecosystems in the Birds Head Seascape, Papua, Indonesia. Fish. Cent. Res. Rep. 15 (5), pp. 6–172. Available: http://www.fisheries.ubc.ca/publications/reports/report15_5.php.

Ainsworth, C.H., Varkey, D., Pitcher, T.J., 2008. Marine ecosystem dynamic models of Raja Ampat, Indonesia supporting ecosystem based fisheries management. In: Bailey, M., Pitcher, T.J. (Eds.), Raja Ampat II. Fish. Cent. Res. Rep. 16 (1), p. 113.

Amarumollo, J., Farid, M., 2002. Exploitation of marine resources on the Raja Ampat Islands, Papua Province, Indonesia. In: McKenna, S.A., Allen, G.R., Suryadi, S. (Eds.), A Marine Rapid Assessment of the Raja Ampat Islands, Papua Province, Indonesia. Bulletin of the Rapid Assessment Program, vol. 22. Conservation International, Washington DC (Chapter 6).

Bailey, M., 2007. Economic analysis of unregulated and illegal fishing in Raja Ampat, Indonesia. Masters Thesis. Resource Management and Environmental Studies. University of British Columbia, Vancouver, 101 pp.

Bailey, M., Rotinsulu, C., Sumaila, U.R., 2008. The migrant anchovy fishery in Kabui Bay, Raja Ampat, Indonesia: catch, profitability and income distribution. Mar. Policy 32, 483–488.

Beamish, R.J., 1995. Climate Change and Northern Fish Populations. National Research Council, Ottawa, Ont., Canada, p. 739.

Chassot, E., Melin, F., LePape, O., Gascuel, D., 2007. Bottom-up control regulates fisheries production at the scale of eco-regions in European seas. Mar. Ecol. Prog. Ser. 343, 45–55.

Chesher, R.H., 1969. Destruction of Pacific Corals by the Sea Star *Acanthaster planci*. Science 165, 280–283.

Christensen, V., Pauly, D., 1992. ECOPATH II—a software for balancing steady-state models and calculating network characteristics. Ecol. Model. 61, 169–185.

Christensen, V., Walters, C.J., 2004. Trade-offs in ecosystem-scale optimization of fisheries management policies. Bull. Mar. Sci. 74, 549–562.

Christensen, V., Walters, C.J., 2005. Using ecosystem modeling for fisheries management: where are we? ICES CM. 2005/M:19. In:

ICES Annual Science Conference Proceedings, Aberdeen, Scotland, UK, 20–24 September.

Christensen, V., Walters, C.J., Pauly, D., 2004. Ecopath with Ecosim: A User's Guide, May 2004 ed. Fisheries Centre, University of British Columbia, Vancouver, Canada. Available: www.ecopath.org.

Coll, M., Santojanni, A., Arneri, E., Palomera, I., Tudela, S., 2007. An ecosystem model of the Northern and Central Adriatic Sea: analysis of ecosystem structure and fishing impacts. J. Mar. Syst. 67, 119–154.

Cox, S.P., Essington, T.E., Kitchell, J.F., Martell, S.J., Walters, C.J., Boggs, C., Kaplan, I., 2002. Reconstructing ecosystem dynamics in the Central Pacific Ocean, 1952–1998. II: The trophic impacts of fishing and effects on tuna dynamics. Can. J. Fish. Aquat. Sci. 59, 1736–1747.

Cury, P., Bakun, A., Crawford, R., Jarre-Teichmann, A., Quinones, R.A., Shannon, L.J., Verheye, H.M., 2000. Small pelagics in upwelling systems: patterns of interaction and structural changes in "wasp-waist" ecosystems. ICES J. Mar. Sci. 57, 603–618.

Dohar, A.G., Anggraeni, D. (Eds.), 2007. Valuasi Ekonomi Sumberdaya Alam di Kepulauan Raja Ampat. Conservation International, Bekerjasama dengan, Universitas Negeri Papua. Economic valuation report for Raja Ampat (in Indonesian), 72 pp. (Available: C. Rotinsulu. CI. Jl.Gunung Arfak.45.Sorong, Papua, Indonesia. Available: chris@conservation.or.id).

Donnelly R., Neville, D., Mous, P., 2003. Report on a rapid ecological assessment of the Raja Ampat Islands, Papua, Eastern Indonesia, held October 30–November 22, 2002. Report from The Nature Conservancy Southeast Asia Center for Marine Protected Areas, Sanur, Bali, Indonesia, 246 pp.

Erdmann M.V., Pet, J.S., 2002. A rapid marine survey of the northern Raja Ampat Islands (Eastern Indonesia). Report from Henry Foundation/The Nature Conservancy/NRM/EPIQ, 36 pp.

Erdmann, M.V., Pet-Soede, L., 1996. How fresh is too fresh? The live reef food fish trade in eastern Indonesia. Naga ICLARM Quart. 19, 4–8.

Fox, H.E., Caldwell, R.L., 2006. Recovery from blast fishing on coral reefs: a tale of two scales. Ecol. Appl. 16, 1631–1635.

Fox, H.E., Pet, J.S., Dahuri, R., Caldwell, R.L., 2003. Recovery in rubble fields: long-term impacts of blast fishing. Mar. Poll. Bull. 46, 1024–1031.

Fulton, E.A., Smith, D.M., Johnson, C.R., 2003. Effect of complexity on marine ecosystem models. Mar. Ecol. Prog. Ser. 253, 1–16.

Fréon, P., Cury, P., Shannon, L., Roy, C., 2005. Sustainable exploitation of small pelagic fish stocks challenged by environmental and ecosystem changes: a review. Bull. Mar. Sci. 76, 385–462.

Goarant, A., Petitgas, P., Bourriau, P., 2007. Anchovy (*Engraulis encrasicolus*) egg density measurements in the Bay of Biscay: evidence for the spatial variation in egg density with sea surface salinity. Mar. Biol. 151, 1907–1915.

Graham, M., 1935. Modern theory of exploiting a fishery and application to North Sea trawling. J. Conserv. Int. Explor. Mer. 10, 264–274.

Halim, A., 2007. Perception monitoring study. Unpublished data. TNC-CTC. Jl Pengembak 2, Sanur, Bali, Indonesia (contact: ahalim@tnc.org).

Halim, A., Mous, P., 2006. Community perceptions of marine protected area management in Indonesia. A report to National Oceanic and Atmospheric Administration (NOAA). NA04NOS4630288. Available: joanne_wilson@tnc.org.

Hughes, T.P., Baird, A.H., Bellwood, D.R., Card, M., Connolly, S.R., Folke, C., Grosberg, R., Hoegh-Guldberg, O., Jackson, J.B.C., Kleypas, J., Lough, J.M., Marshall, P., Nyström, M., Palumbi, S.R., Pandolfi, J.M., Rosen, B., Roughgarden, J., 2003. Climate change, human impacts, and the resilience of coral reefs. Science 301, 929–933.

Hunt, G.L., McKinnell, S., 2006. Interplay between top-down, bottom-up, and wasp-waist control in marine ecosystems. Prog. Oceanogr. 68, 115–124.

Kaczmarsky, L., Draud, M., Williams, E.H., 2005. Is there a relationship between proximity to sewage effluent and the prevalence of coral disease? Carib. J. Sci. 41, 124–137.

Langley, A., Ogura, M., Hampton, J., 2003. Stock assessment of skipjack tuna in the western and central Pacific Ocean. In: 16th Meeting of the Standing Committee on Tuna and Billfish, Moolootaba, Australia, 9–16 July (SCTB16 Working Paper. SKJ-1).

Larkin, P.A., 1977. An epitaph for the concept of maximum sustained yield. Trans. Am. Fish. Soc. 106, 1–11.

Leontief, W.W., 1951. The Structure of the U.S. Economy, second ed. Oxford University Press, New York.

Lett, C., Penven, P., Ayón, P., Fréon, P., 2007. Enrichment, concentration and retention processes in relation to anchovy (*Engraulis ringens*) eggs and larvae distributions in the northern Humboldt upwelling ecosystem. J. Mar. Syst. 64, 189–200.

Libralato, S., Christensen, V., Pauly, D., 2006. A method for identifying keystone species in food web models. Ecol. Model. 195, 153–171.

Marasco, R.J., Goodman, D., Grimes, C.B., Lawson, P.W., Punt, A.E., Quinn, T.J., 2007. Ecosystem-based fisheries management: some practical suggestions. Can. J. Fish. Aquat. Sci. 64, 928–939.

Martell, S., 2004. Dealing with migratory species in ecosystem models. In: Pitcher, T.J. (Ed.), Back to the Future: Advances in Methodology for Modeling and Evaluating Past Ecosystems. Fish. Cent. Res. Rep. 12 (1), pp. 41–44. Available at http://www.fisheries.ubc.ca/publications/reports/12-1c.pdf.

McFarlane, G.A., King, J.P., Beamish, R.J., 2000. Have there been recent changes to climate? Ask the fish. Prog. Oceanogr. 47, 147–169.

McKenna, S.A., Boli, P., Allen, G.R., 2002. Condition of coral reefs at the Raja Ampat Islands, Papua Province, Indonesia. In: McKenna, S.A., Allen, G.R., Suryadi, S. (Eds.), A Marine Rapid Assessment of the Raja Ampat Islands, Papua Province, Indonesia. Bulletin of the Rapid Assessment Program, vol. 22. Conservation International, Washington DC (Chapter 5).

Muljadi, A., 2004. Coastal rural appraisal report in Kofiau and Misool area of Raja Ampat Islands. The Nature Conservancy, South East Asia Centre for Marine Protected Areas, Sorong, Indonesia, 28 pp. Available: amuljadi@tng.org.

Myers, R., Worm, B., 2003. Rapid world-wide depletion of predatory fish communities. Nature 423, 280–283.

Okey, T., Vargo, G.A., Mackinson, S., Vasconcellos, M., Mahmoudi, B., Meyer, C.A., 2004. Simulating community effects of sea floor shading by plankton blooms over the West Florida Shelf. Ecol. Model. 172, 339–359.

Pandolfi, J.M., Bradbury, R.H., Sala, E., Hughes, T.P., Bjorndal, K.A., Cooke, R.G., McArdle, D., McClenachan, L., Newman, M.J.H., Paredes, G., Warner, R.R., Jackson, J., 2003. Global trajectories of the long-term decline of coral reef ecosystems. Science 13 (5635), 955–958.

Pauly, D., Silvestre, G., Smith, I.R., 1989. On development, fisheries and dynamite: a brief review of tropical fisheries management. Nat. Resour. Model. 3, 307–320.

Pet-Soede, L., Erdmann, M.V., 1998. An overview and comparison of destructive fishing practices in Indonesia. SPC Live Reef Fish Bull. 4, 28–36.

Pet-Soede, C., Cesar, H.S.J., Pet, J.S., 1999. An economic analysis of blast fishing on Indonesian coral reefs. Environ. Conserv. 26, 83–93.

Pitcher, T.J., Ainsworth, C., Lozano, H., Cheung, W., 2005. Evaluating the role of climate, fisheries and parameter uncertainty using ecosystem-based viability analysis. ICES CM 2005\M:24, pp. 1–6.

Plagányi, E.E., 2007. Models for an ecosystem approach to fisheries. FAO Fisheries 899 Technical Paper No. 477. FAO, Rome, 108 pp.
Plagányi, E.E., Butterworth, D.S., 2004. A critical look at what Ecopath with Ecosim can and cannot achieve in practical fisheries management. Afr. J. Mar. Sci. 26, 261–287.
Punt, A.E., Smith, A.D.M., 2001. The gospel of maximum sustainable yield in fisheries management: birth, crucifixion and reincarnation. In: Reynolds, J.D., Mace, G.M., Redford, K.R., Robinson, J.R. (Eds.), Conservation of Exploited Species. Cambridge University Press, UK, pp. 41–66.
Purwaka, T.H., Sunoto, 2002. Coastal resources management in Indonesia: legal and institutional aspects. In: Torell, M., Salamanca, A.M. (Eds.), Institutional Issues and Perspectives in the Management of Fisheries and Coastal Resources in Southeast Asia: The World Fish Centre Technical Report 60. ICLARM, Manila, Philippines, pp. 60–90.
Russell, E.S., 1931. Some theoretical considerations on the overfishing problem. ICES J. Mar. Sci. 6, 3–20.
Saila, S.B., Kocic, V.Lj., McManus, J.W., 1993. Modelling the effects of destructive fishing practices on tropical coral reefs. Mar. Ecol. Prog. Ser. 94, 51–60.
Sea Around Us (SAU), 2006. A global database on marine fisheries and ecosystems. World Wide Web site www.seaaroundus.org. Fisheries Centre, University British Columbia, Vancouver, British Columbia, Canada (visited 10 December 2006).
Shannon, C.E., Weaver, W., 1949. The Mathematical Theory of Communication. University of Illinois Press, Urbana, IL, p. 117.
Standing Committee on Tuna and Billfish (SCTB), 2004. Seventeenth Meeting of the Standing Committeee on Tuna and Billfish. Available: http://www.spc.org.nc/OceanFish/Html/SCTB/SCTB17/SCTB17_Final_Report.pdf.
Ulanowicz, R.E., Puccia, C.J., 1990. Mixed trophic impacts in ecosystems. Coenoses 5, 7–16.
Walters, C., Christensen, V., Pauly, D., 1997. Structuring dynamic models of exploited ecosystems from trophic mass-balance assessments. Rev. Fish Biol. Fish. 7, 139–172.

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An evaluation of progress in implementing ecosystem-based management of fisheries in 33 countries

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ABSTRACT

The performance of 33 countries was evaluated for ecosystem-based management (EBM) of fisheries in three fields (principles, criteria and implementation) using quantitative ordination including uncertainty. No country rated overall as 'good', only four countries were 'adequate', while over half received 'fail' grades. A few developing countries performed better than many developed nations. Two case studies test the method. In Indonesia, Raja Ampat and Papua, rated similar to the national evaluation, but better performance might follow successful implementation of a planned EBM initiative. A workshop in Australia rated regional fisheries managed by New South Wales 20% lower for EBM than federally managed fisheries.

1. Introduction

There has been a recent trend towards adopting ecosystem-based fishery management (EBFM). Although there are a bewildering number of different definitions and shades of meaning of ecosystem-based management (EBM) [1,2], there is widespread agreement about the need to move towards a new fishery management system that recognises explicitly how food web linkages and human interventions may affect sustainability in aquatic ecosystems [3–5]. This paper attempts to evaluate the current status of the implementation of EBM in fisheries worldwide.

Many of the issues now considered vital for EBM are implicit in the FAO (UN) Code of Conduct for Responsible Fisheries [6]. There is an urgent need to manage fisheries in a more ecologically sensitive manner and this is the strength of the overarching concept of EBM. Aiming to operationalise this concept, FAO has also issued guidelines for an ecosystem approach to fisheries [7]. Implementation, however, of the stock-specific 'traffic light' reference points approach from the FAO guidelines will be difficult until clear and simply measured EBFM indicators for management are agreed by the international community [8,9], a task that has proved more difficult than some envisaged, especially in data-poor fisheries [10,11]. In the meantime, a simple, practical approach published by Ward et al. [12] is easier to adapt as a basis for evaluating status. Many wish to distinguish EBM from EBFM or the ecosystem approach to fishery management (EAFM): as in Ward et al. [12], we use EBM to denote a holistic approach to the management of fisheries, but not the management nor control of pollution, shipping lanes, recreation and other non-fisheries issues.

In fact, the Ward et al. [12] framework is largely based upon the FAO Code of Conduct which, although it originated in the early 1990s before ecosystem thinking became widespread, provides a very robust scheme of key elements such as ecological health, stakeholder involvement and spatial management. As there is already a published analysis of compliance of over 50 countries with the Code of Conduct [13], we were able to use extracts from this material, together with its estimated uncertainties, to score whether the fisheries are managed in accordance with the WWF framework proposed in Ward et al. [12]. In short, we used the scores countries received under the Code of Conduct assessment to evaluate the specific EBM issues identified in the WWF framework.

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2. Methods

2.1. Selection of countries

Our analysis was based on countries and not on individual fisheries, since under the Law of the Sea Convention (UNCLOS) nation states have a legal responsibility for the control of all fisheries within their EEZs and for their vessels on the High Seas.

We have chosen 33 countries for the main analysis as representing the top 90% of the world fish catch (see Table 2, the world catch in 1999 is taken as the reference point [13]). In addition, Australia (number 46 in the world catch) is included as a case-study example.

2.2. Scoring

Fishery management in the 33 countries was scored against the three main sets of the listed attributes from Ward et al. [12]. These were, overall principles (5 attributes; Table 2, p. 19 in Ward et al.); criteria for success (6 attributes; Table 3, pp. 19–20 in Ward et al.); and implementation steps (12 attributes; Table 6, pp. 50–51 in Ward et al.). Evaluation fields were set up for each of these by assessing material from published country reports on the compliance of over 50 countries with the Code of Conduct for Responsible Fisheries (13) against the criteria detailed in Table 2. Performance scores were allocated on a scale of 0–10, together with their likely ranges: these were set out on individual scorecards for each country. Scores over of 7/10 and above were considered 'good' and hence likely to lead to reasonably effective implementation of EBM, while scores of 4/10 or less are taken as unacceptable or 'fail grades'.

As a test of the utility and consistency of the method, two additional case studies were undertaken. First, scores for NSW and Australian fisheries were obtained from nine expert fisheries scientists, who participated in an ecosystem-based fisheries workshop[1] in July 2007 where a scoring framework similar to Table 1 was distributed. Experts canvassed were from New South Wales Fisheries, the Commonwealth Scientific and Industrial Research Organisation (CSIRO) and local universities. Of the nine individual sets of scores, one was discarded as their scores were completely uncorrelated to all the others; the rest exhibited similar ranges and patterns and hence were averaged.

The second test case was in Indonesia. Raja Ampat, an area of over 600 islands covering an area of about 45,000 km^2 in the "Coral Triangle" [14], is the site of a recent initiative in EBM set up by The Nature Conservancy, Conservation International and WWF-Indonesia with the local regency government [see 15]. Two of us (DV, TJP) have been involved with field teams of ecologists, social scientists and local universities in ecosystem modelling and analysis of field survey and interview data in support of this project. We have taken the opportunity to use this material to score the current fisheries in Raja Ampat against the criteria in Table 1. In addition, we estimated what the scores and their ranges might be after a successful implementation of the EBM project [16].

2.3. Analysis

For each of the three evaluation fields, raw scores were standardised using fixed reference points of zero and 10/10 and then entered into a non-metric multi-dimensional scaling [17] that incorporates a set of fixed anchor points from the 0 to 10 scoring range. Initial results were rotated to lie congruent with the fixed axis [the 'Rapfish' technique, 18]. The anchored and rotated MDS ordination can be thought of as extracting from the multidimensional raw data (in which each scored attribute represents a dimension) a single-dimension congruent with the original performance scores that maximises the differences among the data points along a scale from 0% to 100%. A second axis, normal to the first, is also extracted and may be thought of as expressing differing patterns of scores that achieve the same performance rating in different ways. This technique provided performance ratings on a percentage scale for each country, in each of the three evaluation fields. Uncertainty in the resulting ordination was allowed for by entering the upper and lower extreme values for each attribute score into a Monte Carlo simulation [19], which employed 500 iterations to estimate the upper and lower 95% tiles on the performance rating of each country.

3. Results

Scores allocated to each attribute are tabulated in Table 2. Following the method outlined above, final ordination results are shown in Fig. 1a–c. In these figures two-dimensional ordination plots show the differences in EBM Principles, Indicators and Implementation among the countries. Differences along the x-axis relate to the differences in EBM performance; differences in the vertical direction relate to the differences among the countries that are not due to EBM performance.

Fig. 2 shows how different countries score against the EBM performance rating scale (the x-axis on Fig. 1). Ratings over 70% were considered 'good' and likely to lead to reasonably effective implementation of EBM, while performance ratings of 40% or less represent 'fail grades' that are unlikely to help the implementation of EBM. Scores over 60% but $<70\%$ were considered 'acceptable' but in need of improvement.

For the five WWF EBM principles, there are no outstanding good performance ratings, and only six countries (USA, Norway, New Zealand, South Africa, Australia and Canada) have confidence limits that overlap the 'good' 70% threshold. Three countries (Iceland, Japan and Malaysia) have 'acceptable' scores over 60%. It is disappointing that almost half (14) of the 33 countries have 'fail grades' (Chile, China, UK, Argentina, Brazil, Pakistan, Indonesia, Morocco, Taiwan, Turkey, Viet Nam, Thailand, Russia and Myanmar).

For the six EBM indicators, four countries (Norway, New Zealand, USA and Iceland) achieve 'good' ratings than span the 70% threshold; while three countries (Canada, South Africa and Japan) have 'acceptable' performance levels over 60%. Over half (17) of the 33 countries have 'fail grades' (Mexico, France, Ecuador, UK, India, China, Argentina, Pakistan, Brazil, Indonesia, Morocco, Taiwan, Turkey, Russia, Myanmar, Viet Nam and Thailand).

For the twelve EBM implementation steps, no countries achieve 'good' performance ratings over 70%; while just two (USA and Canada) have 'acceptable' scores over 60%. In this evaluation field, two-thirds (21) of the 33 countries have 'fail grades' (Ecuador, Japan, Denmark, Brazil, Argentina, Malaysia, UK, Netherlands, France, Philippines, India, Indonesia, Pakistan, China, Taiwan, Myanmar, Turkey, Viet Nam, Morocco, Thailand and Russia).

Overall scores for EBM (totalled over the three evaluation fields) show that only two countries have 'good' performance ratings over 70% (Norway and USA), while four countries have 'acceptable' grades between 60% and 70% (Iceland, South Africa, Canada and Australia). But about half (16) of the 33 countries have

[1] "Towards Ecosystem-based Fishery Management in New South Wales"; New South Wales Government Fisheries Research Centre, Department of Primary Industry, Cronulla, NSW, Australia, 25–26 July 2007.

Table 1
Scoring framework used in the EBM performance evaluation

Evaluation Field 1: Five principles of the EBM framework	Score 0–10	Score range
• The central focus is maintaining the natural structure and function of ecosystems, including the biodiversity and productivity of natural systems and identified the important species • Human use and values of ecosystems are central to establishing objectives for the use and management of natural resources • Ecosystems are dynamic; their attributes and boundaries are constantly changing and consequently, the interactions with human uses also are dynamic • Natural resources are best managed within a management system that is based on a shared vision and a set of objectives developed amongst stakeholders • Successful management is adaptive and based on the scientific knowledge, continual learning and embedded monitoring processes		

Evaluation Field 2: Six indicators of successful EBM

Key element	Expression in the fishery (objectives)	Mechanisms and enabling processes	Performance indicators	Score 0–10	Score range
The fishery operates in an effective policy framework	The management system has effective linkages to the conservation and socio-economic policies and strategies for the ecosystems where the fishery operates. The management system appropriately reflects national and international goals and objectives for conservation and sustainable use. Subsidies and incentives lead to improved EBM outcomes in the fishery	Review of regional and national policies and strategies to ensure consistency with EBM principles. Inter-agency procedures are efficient, effective and accountable. New subsidies and incentives reviewed by stakeholders to confirm ecological viability	The absence of policy inconsistencies that will prevent a fishery from achieving EBM. Inter-agency cooperation is effective and efficient. The absence of perverse subsidies and incentives in the fishery system		
Social, economic and cultural context of the fishery is incorporated	Stakeholders are identified from all areas of relevance to the fishery, and effectively participate in the management system. The management system and the implementation of objectives and targets are agreed across all stakeholders for both stock management and ecosystem integrity. Institutional changes result in increased integration and cooperation amongst stakeholders. Management decisions are based on the long-term social, economic and cultural benefits of the society	Procedures are in place for effective participation of stakeholders in all aspects of the management system (such as Management Advisory Committees, Consultative Councils). Management procedures are publicly accessible, and implemented according to a publicly available plan of the management. Regular review and revision procedures are in place to identify improvements to the management system. This should include professional assessment that is independent of the fishery and management agency	The fishery management plan is easily available and is periodically (at agreed regular intervals) open to public review and assessment. Fisheries status reports that include stock and ecosystem performance reports are periodically (at agreed regular intervals) distributed for public review and evaluation		
Ecological values are incorporated	Ecosystem values are identified, including ecosystem connections, conservation status, state of ecosystem integrity and critical habitat for utilised and non-utilised species. Agreed objectives, targets, strategies and performance indicators for enhancing or maintaining ecosystem integrity are developed and implemented. Achievement of the ecosystem objectives is assessed within the fishery management system in partnership with conservation and research sectors	Ecosystems have been mapped where the fishery operates, and the conservation status of important species and habitats determined. Habitats, species and ecosystem function vulnerability to fishery impacts have been assessed, and the targets and harvest strategy adjusted to be precautionary. Assessment of the fishery performance for ecological objectives is undertaken in conjunction with stakeholders, and procedures and outcomes are made public	The ecological integrity of specified sensitive habitats is not declining. Species considered at high or medium risk from fishing (or their surrogates) are identified and their status used as performance indicators. Populations of non-utilised (specified) species vulnerable to fishing impacts are not declining. The bycatch of (specified) protected or otherwise icon species is declining by an agreed proportion each year, or reduced to an agreed level considered acceptable		
Knowledge of utilised species is adequate	Agreed objectives, targets, strategies and performance indicators for stock status are developed and implemented. Achievement of fishery objectives is assessed within the fishery management system through comprehensive consultative structures Ecosystem dynamics are fully incorporated into stock assessment models and decisions are cautious. Effective	Stock assessments are timely, open to stakeholder participation, and fully transparent and accountable. Harvest strategies are cautious, and well-buffered against unpredicted failure of assumptions. No take zones' and marine-protected areas are designed to	Target and limit reference points are set at a precautionary level. Limit reference points for stock size and structure are not violated. The age structure and natural distributional range of the population are minimally altered. Stock assessments are open, inclusive and		

Table 1 (*continued*)

Evaluation Field 2: Six indicators of successful EBM

Key element	Expression in the fishery (objectives)	Mechanisms and enabling processes	Performance indicators	Score 0–10	Score range
	no-take zones are implemented as 'insurance' against unpredicted failure of the management system in respect of the target stock, associated non-target catch and bycatch, and wider ecosystem values	benefit both fisheries management and broad ecosystem goals. Catch levels are set within ecologically defined limits that are understood and agreed	participatory. No-take zones are agreed and adequately implemented as part of the fishery management system		
The resource management system is comprehensive and inclusive, based on reliable data and knowledge, and uses an adaptive approach	The fishery management system is structured using ecological classification (such as ecoregions, bioregions, and habitat classes). Baseline data or benchmarks are available for each performance indicator. Management data are collected for stock management and ecosystem integrity parameters. Arrangements are in place to facilitate the use of data from partner agencies, research collaborators or other sources. Stock and environmental assessments are conducted in collaboration with fishery operators, partner conservation agencies and other stakeholders e.g. Environmental Non-Government Organisations (ENGOs). The management system responds to new information and data in a timely and effective way. Procedures are in place to recognize and adopt new knowledge or data of importance to ecosystem integrity or stock management. Ecological risks are assessed in a comprehensive manner, and a precautionary decision-making framework is used to manage risks. Gaps in knowledge related to high or medium risks are given priority for research funding and implementation	An ongoing research programme is in place to improve basic knowledge of the life history characteristics of target species, associated and dependent species and the wider ecosystem where the fishery operates. The management system includes monitoring to evaluate the status of ecological indicators. Stakeholders participate in management decisions. Ecological risks are continuously reviewed to provide for alteration to the harvest strategy as appropriate	The amount and type of fishing effort in each habitat class. The amount and type of bycatch and discards is declining by an agreed proportion each year, or reduced to an agreed level. Bycatch of protected species is declining by an agreed proportion each year, or reduced to an agreed level. Research projects reflect the key ecological issues in the fishery. Comprehensive fishery data monitoring system on targeted species and bycatch is in place. The amount and type of fishing effort on each level of the population of the target species		
Environmental externalities are incorporated.	Cross-boundary issues are identified, and addressed within the management system. The long-term dynamics of ecosystems are incorporated into the development of objectives and targets. The management system considers the full range of human uses and aspirations for the ecosystems being managed	Statutory or other procedures are in place to ensure that fisheries managers are involved in management decisions that may affect the stock or the ecosystems where the fishery operates. Ecological risks and harvest strategies contain measures to assess and incorporate risks from long-term changes in ecosystems or the effects of their uses. Fishery managers and operators understand and are accountable for their decisions and actions and the impacts of these 'in the water'	Critical habitat for the fishery and identified key ecosystem components are protected from water pollution, coastal development or other externalities. Environment-protection strategies take into account the use by fisheries of coastal areas. Allocation of resources for harvest (of exploitable stocks) is made equitably across all legitimate claimants (e.g. requirements of the ecosystem; traditional, subsistence, recreational and commercial fishers) and recognises ecological constraints		

Evaluation Field 3: Twelve steps in implementing EBM

Component step	Involving	Intended outcomes	Score 0–10	Score range
Identify stakeholder community	Fishery management agencies, conservation agencies, conservation NGOs, local community groups, scientific/academic research community, fisher associations or cooperatives, higher and lower levels of government, fish processing/distribution groups, Indigenous representatives	A formal network of interested parties with whom the fishery representatives will participate to prepare and review the management of the fishery. A transparent and fully accountable process enabling the participation of all interested parties in the process of managing the fishery		
Prepare a map of ecoregions and habitats	Conducted by the fishers, research community, fishery managers, stakeholders and partners. Covers the full area of fishery operations. The focus is on areas where the fishes are, where they	Maps of the ecosystems throughout the fishery at scales of resolution consistent with the scale of the fishery. Resolved habitats at a scale consistent with the potential impacts of the		

	are fished, and any specific spawning, nursery or similar obligate habitats or locations. High resolution is needed in benthic primary producer habitats (such as algal beds, seagrasses, mangroves, coral reefs)	fishery. Coherent with other ecosystem classification initiatives (at both larger and smaller scales). Major features and exceptions documented (e.g. highly migratory species, oceanographic currents or features, boundary mismatches between taxa). Major uncertainties identified and documented as guidance for research and investigation programmes
3. Identify partners and their interests/ responsibilities	Conservation, environment protection, and coastal planning agencies from all levels of government. Major users and managers of other, possibly co-located, resources (e.g. tourism, mining, oil/gas, transport, and communications). Directly affected local communities	Clarify specific roles and responsibilities for management in the marine environment. Engage with other supportive interests. Promote the opportunity for coordination and integration, improved efficiency across government and better outcomes for marine management, better agency outcomes for lower cost, more accountability in government, more effective long-term solutions to marine ecological problems, and shared approaches to problems held in common
Establish ecosystem values	Fishers, research community, fishery managers, stakeholders, partners and the public; designed to identify all major uses and all major natural and ecosystem values throughout the area where the fishery operates	A detailed distributional analysis of the main attributes of the ecosystem where the fishery operates. A clear and agreed expression of the natural and use values, which could include: highly valued habitats; representative areas dedicated as reserves; protected species feeding, breeding, or resting grounds; fishing, spawning grounds, recruitment areas and migration paths for commercial species; highly productive areas such as upwellings; areas popular for recreational fishing or diving; areas used for ports and harbours; areas of high scenic and wilderness amenity; high cultural and historic value; traditional hunting grounds for indigenous peoples; areas of high tourism value; areas used for dumping of dredge wastes, military training, etc.
Determine major factors influencing ecosystem values	Establishing cause–effect relationships; consider factors both internal and external to the fishery management system. Conducted by the fishers, research community, fishery managers, stakeholders and partners	Identified hazards to marine ecosystems and their values from the full range of actual and potential human impacts that occur in the fishery region. These could include: extent of loss/damage of marine habitats; effects of specific fishing gear on benthic habitats; effects of pollution from coastal rivers on inshore habitats; risk of marine pest invasion and disruption to critical habitat or fishing operations; effects of the removal of the biomass of harvested species (in all fisheries) on trophically dependent species
Conduct ecological risk assessment	ERA conducted with participation of all stakeholders and partners, fishers, research community and the fishery manager: uses broad multi-disciplinary knowledge base; identifies key areas of uncertainty; open for public scrutiny and review; fully peer reviewed by independent authorities	Agreed estimates of high, medium and low risks of the fishery to the ecosystem values identified in step 5, such as the risk of the fishery to the protected species, and to the ecosystem, habitats, species and genetic diversity
Establish objectives and targets	Fishers, research community, fishery managers, stakeholders and partners. Performance objectives and targets established for: high- and medium-priority risks from the ERA; important aspects of the ecosystems (including protected species, critical habitat); stocks	Agreed and shared goals for specific elements of ecosystems. Specific performance objectives and targets for important elements of the ecosystem. Objectives and targets that are comprehensive and precautionary in terms of valued aspects of the ecosystems. Could include: maintaining or recovering population sizes of protected species; maintaining the distribution, area, species diversity and trophic structure of important habitats; reducing fishing effort in specific areas to help protect populations of benthic fauna; increasing the distribution and diversity of benthic fauna considered to be affected by fishing; rehabilitating marine ecosystems to a past (healthier) condition
Establish strategies for achieving targets	Fishers, research community, fishery managers, stakeholders and partners. Focus is on identifying appropriate and workable strategies to achieve objectives and targets, and on specific capacity matched to responsibilities for implementing strategies.	Series of prioritised strategies that define workable activities and responses to achieve specific objectives and targets identified in Step 7. Includes who is responsible, what funds and time frames are involved, what controls are needed and where data/outcomes

Table 1 (*continued*)

Evaluation Field 3: Twelve steps in implementing EBM

Component step	Involving	Intended outcomes	Score 0–10	Score range
	Strategies designed based on best understanding of the cause–effect relationships developed in Step 5, and matched to highest-priority needs for corrective actions identified in Step 6 (ERA). Use of incremental strategies where necessary and unavoidable	are reported and assessed. Strategies could include: declaring a network of sanctuary-protected zones; establishing buffer zones, where only specific uses, or types of fishing, are permitted research on improving gear design to reduce impacts on a sensitive habitat, or reduce the bycatch of an important species; improved fishery independent monitoring of catch, or bycatch; reducing pollution from coastal rivers; constructing fish escapement panels in trawl nets to avoid catch of a certain type and size of fish, or to reduce overall fish bycatch; implementing an industry code of practice to reduce risks of bait discards to bird populations		
Design information system, including monitoring	Fishers, research community, fishery managers, stakeholders and partners. Focus is on capture of appropriate data/information to determine if strategies are working as expected; objectives and targets are being achieved; cause–effect models are correct; fishery impacts are being reduced. Collaboration and contributions from partners identified	Efficient and effective fishery information system that provides data and information on stock and ecosystem performance (additional to information needed for stock management); identifies specific effects of fishery strategies on ecosystem values. Could include: Periodic mapping of important habitat distributions; population census of important protected species; species diversity in fished habitats; distribution of fishing effort by gear types and fine spatial scale; size/age classes in harvested species; species diversity in closed areas		
Establish research and information needs and priorities	Fishers, research community, fishery managers, stakeholders and partners. Focus is on identifying specific high-priority areas of uncertainty, and on quality science outcomes, for both stock and ecosystem issues. Collaboration and contributions from partners identified. Research strategies are fully peer reviewed or independently audited	Comprehensive research programmes targeted at resolving key ecosystem and stock issues in the fishery. Could include: habitat mapping; impact of fishing on specific habitat types; effects of coastal development on recruitment of harvested species; design of monitoring programmes to resolve important changes in habitats; biological data of key species (both utilised and nonutilised); determining the dietary preferences of harvested species and their major predators; species composition of bycatch with different gear types used in the fishery		
Design performance assessment and review processes	Fishers, research community, fishery managers, stakeholders and partners. Focus is on a process that is participatory and inclusive. The locations, timing and resourcing enables partner and stakeholder participation in reviews of performance of the fishery in relation to stock and ecosystem values. Performance outcomes peer reviewed by independent authorities	Periodic (but regular) forum for discussion, review and assessment of fishery performance by partners, stakeholders and the public. Periodic (but regular) forum for review, assessment and revision of monitoring data, objectives and targets by stakeholders and partners		
Prepare education and training package for fishers	Fishers, fishery managers, extension experts and stakeholders and partners	Outreach programme to provide training and support for fishers about new fishery management, ecosystem or other EBM initiatives, and provide local technical support for assessment and resolution of ecosystem issues		

Details of the three evaluation fields were taken from Ward et al. (2002). For each attribute, scores and ranges were allocated based on material in the Code of Conduct country reports (Pitcher et al. 2006). Scores of 7/10 and above were considered "good"; scores of 4/10 and below represented poor or "fail" grades.

Fig. 1. EBM performance ratings for fisheries in 33 countries in three evaluation fields: principles, indicators and implementation steps, taken from Ward et al. (2002). Figure shows two-dimensional ordination plots from the MDS analyses; horizontal axis indicates performance score on a percentage scale; vertical position relates to the other distinguishing features among countries; thin lines are 95% tiles from Monte Carlo simulations using errors on each score. Country abbreviations as below: Arg = Argentina, Aus = Australia. Brz = Brazil, Can = Canada, Chl = Chile, Chn = China (Peoples Republic), Den = Denmark, Edr = Ecuador, Fra = France, Ice = Iceland, Ind = India, Ids = Indonesia, Jap = Japan, Mal = Malaysia, Mex = Mexico, Mco = Morocco, Mya = Myanmar, Hol = Netherlands, NZ = New Zealand, Nwy = Norway, Pak = Pakistan, Pru = Peru, Phl = Philipinnes, Rus = Russia, SA = South Africa, Skr = South Korea, Spn = Spain, Twn = Taiwan, Thl = Thailand, Tky = Turkey, UK = United Kingdom, US = United States of America, Vet = Viet Nam.

'fail grades' of 40% and less (UK, Argentina, France, India, China, Brazil, Pakistan, Indonesia, Taiwan, Morocco, Turkey, Viet Nam, Myanmar, Russia and Thailand).

3.1. Test cases

Fig. 3 shows the results from the test cases plotted on two-dimensional ordinations against a background of the overall final results from Fig. 1. It is evident that the EBM performance rating for New South Wales fisheries is some 10–15% lower than fisheries managed by the Australian Commonwealth. As all Australian fisheries share similar features, we see only small differences on the vertical axis between New South Wales and Australian Commonwealth fisheries.

In all three cases, the Raja Ampat ratings of today's performance in EBM are not significantly different from the overall Indonesian value along the EBM performance axis. Unsurprisingly, there are, however, large differences the vertical axis that express differences between this small region of Papua and Indonesia as a whole. Fig. 3 also shows our projections of what ratings Raja Ampat might achieve if the present EBM plans were to be fully implemented.

4. Discussion: the challenge of implementing EBM worldwide

The comparison of Indonesian scores for Raja Ampat and the overall values for Indonesia, which were independently arrived at, show similarities that provide encouraging validation for the

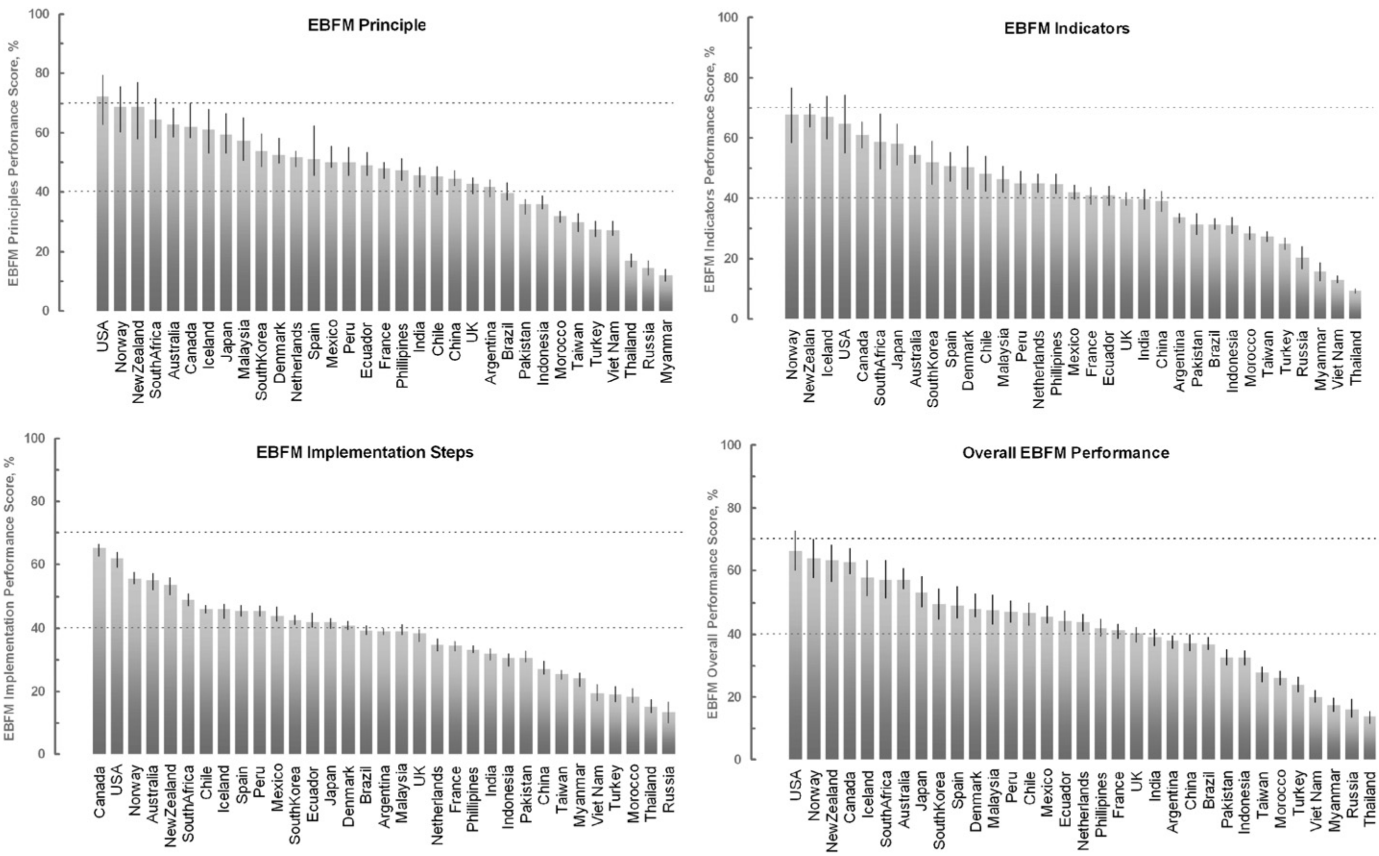

Fig. 2. EBM performance ratings (vertical bars) for fisheries in 33 countries in three evaluation fields: principles, indicators and implementation steps, taken from Ward et al. (2002), and an overall rating that averages the other three scores. Countries are shown in order of performance rating from left to right: thin lines are 95% tiles from Monte Carlo simulations using errors on each score. Upper broken line indicates "good' ratings or 70% or more; lower broken line shows "poor" or "fail" scores of 40% or lower.

method. Moreover, vertical axis results, expressing differences among fisheries, are also encouraging for the method. For example, there are only small differences on the vertical axis between New South Wales and Australian Commonwealth fisheries. Although Indonesia falls in the lower quartile of "fail" grades overall, Fig. 3 shows that the Raja Ampat region of Papua might achieve EBM ratings as high as the top five developed countries if present plans were to be successfully and fully implemented.

Most countries achieved lower ratings for Indicators than for EBM principles (evaluation field 2: 22/33); while almost all countries had lower ratings for implementation steps (evaluation field 3: 30/33). This finding is not surprising as it is easier to publish good intentions for EBM principles than to actually achieve the tangible steps towards EBM scored in evaluation field 3. On average ratings were 9.7% lower for implementation steps. One country, Myanmar, went significantly against this trend by having 12% higher performance on implementation steps, presumably reflecting the difficulty in finding any published principles for this country, as opposed to documented brave conservation efforts by a few individuals.

One of the EBM implementation steps has especially low scores in our analysis. "Setting up training courses in EBM for fishers and managers" averages only 1.0/10 (1.3 standard deviations below the mean), while 19/33 countries score zero. This 'training course' action would likely be a final implementation step in EBM, so that only countries that have already moved some way towards EBM will be able to achieve a reasonable score. Two other low-scoring questions are the "implementation of ecological risk assessment" (average 2.7/10; 7 countries with 0), and "strategies agreed among all stakeholders" (average 2.9/10: 7 countries with 0).

Our analysis reveals that only a few countries in the developed world are clearly moving towards EBM, but it is most interesting that several developing countries rank above their more developed neighbours (e.g., Malaysia, Peru, Mexico, Ecuador and South Africa), especially in Field 1 covering EBM Principles even if their ratings on Field 3, implementation steps, are generally lower. This may represent the actions of a few brave and progressive fishery legislators and managers in these countries and the more community-based nature of local fisheries management. Indeed, moving towards participatory fisheries management is a key aspect for success in implementing EBM. Many developing countries recognise that, in spite of some achievements towards the implementation of such approach, there is a need for capacity-building through awareness and direct technical assistance to help build their national capacity for the task [20].

Notable among the EBM scores are the dismal ratings of many developed European countries in spite of the Common Fisheries Policy undergoing an ecosystem-based reform in 2002. This can be seen graphically as a long horizontal cluster of high Human Development Index (HDI) countries to the left of the highest-ranking countries in Fig. 4. Despite academic excellence, widespread awareness of the issues and policy work emphasising the need to move towards EBM, to date it does not seem to have led to much clear regulation or action to implement tangible actions. Some may speculate on the reasons for this lamentable inertia among developed countries that undoubtedly have the resources for implementation. Bianchi et al. [20] suggest that such failures

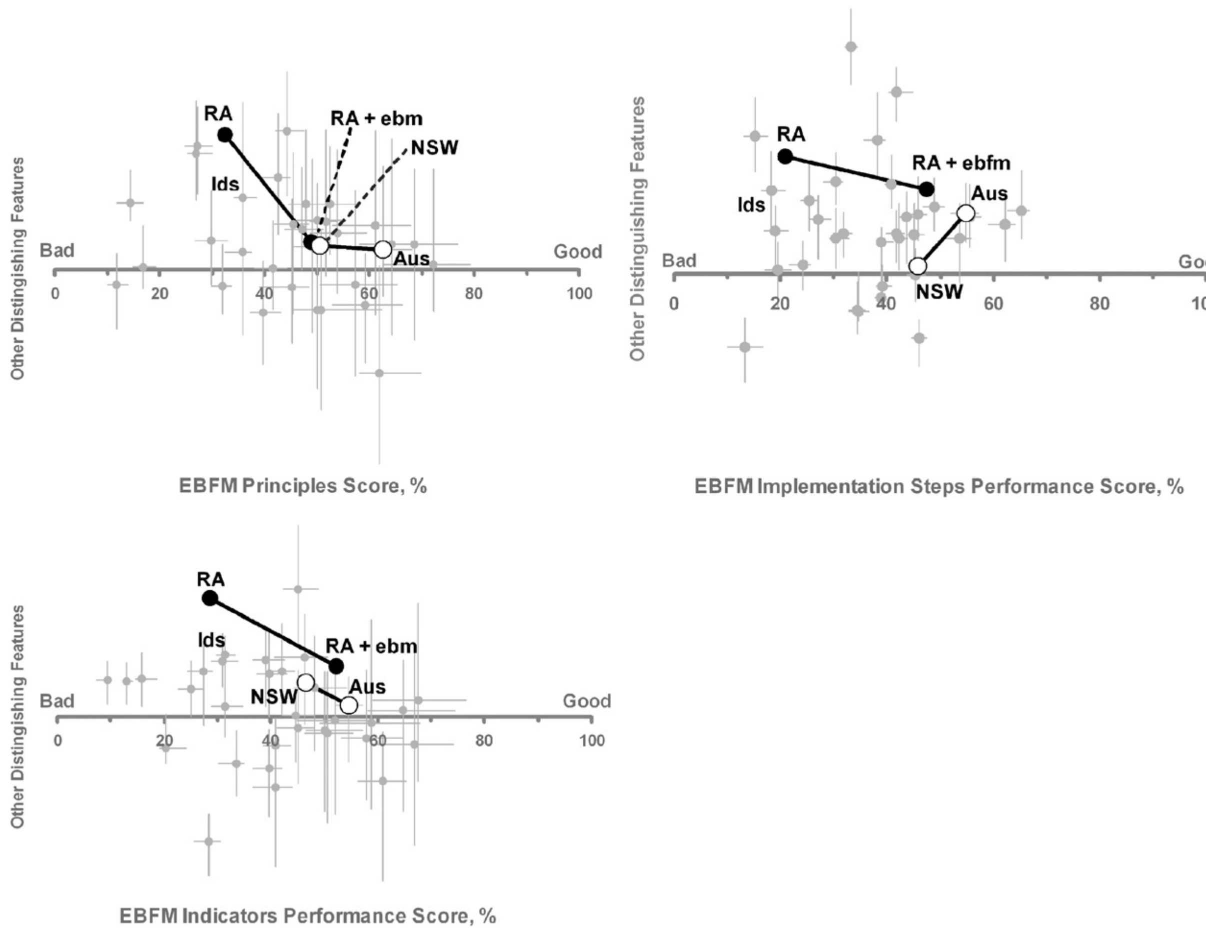

Fig. 3. Three fields of EBM performance ratings for Australian and Indonesian fisheries test cases. Main axes, symbols and confidence limits as in Fig. 1. For clarity, countries are unlabelled in this plot apart from Indonesia (Ids) and Australia (Aus). Rating for the state of New South Wales (NSW) is shown connected to the overall Australia values (open circles). Raja Ampat rating (closed circles, RA) is shown connected to hypothetical rating if a recent EBM initiative were to be successful (RA+EBFM) (see text for more details).

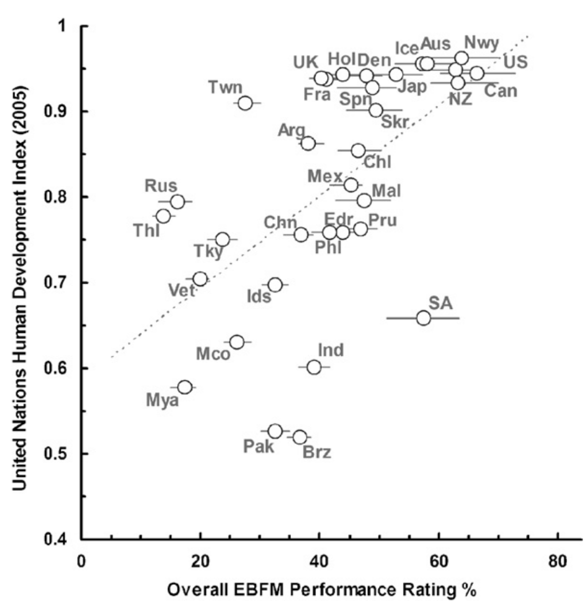

Fig. 4. Plot of United Nations Human Development Index for 33 countries (2005 data) against estimated overall EBM performance ratings. Thin horizontal bars are 95% limits of EBM values. Broken line: regression of HDI on EBM; sig at 99% level; COD = 0.29***.

may result from responding piecemeal to specific international agreements, advocacy pressure, trade requirements or immediate crises and not as a result of the development of a comprehensive, EBM plans for all fisheries in an ecosystem. EBM is increasingly recognised as providing the principles and methodology for area-based management or marine spatial planning for all maritime users. Whilst the late nineties also saw the blossoming of 'Oceans' approaches aimed at developing and applying EBM principles to multiple sectors in multi-stakeholder processes, the gradual pace of these reforms and their perceived expense has meant that few have been implemented. The South East Regional Marine Plan in Australia and the Benguela Current Commission are the two successful examples. What is evident, however, is that these processes are needed to implement comprehensive marine-protected area networks and to restructure fisheries, and this remains a key political challenge.

Our analysis is based on the jurisdictional role of countries, while an alternative approach would focus on the undoubtedly differing performances of individual fisheries in achieving EBM, but this approach would take a lot of resources to develop a global picture. Overall, however, our EBM performance ratings correlates quite well with UN Human Development Index (HDI, Fig. 4), although the correlation is not a strong one (COD = 0.29%, $P < 0.01$). This creates a considerable challenge for international agencies, governments and NGOs that wish to encourage the adoption of EBM.

References

[1] US Commission on Ocean Policy. An ocean blueprint for the 21st century. Final report of the USA Commission on Ocean Policy to the President and Congress, Washington, DC, USA, 2004.
[2] Hall SJ, Mainprize B. Towards ecosystem-based fisheries management. Fish and Fisheries 2005;5:1–20.
[3] Pikitch EK, Santora C, Babcock EA, Bakun A, Bonfil R, Conover DO, et al. Ecosystem-based fishery management. Science 2004;305:346–7.
[4] Hilborn R, Punt AE, Orensanz J. Beyond band-aids in fisheries management: fixing world fisheries. Bulletin of Marine Science 2004;74(3):493–507.
[5] Brodziak J, Link J. Ecosystem-based fishery management: what is it and how can we do it? Bulletin of Marine Science 2002;70(2):589–611.
[6] FAO. Code of conduct for responsible fisheries. Rome: FAO; 1995. p. 41.
[7] FAO. The ecosystem approach to fisheries. FAO technical guidelines for responsible fisheries, vol. 4 (Suppl. 2). Rome: FAO; 2003. p. 112.
[8] Collie J, Gislason H. Biological reference points for fish stocks in a multispecies context. Canadian Journal of Fisheries and Aquatic Sciences 2001;58:2167–76.
[9] Caddy JF, Mahon R. Reference points for fisheries management. FAO Fisheries Technical Paper, vol. 347, 1995, 83p.
[10] Link J. Translating ecosystem indicators into decision criteria. ICES Journal of Marine Science 2005;62:569–76.
[11] Rice JC. Evaluating fishery impacts using metrics of community structure. ICES Journal of Marine Science 2000;57:682–8.
[12] Ward T, Tarte D, Hegerl E, Short K. Policy proposals and operational guidance for ecosystem-based management of marine capture fisheries. World wide fund for nature. Sydney, Australia, 2002, 80p. (Although developed by WWF Australia, this framework is an WWF International fisheries policy paper).
[13] Pitcher TJ, Kalikoski D, Pramod G, editors. Evaluations of compliance with the UN code of conduct for responsible fisheries. Fisheries Centre research reports, vol. 14(2), 2006, 1191. Also available at: ⟨http://www.fisheries.ubc.ca/publications/reports/report14_2.php⟩.
[14] Coremap. Penelitian Terumbu Karang Tingkat Lokal, Kab. Raja Ampat, Provinsi Irian Jaya Barat. Sekretariat COREMAP Tahap II WB, Dinas Perikanan dan Kelautan, Kab. Raja Ampat, Indonesia, 2005, p. 75 (in Bahasa).
[15] Ainsworth CH, Varkey DA, Pitcher TJ. Ecosystem simulation models of Raja Ampat, Indonesia, in support of ecosystem-based fisheries management (pp. 3–124). In: Pitcher BM, Pitcher TJ, editors. Ecological and economic analyses of marine ecosystems in the birds head seascape, Papua, Indonesia: II. Fisheries Centre research reports, vol. 16 (1), 2008, 186p.
[16] Varkey D, Pitcher TJ, Ainsworth CH. Ecosystem based management: the influence of a project in Raja Ampat, Papua, Indonesia (pp. 165–175). In: Pitcher BM, Pitcher TJ, editors. Ecological and economic analyses of marine ecosystems in the birds head seascape, Papua, Indonesia: II. Fisheries Centre research reports, vol. 16 (1), 2008, 186p.
[17] Kruskal JB, Wish M. Multidimensional scaling. Beverley Hills, CA, USA: Sage Publications; 1978.
[18] Pitcher TJ, Preikshot DB. Rapfish: a rapid appraisal technique to evaluate the sustainability status of fisheries. Fisheries Research 2001;49(3):255–70.
[19] Kavanagh P, Pitcher TJ. Implementing microsoft excel software for rapfish: a technique for the rapid appraisal of fisheries status. Fisheries Centre research reports, vol. 12 (2), 2004, 75p.
[20] Bianchi G, Cochrane KL, Vasconcellos M. Implementing the ecosystem approach to fisheries. In: Wrammer P, editor. Fish, trade and development. Royal Swedish Academy of Agriculture and Forestry, 2008, (in press).