Scales for scales

An open look at the open sea

James A. Rising

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ABSTRACT

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Fisheries are among the most complex and tightly coupled social-ecological systems. This thesis develops new perspectives on the spatial features of fisheries, and on common pool resources in general. The central model of the work is the Distributed Commons, a commons spread across space with local and cross-boundary interactions. The model is founded in evidence from historical analysis and complexity theory, and offers insights for management and broader sustainable development policy. The second part of the thesis uses empirical analysis, applying Bayesian and econometric techniques, to study the spatial features exposed by the model. Finally, a computational model is calibrated for exploring the consequences of this theory through experiments. The implications of the Distributed Commons model are relevant to many areas of sustainable development, including atmospheric pollution, environmental degradation, and the use of ecosystem resources.

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For my mother, whose love never wavered.

Preface

This thesis is the first of five planned books intended to consolidate my work and progressively building a framework for understanding the connections between the food system, biodiversity, policy, and human interactions. Tentatively, the remaining volumes are:

Crop Circles: Dynamic Feedbacks in Agriculture

> Combines statistical and biological models for investigating the impacts of weather shocks on agriculture, and studies the spatially correlated and temporally extended effects of the ENSO cycle.

Growing Models: A New Foundation for Food System Modeling

> Applies insights from computer science to scientific research in the realm of agricultural modeling, by describing a new structure for distributed modeling.

Eating away at the Edges: Integrated Modeling of Food System Choices

> Develops a spatial, dynamic, multi-scale, data-driven, and policy-explicit model of how the food system has evolved and will continue to evolve.

Planet on a Plate: Protecting Biodiversity through the Food System

> Studies the interconnections between biodiversity and agriculture, and how biodiversity can be enhanced through policy choices.

The ultimate goal of this research agenda is to provide new tools to inform policy for the global food system, while building an approach that can be applied to a wide variety of other systems and inform the scientific approach generally. To do so, three major challenges must be resolved at the leading edge of both theory and practice.

The first challenge is to identify better approaches for bringing empirical data into the study of integrated systems. The over-determined nature and endogenously driven
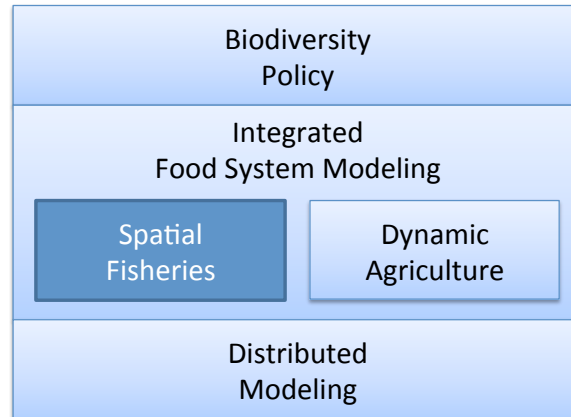
Figure 1: Conceptual relationships between the five proposed books. The spatial fisheries and dynamic agricultural studies (representing books 1 and 2) are building blocks for the integrated model (book 4). However, before this model is developed, the tools for distributed modeling (book 3) form a foundation for it. Finally this model is used to inform broader questions of biodiversity policy (book 5).

dynamics of integrated systems makes data difficult to analyze and interpret. This is a central concern in this volume, where exogenous forcings, statistical models, and process models are all brought to bear on disentangling the effects of fisheries ecology and fisheries management.

The second challenge is to provide frameworks for developing detailed and comprehensive models. Models that include many elements, feedback loops, and ultimately different perspectives, are important both for understanding policy implications and to support the advancement of knowledge within the scientific community. This volume uses a number of tools designed with this concern at the fore, but the ultimate solutions to this problem will not be explored until volume three, under the heading of distributed modeling.

The third challenge is to bring together empirics and distributed modeling, and to develop a truly broad and insightful model of the global food system. While this will be the central focus of the fourth volume, the foundations are laid in this work. The models of spatial dynamics refined here provide a basis to ensure that the strong link between the natural world and the benefits it provides to humanity is represented in future research.

Chapter 1

Introduction

1.1   A New York story

> Boats crossing the bay were escorted by schools of playful whales, seals, and
> porpoises. Twelve-inch oysters and six-foot lobsters crowded offshore waters,
> and so many fish thrived in streams and ponds that they could be taken by
> hand.

These words describe New York Bay, the ocean entrance to what would become New York City, as experienced by early explorers in the 17$^{\text{th}}$ century (Burrows and Wallace, 1998). By 1900, the city's marine waters were a vast dead zone (Jackson *et al.*, 2011, p. 195). Landings of oysters in New York State declined from over 6000 MT in 1887-1926, to 45 MT in 1967 (McHugh, 1972), and lobster throughout the region is now considered unsafe to eat (NYS DOH, 2015).

Many fisheries in this area share similar histories of degradation, which are further shared by fisheries in many regions of the globe. The case of hard clams in the nearby Great South Bay of Long Island offer an example of these trends, and set a scene for the broad themes of this dissertation. The hard clam (northern quahog) received its scientific name, *Mercenaria mercenaria*, from its common use among the local Lenape tribes, famously for money. By 1950, after the decline of the more profitable oyster fishery due to overfishing, Great South Bay provided over 50% of the United States supply of hard clams (McHugh, 1991). But

THE OPEN SEA.

Figure 1.1: Reproduced from Kirby (1884, p. 25).

in 2005, landings in Great South Bay were about 1% of their 1967 peak (Kraeuter *et al.*, 2008).

Overfishing and pollution drove this decline, but other features of the system reinforced the trend in ways that are relevant to this study. Pressure on the clam has precipitated a regime shift in the Great South Bay ecosystem. When clams were in high abundance, they could filter 40% of the water in the Bay daily (Kassner, 1993). Overharvesting of clams decreased this capacity to filter the water, leading to algal blooms and an increased dominance of lower trophic-level species. At the same time, the elimination of top predators allowed the clam's predators, such as starfish and blue crab, to proliferate (Polyakov *et al.*, 2007). Finally, migratory species provided an important connection between the open ocean and the bay, which weakened as these species also declined (Nuttall *et al.*, 2011). In short, the decline of the clam fishery was a combined consequence of fishing effort and internal system feedbacks, in which ecosystem interactions and cross-boundary dynamics play a central role.

The core solution explored by this dissertation is spatial management, which is also an integral part of the clam fishery's story. Starting in 2002, the Nature Conservancy (TNC) purchased underwater rights to 22% of the Great South Bay. Within the area, TNC constructed 70 spawner sanctuaries, each a 0.4 Ha marine reserve stocked with adult clams. Rather than relying on 24-hour patrols, TNC developed partnerships with local communities and a system of incentives. As a result, in 2008, juvenile clams soared 4000 % throughout the central Great South Bay region (LoBue and Udelhoven, 2013).

While the story of the Great South Bay clam is remarkable, its features are common and addressed throughout this dissertation. Overfishing is a common feature of modern fisheries, and some of the responsibility can be placed on historical approaches to fisheries management, as discussed in chapter 2. The central premise of this thesis is that fisheries are best thought of as spatial bioeconomic systems, and that systemic feedbacks drive their behavior. More specifically, these spatial dynamics have cross-scale features, as shown in the emergence of a regime shift precipitated by losses of clams and explored in chapter 3. Cross-boundary issues, such as the effect of migratory fish on clam ecosystems, are an immediate consequence of this perspective, and explored in chapter 4. Species interactions are

also central to the dynamics of the system, as with clams and their interactions with plankton and predators, and inferring their characteristics from data is an important problem studied in chapter 5. One beneficial consequence of cross-boundary effects and ecosystem interactions is the powerful potential of marine reserves, like the TNC's, and the extent of this potential is studied in chapter 6. Within this context, spatial management methods like TNC's have considerable potential for intervening in the system, and experiments to understand these opportunities better are studied in chapter 7.

1.2 Big fish

Fisheries are among the most economically and socially important, ecologically diverse, and chaotically dynamic systems on the planet. With the popularity of books like *Cod: A Biography of the Fish that Changed the World*, and the effectiveness of campaigns by Greenpeace and the Monterey Bay Aquarium, it hardly seems necessary to point out the importance of our oceans. These motivations tend to fall into two camps, to which I will suggest a third.

First, fisheries form an important contribution to the global food system. Fish provide roughly 20% of global protein consumption (Hartmann, 1994), and are an essential source of protein for many coastal communities. Wild-caught fish account for half of global fish harvests, and the remaining half produced by aquaculture is vitally supported by 22% of wild catch (Tuominen *et al.*, 2003). The contribution of fisheries to food will become more pressing as the global population and food demand increase (Tilman *et al.*, 2011). Fish can continue to be an important source of nutrients even as populations increase, but only if fish resources are sustainably managed (Merino *et al.*, 2012).

Fisheries are the bedrock of communities in many regions, particularly in many developing countries. Wild-catch fisheries and aquaculture can either support economic growth if properly managed (Kent, 1997, World Bank, 2005), or cause deep disruptions when ecosystem services are degraded (Cullather, 2010). Proper nutrition is essential for health and economic growth in developing countries (Dixon *et al.*, 2001), and a productive food sector supports the structural changes which drive industrial development (Rostow, 1990, Diamond

and Ordunio, 1997, World Bank, 2007).

Second, marine ecosystems include some of the most vibrant remaining biodiversity hotspots on earth, a source of intrinsic value. Mesopelagic fish alone likely represent 11 000 to 15 000 million tons of the biomass of the ocean (Irigoien *et al.*, 2014), dwarfing an estimated 960 million tons of land vertebrates (Smil, 2003, p. 186). While the biodiversity of the oceans is much less well understood than that of land, it has long been a source of fascination, as the wood cuts used throughout this thesis from Kirby (1884) show. Food webs in marine ecosystems are also longer and more complex than terrestrial food webs, spanning seven orders of magnitude in species size (Azam *et al.*, 1983). The slower impact of humanity on marine ecosystems has resulted in an estimated 36 times fewer anthropogenically influenced marine extinctions than terrestrial ones (McCauley *et al.*, 2015).

Third, the interaction between fishing communities and vibrant ecosystems provides a perfect context to explore the consequences of complexity in sustainable development. Fisheries are often treated as the archetypical commons, and produce a "social trap" where short-term motivations for individual fishers are at odds with the long-term interests of the fishing community (Anderson and Seijo, 2011). This feature is shared by global atmosphere and many other social-ecological systems, and solutions for better fisheries management have implications for a broad range of common resources. Fisheries are also a prime example of our reliance on ecosystem services, and one of the few areas where humans depend explicitly on wild ecosystems. Fisheries are among the most tightly coupled of all social-ecological systems, where slight changes in management can quickly produce significant consequences and win-win management is possible but often lost. And finally, the open ocean could be a poster-child for uncertainty due to both chaos and our lack of knowledge about their state and dynamics. All of these issues are central to sustainable development, and finding better opportunities in fisheries can facilitate progress in many other areas. Fisheries provide a vibrant case study for the development of new bioeconomic perspectives and models.

The importance of fisheries is all the more significant because of the rate at which they are being lost. An ongoing debate looks at the evidence for fisheries collapse (Worm *et al.*, 2006) against the weight of uncertainty and examples of effective management in developed countries (Costello *et al.*, 2008, Pauly *et al.*, 2013). While there is no consensus, and this

work will not contribute directly to that debate, a few points seem clear.

Many regions have experienced significant and long-term loss of biomass and biodiversity. An estimated 63% of assessed fish stocks are in need of rebuilding (Worm *et al.*, 2009). Sharks and rays (Dulvy *et al.*, 2008), groupers (Sadovy de Mitcheson *et al.*, 2013), diadromous fish (Limburg and Waldman, 2009), and marine mammals (Davidson *et al.*, 2012) are all at heightened risk of extinction. Marine biodiversity is also threatened by invasive species, which continue to spread around the globe (Bax *et al.*, 2003).[1]

In many cases these trends have coincided with regime shifts. A regime shift occurs when a system enters a new stable state. Among fisheries, this new, post-disruption state is often characterized by reduced biodiversity and the abundance of a few low trophic-level species, such as algae and jellyfish, at the expense of higher trophic levels (Folke *et al.*, 2004). Although this process is not clear in the trends of the trophic levels of global catches (Branch *et al.*, 2010), historical ecology suggests that shifts in trophic levels are already long entrenched (Jackson *et al.*, 2001, Sumaila and Pauly, 2011). Importantly, once a regime shift occurs, reversing it can be very difficult.

The stressors on marine ecosystems have continued to expand. These include temperature increases, implicated in coral bleaching events (Lesser, 2011), and ocean acidification (Kroeker *et al.*, 2013), both of which are driven by anthropogenic climate change, and are much better understood than a decade ago. Also, the extent of fishing has continued to increase, in the Pacific (Berkes *et al.*, 2006), Atlantic (Alder and Sumaila, 2004), and globally (Watson *et al.*, 2013).

Fisheries management is intended to safeguard against over-exploitation, to minimize the possibility of collapse and maximize the potential of the fishery. All fisheries management aims to control how much of which resources are harvested, whether through a scientifically set quota or by alleviating impacts on non-target species and specific age ranges through gear management and seasonal closures. These approaches have clear benefits over an absence of management and can stop fishery collapse in many cases (Costello *et al.*, 2008). Many developed countries now use robust and active management practices, and global

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[1]San Francisco Bay harbors over 200 invasive species, and New York Bay probably has a similar number (Ruiz *et al.*, 1997).

fisheries production appears to have stabilized.

However, the situation for many areas continues to worsen, and this dissertation is motivated by this continuing crisis. These failures stem from many sources. Strong drivers foster the conditions that lead to over-exploitation. Uncertainty in the biological state of the stock, the rest of its ecosystem, and the true extent of fishing make it difficult to set catch limits. An accidental over-estimation of the stock can lead to a vicious cycle of smaller breeding populations and louder lobbying from fishing communities fighting for their immediate needs. Maintaining stocks at their optimum has been likened to "keeping a marble on top of a dome fastened to the deck of a rolling ship seen through salt-sprayed goggles" (Roughgarden and Smith, 1996).

Fisheries are extremely complex social-ecological systems, operating across spatial scales spanning from tenths of micrometers to hundreds of kilometers (Azam *et al.*, 1983) and impacted by both tiny fishing communities and vast industrial fleets. Fish stocks naturally have erratic population fluctuations that span decades, with wide variation of stock levels observed in the natural record of ocean floor deposits (e.g., Baumgartner *et al.*, 1992). They are impacted by a wide range of human impacts outside of fishing, including climate change, ocean acidification, dead-zones, pollution, and invasive species movements.

On top of this, management practices vary widely. While many developed regions and some of the most productive fisheries have strong management, around 80% of global catch comes from developing regions (Hance). Few fisheries in the developing world have the capacity to perform stock assessments at all, much less on a regular basis. Few have the capacity to enforce catch limits informed by recent assessments, or to enforce protected areas. Few have open and trusted communication channels between fishing community, scientists, and external government managers. Fishing communities have local knowledge that is difficult to incorporate into region-wide management. Under the stress of falling catches and increasing costs, they oppose policies that would further reduce their catch, particularly given the erratic history of management.

1.3 Fish in space

This thesis focuses on the importance of spatial understandings of fisheries, a perspective that is growing in prominence. Fisheries are no longer alone in their demand on the oceans resources. Wind farming, oil exploration, fiber optic lines, and new political jostling is threatening to carve up what is left of the oceans once unfathomable abundance.

A growing number of economists and scientists now believe that carving up the oceans– an approach called spatial management– may be the most effective way to balance these competing demands. Designating a kind of property rights on segments of ocean can ensure that regulated profit and marine conservation both have their space.

This thesis argues that if we are to embrace this approach, we need new institutions of spatial management, because enclosure as it was performed on land will result in even more dysfunctional relationships with the underlying environment than those that the land enclosure process produced.[2] The spatial features of fisheries have long been underappreciated. This has been as much due to our traditionally impoverished conceptual models as our lack of information. The traditional economic view of a fishery is as a single, well-mixed stock, for which spatial boundaries are meant to describe hulls within which populations never leave (e.g. Gordon, 1991). Even where newer multispecies ecosystem models are used, the potential for effects from outside the fishery boundary is ignored (e.g. May *et al.*, 1979). We need new paradigms of spatial management that embrace the fluidity of the marine medium. Creating these new paradigms requires a closer look at the spatial features of fisheries. Recent work that studies spatial features of fisheries finds that space matters in a number of surprising ways.

New studies of fish movements have found many species to both have more site fidelity than expected, and travel much further than expected. Block *et al.* (2011) finds that top predators tagged within the California Current large marine ecosystem often traveled more than 2000 km from it, but also returned to it with over 40% of species spending more than 80% of their time there. While individuals of some species always stay within a "home

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[2]Land enclosure has both benefits and costs, but in many cases has disrupted social-ecological systems that supported communities in with sophisticated institutions. For a survey of the consequences of land enclosure, see Scott (1998).

range" (e.g. Booth *et al.*, 2013), the extent of forays taken by other species from their home ranges can be idiosyncratic to the individual fish (Meyer *et al.*, 2000). These studies has been used to show that even very small MPAs can be effective, but also that the movement occurs within much larger "meta-populations" (Grimm *et al.*, 2003).

The spatial connectivity of a species population is an important ingredient in resilience and population health. As individual patches of a fish environment become degraded, or the path between them is obstructed, the resilience and long-term survivability of the entire population decreases (Hughes *et al.*, 2005). Spatial features can also hasten collapse among species with high levels of site-fidelity (Elmhirst *et al.*, 2009). Even if the total stock across a region is high, smaller areas can experience functional extinction of a species, resulting in a regime shift where the species can no longer attain its previous position within the ecosystem.

A recent literature has emerged around spatial bioeconomic models, finding that proper management often relies on understanding the spatial features of both the ecological and economic systems. Much of this literature has emphasized the differences in optimal management with and without space Smith and Wilen (2003), Sanchirico and Wilen (2005).

However, spatial fisheries management can also offer new opportunities. Marine spatial planning (MSP) has risen to prominence in the past 15 years as an essential part of the fisheries management process (Douvere, 2008). This reflects the increased possibility of conflict in the seas across its multiple uses, as well as the new emphasis on ecosystem-level management. Understanding the short-term and long-term impacts of marine spatial planning on stakeholders can improve this process (Smith *et al.*, 2010).

Marine protected areas (MPAs) are one of the most common forms of spatial management, where setting aside even small areas from fishing can produce benefits throughout a region. No-take marine reserves are like national parks, but with a crucial difference. In an ocean pushed to its limit, excluding areas from exploitation can actually result in more resources for everyone. I study these effects in chapter 6.

Unlike catch limits, which are a species-specific intervention, marine protection is an intervention for the entire ecosystem. MPAs allow ecosystems to recover, but the benefits of protection stem more from what we do not know than from what we do know. If we

could count the fish in the sea, and predict the consequences of fishing them down to any given level, quotas would capture the requirements of management perfectly.

In contrast, it is difficult to target or even to predict what the consequences of protection will be. In the absence of human intervention, species dominance can shift in complex ways. Invasive species can expand, and new top predators can find an attractive home. In addition, the effects of pollution, eutrophication, and climate change cannot be excluded. Despite these many uncertainties, marine reserves have been shown to greatly benefit fish stocks. A global study of the effects of reserves found that, on average, fish biomass increases 450%, fish size increases 28%, and species richness increase 21% (Lester *et al.*, 2009). While some fisheries saw decreases in each of these, the general result is robust across the vast majority of studies and regions.

However, marine spatial planning is complicated by two effects that have recently become more clear. Fish populations are not stationary, and climate change is causing even general ranges to shift. Populations in temperate regions have moved toward the poles in both the northern (Doney *et al.*, 2012) and southern (Last *et al.*, 2011) hemispheres. As different species respond differently to climate change, and do so at different stages in their life cycle, the potential for ecosystem disruption increases (Edwards and Richardson, 2004).

Population shifts are one form of cross-boundary effects, which are becoming a greater concern as the spatial extent of fish interactions is mapped. Fish do not respect political boundaries, so one country can impact another through its fisheries management. These "straddling stocks" have become a growing source of conflict (Miles and Burke, 1989). Industrial and local communities also experience cross-boundary conflict, as fleets of industrial ships fishing for the international market can deplete an entire region's resources.

1.4 Fishing for answers

This thesis seeks a deeper understanding of the opportunities for spatial policy in fisheries. Two main threads are intertwined. First, I seek to contribute to an understanding of spatial management that goes beyond the creation of area closures and ownership. Second, I want to engage seriously the issue of cross-boundary flows, as both a consequence of spatial

perspectives and an opportunity for benefits and conflict.

New approaches in data analysis and computational modeling have much to inform these issues. Progressively more data is becoming available on the spatial features of fisheries. Fisheries scientists need models incorporating this new knowledge and capturing its dynamics.

I present an approach peppered with computational models. The chapters below include a Bayesian MCMC, discussions of cellular automata, a network dynamics model, a physically-based diffusion model, and an agent-based model. These methods are by no means the purpose or motivation for the work presented here, but they do play an important role. While analytical and statistical models (which are also applied below) can provide exact solutions, computational models are mostly beneficial for their dynamics and simulations. They allow us to incorporate a broader range of factors, and make sense of the behaviors that result through their use in multiple experiments and under different assumptions.

Investigating spatial interactions begs for a revisiting of the idea of the commons and its application to fisheries. Fishing areas are often considered quintessential common pool resources, but the cross-boundary effect and complex dynamics described above require a more nuanced vision. A central goal of this dissertation is to fill out such a vision.

The first part of the thesis explores qualitative arguments around spatial commons, culminating in a new conceptual model of the "Distributed Commons". The second part seeks to ground this model in empirical analyses, which together inform a concrete, computational model of the Distributed Commons presented in the last chapter. This final model provides new opportunities for many experimenting with policy changes and applications to specific contexts.

Developing a new conceptual model of the commons requires revisiting the origins of the current one. The second chapter describes the development of the concept of the fishery commons through the history of the World Bank fishery projects.

The third chapter considers the large-scale, emergent consequences of local interactions in fisheries. To do so, it studies the model of self-organized criticality and looks at its past and potential treatment in fisheries.

The fourth chapter develops a theoretical model of distributed commons, which provides a new perspective on spatial fishery management. This model expands the idea of a commons, and builds on a broader understanding of cross-scale and cross-boundary interactions. This model will be returned to from a computational perspective, after the a number of other techniques are applied to ground the model in data.

Starting the second part, the fifth chapter studies the use of catch-only models for studying spatial movements of fisheries. These results will be used to parameterize fish diffusion in the computational model.

The sixth chapter analyzes the benefits of marine protected areas for exploited fisheries. This will draw upon the fish population distributions from appendix 2 and the logistic model studied in chapter 4. The results will further inform the way that unfished areas affect fished areas in the computational model.

The seventh chapter will present and calibrate the computational model of distributed commons, and perform a number of experiments considering different features of it.

Combined, these perspectives provide a rigorous sketch aimed at informing the evolution of fisheries management. Fisheries science has been engaging progressively more with these features: the systemic feedbacks between fishers and ecosystems, the spatial structure of fishery systems, and the management implications of these features.

There remains a gap between the material studied here and active fisheries management. The political economy of management decisions is only considered in a theoretical light, and the models of fish ecology are consistently simplistic. It remains a task for future work and the broader management community to understand how applicable these results and models are to real contexts.

At the same time, as studies of spatial features in tightly coupled socio-ecological systems, these studies have applicability beyond just fisheries. Each chapter has two parallel aims: to inform fisheries policy and to identify broader implications for spatial complexity in sustainable development. In this context, the simplicity of the models used here is a strength, lending them to general results and methods for non-fisheries research. It is my hope that the insights drawn here from fisheries help inform sustainable development in many other contexts.

We are living through a global crisis in fisheries management. Increased demands on fisheries now impact every corner of the ocean, with collapsing fish populations in many regions. Tradable catch shares have grown to well-justified prominence just as the single-stock approach that underlies them is being undermined. This year, the US's landmark fishery management policy, the Magnuson-Stevens Act, is being debated in congress to be reauthorized and amended, amidst a confusing mix of skepticism and institutional learning, of failures and successes. I believe that the Distributed Commons has much to inform this debate, and paves the way for a much longer scientific inquiry into the spatial domain in social-ecological systems and sustainable development.

Part I

Theoretical Foundations

Chapter 2

Creating the Fishery Commons

IMAGINARY CUTTLE-FISH TAKING HOLD OF A SHIP.

Figure 2.1: Reproduced from Kirby (1884, p. 68).

This chapter provides a close inspection of the conceptual foundations of fisheries management. One of the central goals of this thesis is to develop new understandings of fisheries,

and common resources in general, to open up new opportunities for research and policy. The economic perspective on fishery commons developed out of a historical process which continues to shape the ideas in this work. Better understanding that process can inform new conceptual opportunities.

The tragedy of the commons is one of the most pervasive and insidious problems in social science. Forests, fisheries, clean water, and greenhouse gases, for example, can each act as open-access resources fraught with difficulties related to the tragedy of the commons. According to both economic theory and numerous case studies, many of the commons in the international arena and in developing countries are particularly difficult to manage, due to weak ownership rights, unpredictable dynamics, and widely dispersed resources (Ostrom, 2009).

Since its introduction by Garrett Hardin in 1968, the concept of the tragedy of the commons has evolved and achieved a central role in resource management. In 1979, according to Godwin and Shepard, the commons was "the dominant framework within which social scientists portray environmental and resource issues." Thirty years later, the World Development Report of 2010 claimed, without further justification, that "climate change is a crisis of the commons" (World Bank, 2010). However, the hegemony of the commons idea is debatable. Political scientists critique the commons as a vast simplification, inappropriately abstracting across diverse resource types and institutions (Godwin and Shepard, 1979). Historians point out that its founding myth is based on faulty history (Cox, 1985). Elinor Ostrom and others, through case study work, have identified countless local experiences that are inconsistent with the commons story, due to diverse local institutions that support the sustainable management of common resources (Ostrom, 1990, Dietz *et al.*, 2003). This has led to backlash against Hardin's characterization, reframing the discourse as a "comedy of the commons" and arguing that both commons and their management opportunities provide intrinsic benefits (Rose, 1986, McCay, 1996). Even so, there remains a consensus that although "communal property" regimes can viably support sustainable resource usage, "open-access" regimes cannot (Feeny *et al.*, 1990).

In this article, I will explore how this paradigm achieved its current position by examining the World Bank's role in fisheries development. Fisheries are of interest both as a tool

for economic development and as a key concern for sustainability science. They have potential for helping poor communities and supporting country-wide nutrition. Unfortunately, they exhibit many characteristics where tragedies of the commons would be most expected, such as pervasive uncertainty and mobile stocks (Schlager, 1994), strong outside pressures (Alder and Sumaila, 2004), and large and poorly-constrained communities (Ostrom, 2009). Globally, fisheries show a startling mix of robust traditional management practices and devastating collapse. Better understanding of the role of the concept of the commons in the formation of institutions can both help explain this variation and improve planning for the future.

This paper makes some assumptions concerning a partial theory of history. It holds that society, through its actors and institutions, is constantly creating ideas that are simultaneously shaping the perspectives taken by their adherents. As the West developed its idea of "the commons," it changed how it advised other countries to manage their "open resources". This economics-motivated vision of the commons is limited, and management through it can be culturally and environmentally disruptive. This paper does not attempt to analyze the empirical power of the idea of the commons. Tragedies of the commons frequently arise in the realms where economic theory predicts them. However, it is unclear how many of these problems result from the perspectives that resource management institutions bring to their tasks. By understanding the blind-spots inherent in our ideas of the commons, we can recognize greater possibilities for managing these systems.

The key historical agent in this study is the World Bank's rural development groups, through the concerns and designs of their projects and loans as described in World Bank documents on early fisheries projects, between 1963 and 1994. These documents are listed in table 2.1, along with citation names used throughout the paper.

The first section provides an overview of the intellectual developments that followed Hardin's work. The second section provides a framework for understanding the history of fisheries projects at the World Bank over this period. These two histories are then juxtaposed to explain the shifts in awareness of commons problems and the debate concerning the role of equity, in sections three and four. The documents concerning fishery projects in Panama are used as a case-study of these shifting concerns in section five.

Year	Report or Discussion Paper	Citation
1956	Iceland: Mission report	WBS Iceland 1956
1963	China - Deep-Sea Fisheries Project	WBP China 1963
1966	Korea - The economy (Vol. 4 of 8): Fisheries	WBS Korea 1966
1967	China - Second Deep-Sea Fisheries Product Project	WBP China 1967
1969	Ghana - Fisheries Project	WBP Ghana 1969
1970	Senegal - The current economic situation and prospects (Vol. 4 of 7): Fisheries	WBS Senegal 1970
1971	Panama - Fisheries Project	WBP Panama 1971
1971	Mauritania - The current economic situation and prospects (Vol. 3 of 4): Fisheries	WBS Mauritania 1971
1973	Iceland - Fishing Harbors Rehabilitation Project	WBP Iceland 1973
1975	Panama - Fishing Port Project	WBP Panama 1975
1977	India - Gujarat Fisheries Project	WBP India 1977
1977	Panama - Second Fisheries Project	WBP Panama 1977
1978	India - Andhra Pradesh Fisheries Project	WBP India 1978
1979	People's Democratic Republic of Yemen - Second Fisheries Development Project	WBP
1979	Maldives - Fisheries Project	WBA Maldives 1979
1979	Panama - First Fisheries Project Audit	WBA Panama 1979
1979	Philippines - Fishery Training Project	WBA Philippines 1979
1980	Yemen Arab Republic - Fisheries Development Project	WBP Yemen 1980
1980	Rethinking artisanal fisheries development: Western concepts, Asian experiences	WBR 1980
1983	Maldives - Second Fisheries Project	WBP Maldives 1983
1983	Panama - Second Fisheries Project Audit	WBA Panama 1983
1984	Somalia - Fisheries Exploration - Pilot Project	WBP Somalia 1984
1987	India - Gujarat Fisheries Project	WBP
1989	Tunisia - Second Fisheries Project	WBP
1992	Fisheries development, fisheries management, and externalities	WBR 1992
1992	Maldives - Second Fisheries Project Completion Report	WBP Maldives 1992
1992	Yemen - Fisheries Development Project Completion Report	WBA Yemen 1992
1994	Managing fishery resources: proceedings of a symposium	WBR
1994	World fisheries: avoiding a natural resource disaster	WBR
1994	Mauritania - Fisheries sector review	WBS Mauritania 1994
1994	Managing redundancy in overexploited fisheries	WBR
1994	Fisheries in the Maldives and Yemen	WBA Maldives/Yemen 1994
1995	Fisheries Subsector Capacity Building Project - Development Credit Agreement	WBR
1996	Managing transboundary stocks of small pelagic fish: problems and options	WBR

Table 2.1: World Bank published documents concerning fisheries, 1956 - 1996. The documents are classified by their citation as follows: WBS are economic or sector reports, WBP are project reports, WBA are project audits, and WBR are staff discussion working papers and institutional reports. The project reports are often treasure troves of supplemental documents, including descriptions of concurrent UNDP/FAO projects and institutional structures, and documents from national institutions.

2.1 A short history of tragedy of the commons

Concern over the management of the commons has had a long history in the West.[1] Hardin was inspired by Lloyd (1833), who asked, "Why are the cattle on a common so puny and stunted? Why is the common itself so bare-worn and cropped so differently from the adjoining enclosures?" Within fisheries, the debate reaches back at least to the identification of the problem in Grotius, Mare Liberum (lit. The Free Sea, 1609), and the political and economic solution in Selden, Mare Clausum (lit. The Closed Sea, 1635). During the 1950s, seminal works applied economic theory to fisheries, and argued that severe inefficiencies are inevitable without property rights (Gordon, 1954, Scott, 1955, Schaefer, 1957).

The "tragedy of the commons" was coined by Hardin in 1968:

> Picture a pasture open to all. [A group of herdsmen uses the pasture, and it initially provides more than enough for all of their sheep.] ... The rational herdsman concludes that the only sensible course for him to pursue is to add another animal to his herd. And another.... But this is the conclusion reached by each and every rational herdsman sharing a commons. Therein is the tragedy. Each man is locked into a system that compels him to increase his herd without limit– in a world that is limited. ... Freedom in a commons brings ruin to all.

The necessary elements of a commons tragedy include that (1) the commons are "open": an agent's increased use of it is not restricted; (2) resources are limited: one agent's use of them diminishes the resources available to other agents; (3) agents want to increase their use of it for their individual benefit, despite the collective loss. In the absence of strong management which changes these conditions, the common resource will be excessively depleted. The tragedy of the commons is not simply that it is over-used, but that there is no individual incentive to maintain it.

However, not all commons are subject to the tragedy of the commons. Since Hardin's paper, hidden assumptions and key distinctions have been identified to explain a growing list of common-pool resource with robust management regimes. Ciriacy-Wantrup and Bishop

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[1] Dietz *et al.* (2002) provides a more complete survey of the intellectual history of commons.

(1975) described the difference between open-access regimes (*res nullius*), the classic "free-for-all" situation, and common property (*res communes*). Common property can support a variety of management institutions within the group that shares a given resource. This distinction was further explored by Ostrom and Ostrom (1977) by recognizing a spectrum between high and low "subtractability of use" and high and low "difficulty of exclusion". "Open-access resources" are those with both high subtractability and high difficulty of exclusion, and not all "common-pool resources" are open-access.[2] Levine (1986) argued that the economic theory of the commons is far from self-evident, and its unexpressed assumptions include that "individual rationality [runs] counter to 'group rationality" and that "behavior in a commons system [is] independent of social and historical context". Furthermore, the source and composition of the rights held by resource users varies widely. Institutional arrangements can endow different users with the right to *access* (or enter) a resource, to *withdraw* resource products, to *manage* or alter the resource system, to *exclude* other users, and to *alienate* by selling or leasing other collective-choice rights (Schlager and Ostrom, 1992, Ostrom, 2000).[3]

Concurrent with the development of a better understanding of the commons, many authors investigated policy implications for proper management. The core economic solution to common-pool problems is privatization, later expanded to include a variety of methods of "restricting access and creating incentives" (Ostrom *et al.*, 1999). Modern theory focuses on the variety of factors which influence whether the expected benefits of managing a commons exceed the perceived costs, and on the complex relationship between governments and local management institutions. The interactions between resource users within social-ecological systems and their institutional context are complex and varied, producing a wide variety of potential common property regimes. As a result, no "one-size-fits-all" solution exists for managing the commons (Ostrom, 2010).

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[2]While historically, "common property theory" was the economic study of open-access resources, I will reserve the term "common property" for resources managed by an exclusive group, and use the terms "common-pool" and "the commons" for resources that admit both open-access and common property regimes.

[3]Ostrom (2009) identifies the size of the resource system, its productivity, its predictability, the mobility of its units, the number of users, leadership, norms and social capital, knowledge, importance of the resource, and the ability to create rules as important factors determining the capacity of users to self-organize.

The major approaches to adjusting this cost-benefit balance are closing the commons through property rights, and encouraging self-organization of user communities by enforcing boundaries on the commons. Higher-level institutions can strengthen local ones, through state legitimization, enabling legislation, cultural revitalization, capacity building, and institution building (Pinkerton, 1989, Berkes, 2002).

However, centralization and higher-level management also has the potential to undermine local institutions. The centralization of decision-making, increased role of science and economics, development policies aimed at growth, and market forces have all been shown to undermine local governance of commons (Berkes, 2002). Furthermore, *de jure* regimes can have complex relationships with the *de facto* rules under which resource users act. For example, when many countries nationalized their natural resources in the 1960s, community management practices lost their legal standing and many resources became *de facto* open-access when governments lacked the capacity to manage the resources themselves (Ostrom, 2000). Klooster (2000) makes the role of development organizations explicit:

> Analysts acknowledge that these complex resource management systems are often susceptible to break down following intervention from the state, commercialization, land degradation, population pressures, encroachment, and the expropriation of disproportionate shares of common resources by a few members of the community. A common lament equates the development process with enclosure. The "real tragedy of the commons" is the destruction of common pool resource management systems and subsequent degradation following the intrusion of modernizing states and modern economic relationships.

This dynamic plays a key role in the present study, as the history of the economic development of commons resources has gone hand-in-hand with the undermining of traditional management and subsequent resource collapse. As a central document in this study, WBR 1980, argues, "As commercialization proceeds... restraining norms may collapse, competition may intensify, and fishing effort may escalate. Development projects themselves may accelerate this process."[4]

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[4]An extensive literature discusses open-access resources in international development (e.g., Norton, 2004,

2.2 Fisheries at the World Bank

The World Bank's approach to agriculture in general and fisheries in particular has shifted multiple times in the past 50 years. For clarity, I divide the time between 1963 and 1994 into four periods based on the attitudes as reflected fisheries projects (see figure 2.2). Roughly speaking, the World Bank focused on capital-intensive projects until the early 1970s; then emphasized small-holder poverty during McNamara's tenure as World Bank president; shifted to land reform during the 1980s; and finally integrated a vision fisheries management starting in 1992. This section provides a general overview of these attitudes.
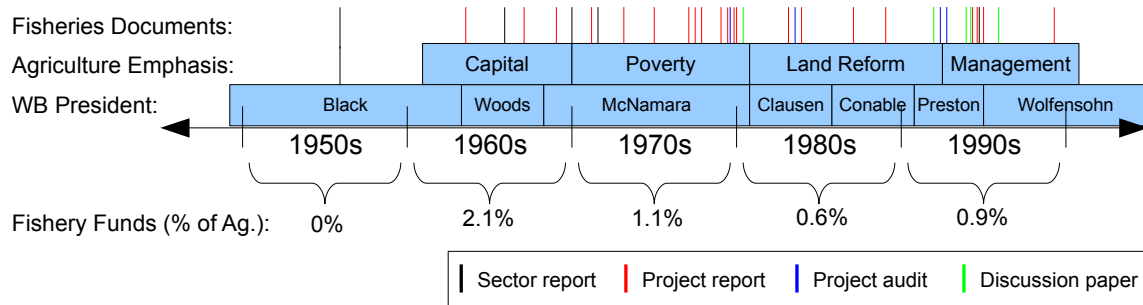
Figure 2.2: World Bank presidents (lower blue bar), attitudes within agriculture (upper blue bar), and fishery projects (vertical lines). The "Agricultural Emphasis" titles are only suggestive simplifications. The bottom row shows the portion of agricultural funding from the World Bank used on fishery projects, from Kapur *et al.* (1997).

The World Bank public archives list only 32 documents on fisheries before 2001, including 15 project reports and 5 fishery sector studies. This reflects the World Bank's early sense of its "comparative advantage" in supporting capital-intensive projects (Kapur *et al.*, 1997). Early fisheries documents display this bias, financing large ship purchases (WBP China 1963, WBP China 1967, WBP Panama 1971) and port infrastructure (WBP Ghana 1969, WBP Iceland 1973, WBP Panama 1975).

The Bank began to push agricultural development in 1963 with the inauguration of George Woods, which coincides with the earliest World Bank fishery project (WBP China 1963). The objective of most early World Bank fishery projects was to support economic growth and increase national fish consumption, in support of national agencies with these

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Dasgupta, 2009).

objectives.[5] Early World Bank documents suggest a "command and control" view of fisheries. Their text focuses on quantities of ships to add or kinds of equipment to modernize (WBP China 1963, WBP Ghana 1969, WBP Panama 1971). These discussions are engaged with government needs, with very little indication of communication with existing fishing communities.

Some word choices from the documents of this period are particularly evocative: projects aim to support the fishing "industry", and see it as a useful "source of animal protein" (e.g., WBP Panama 1971). Management during this time period appears to have been considered a process of managing biomass, rather than managing a fishery system consistent of fishing communities and ecosystems.[6] These assumptions undermined project planning as the World Bank failed to anticipate the role that local communities would have in project implementation, and led to an under-appreciation for the "international dimensions" of fishery projects (WBA Panama 1983). Within agriculture, three groups debated and collaborated on agricultural projects: the technical agriculturalists, generalist operators, and redistributional reformists. WBA Panama 1983 suggests that the lack of integration of different camps is to blame for much of the failure in fishery projects, particularly where engineering expertise and policy expertise where not combined.

During the 1970s, the emphasis displayed in fishery projects shifted to focus on a wider range of needs in poor communities, to "increasing the flow of benefits to the rural population and raising the standard of living of the fishermen's families" (WBA Panama 1983). McNamara's tenure, 1968 - 1981, increased lending to agricultural and rural development

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[5]The growth of productivity in agriculture is considered a foundational stage of economic growth (e.g., Rostow, 1991). Agricultural policies in development have reflected the concerns of many time periods. Near the end of the colonial era, irrigation and other agricultural projects took on a simultaneously massive scale and limited scope, typically aiming at a single cash crop and replacing local institutions (Westad, 2005). In the 1950s African leaders sought an agricultural sector that was "economically strong and politically weak", and developed collectivist farming practices which ultimately failed across the continent (Cooper, 2002).

[6]Another interesting issue surrounds "fishing cooperatives" (e.g., WBS Korea 1966). In modern management literature, cooperatives are considered foundational to solving commons problems. However, in the World Bank documents, the cooperatives are considered solely as serving the economic needs of fishers, such as constructing joint deals with buyers. They seem to not have had a role in reacting to over-fishing problems, although both cooperatives and over-fishing are frequently discussed in the same documents.

projects. In this, McNamara emphasized projects that favored smallholders and the very poor. Kapur *et al.* (1997) asserts that "McNamara's antipoverty agenda delivered resources to a large number of needy people, and many small cultivators achieved lasting gains in productivity."

However, Kapur *et al.* (1997) also critiques McNamara's agricultural and rural development interventions for delaying the development of self-reliant institutions. By providing external support for external policies, the World Bank undermined the potential for self-organization. In this way, the World Bank acted analogously to government agencies intervening into local fishery commons.

World Bank fishery projects in the 1970s had a consistently poor track record for their stated goals (WBA Panama 1983). Of the 7 projects audited before 1982, one was canceled, two were left mostly unimplemented, and three were implemented approximately halfway (the remaining one was completed). The rate of return for the six implemented projects was zero or negative for half, and low and substantially supported by an unintended use of funds in two others. In addition, catches per fisherman are reported to have dropped or otherwise under-performed in four.

Year	Project Name	Completion	Cost Ratio	Returns Ratio	Catches
1968	Ecuador - Fisheries Project	4 of 12 ships	1.02	negative	"much lower than expected"
1969	Ghana - Fisheries Project	10 of 40 ships	0.78	0.2	"limited landings"
1970	Indonesia - First Fisheries Project	incomplete	1.93	negative	"far below targets"
1971	Panama - First Fisheries Project	26 of 40 ships	1.01	0	decreased catch per vessel
1971	Tunisia - First Fisheries Project	187 of 335 boats	0.94	1.2	high prices offset low catches
1973	Yemen - First Fisheries Project	complete	1.48	0.4	(not mentioned)
1974	Iran Fisheries Project	unimplemented	N/A	N/A	N/A

Table 2.2: Project audit summaries from WBA Panama 1983. Cost and return ratios are relative to the expected values at project initiation.

Throughout the World Bank's history, it has "expressed an appreciation of the organizing and efficiency-enhancing capacity of self-adjusting, or only lightly regulated, markets", while recognizing the need for legal supporting institutions and government intervention in cases of market failures (Kapur *et al.*, 1997). Open-access tragedies are one form of market failure, although the aim of the government intervention encouraged by pro-market economists is predominantly to better support market functioning, from which more direct benefits are expected.

The World Bank's advocacy of private ownership has shifted over its existence, centered around a key distinction between private firms and the market. During the 1950s and 1960s, the World Bank aimed its policies at supporting markets composed primarily of private firms. However, McNamara's tenure introduced a decade-long distinction between the ownership and control of industries, recognizing that firm decisions depend more on public and private bureaucrats than legal owners. In light of this, the World Bank recognized that "country mixes of ownership reflected local histories, ideologies, and politics, and, given the liberation of markets, interveners such as the Bank should be slow to second-guess local ownership choices" (Kapur *et al.*, 1997). Some fishery projects in the 1970s, most notably in Panama, operated in the context of an existing history of local management techniques, which the World Bank did not attempt to critique until reviews in the 1980s. However, internally, the World Bank never had a unified perspective, and the markets-ownership distinction was rejected by many of the Bank's staff (Kapur *et al.*, 1997).

This market-ownership distinction was abandoned with the Berg Report, in 1981 (*Accelerated Development of Sub-Saharan Africa*). This report argued that government agencies were inherently inefficient, and privatization of state enterprises became a norm. In agriculture and fisheries, the World Bank, along with many other institutions, began to argue that governments should withdraw from economic management (Kapur *et al.*, 1997). The rise of adjustment lending in the 1980s also drove a decline in money available for fishing projects.

This change was reflected within rural development as a push for decentralization and land reform. Access rights began to be emphasized across all sectors of development, and fishery project reports during this time period show this concern.

Two key ideas are conflated in this shift, as it relates to the commons. Private ownership is held up as a means to secure efficient markets and accelerate development. Private ownership is also upheld within commons, as a solution to an open-access problem. However, the situation is different in these two cases. In the case of agency privatization, bringing an institution into the market with private ownership is said to produce monetary benefits. In the case of commons management, a resource already in a market is in a state of legal ambiguity, and benefits are to be gained by affirming some ownership rights while deny-

ing others. Privatization transfers ownership from a government to a firm, while access restriction transfers ownership from a community to a firm.

This distinction makes the drive for governments to cease their interventions in the commons in the 1980s difficult to interpret. It is unclear if government withdrawal entailed handing resources over to the free-market by means of extensive legal reform, institutionalizing private ownership where it previously was not, or if it resulted in giving local communities freer reign to self-organize. The World Bank documents studied here suggest that the emphasis shifted between these interpretations during this pivotal decade.

By 1993, the need for environmental impact assessments was recognized and institutionalized by the guiding "Agriculture Sector Review" (World Bank, 1993). This report reflects a recognition of past environmental failures, including over-exploitation. The report called for "environmentally friendly technologies and management practices" (World Bank, 1993), and discussed policies aimed at reducing unsustainable pressure on natural resources. At the same time, both the importance of common resource management and a set of economic solutions appears to have become fully established, as reflected in fishery reports at that time. The core principles of this approach are familiar: the need for proactive management of fisheries resources to avoid over-use, and the importance of secure property rights to ensure secure management regimes.

2.3 Commons at the World Bank

The first mention of the commons in World Bank fishery documents is in WBR 1980, a document largely aimed at discussing its implications. The commons, as such, is not mentioned again in project reports until documents in 1992, 1994, and 1996, addressed internationally, discussing fisheries management with view toward access rights. With the exception of the 1980 report, all fishery documents prior to 1992 are nation- or project-specific. The document titles give a sense of their concern over the tragedy. The 1994 document is "World fisheries: avoiding a natural resource disaster", suggesting that bad management was recognized as rampant. In 1996, "Managing transboundary stocks of small pelagic fish: problems and options" identifies fish populations where national property laws

are absent as particularly problematic (a view shared by current marine economists). These documents suggest that by the mid-1990s, the importance of the open-access problem and the economic perspective on it and solution to it had emerged.

Starting in 1994, these ideas began to be incorporated into country projects. WBS Mauritania 1994 highlights four approaches to fishery management: licenses, quotas, taxes, and territorial use rights. The potential for management through traditional methods or through technology and time restrictions as described in Panama reports two decades earlier is entirely ignored.

Figure 2.3 shows some simple word analyses, which I will not dwell upon. Although it is impossible to draw conclusions from them, the trends described in the figure caption corroborate this shifting view of the commons.

The core of commons fishery problem is over-fishing, and the solutions proposed for over-fishing evolved during this time period. Early documents expressed concern over decreases in "catch per vessel" as the number of vessels fishing increased, focusing on the revenue-needs of the fishers. However, reports written before and shortly after Hardin's article consistently expressed uncertainty about whether these decreases represented ecological over-fishing (WBS Korea 1966, WBP China 1967, WBS Senegal 1970), that is, a drawing-down of the resource as opposed to a wider division amongst harvesters. Some reports offered rationales for why over-fishing was impossible (e.g., WBP Panama 1971), and conservation is only mentioned as a strategy for management twice before 1992 (WBS Iceland 1956, WBP Panama 1971). Where over-fishing became a problem, the first solution was often an expansion of fishing area (WBS Senegal 1970, WBP Panama 1971). However, placing limits on the level of fishing effort was soon considered, through limiting the capacity of fishers to bring fish to market (e.g., WBP Panama 1971), or by limiting the amount of time they spent on the waters (e.g., WBA Panama 1983).

A similar story unfolds with global tuna stock.[7] Several World Bank reports catalog the

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[7]The documents that include tuna stock development are WBP China 1963, WBP China 1967, WBS Korea 1966, WBP Ghana 1969, WBA Maldives 1979, WBP Maldives 1983, WBP Maldives 1992, WBS Mauritania 1971, WBS Mauritania 1994, WBP Panama 1971, WBP Panama 1975, WBA Panama 1983, WBS Senegal 1970 and WBP Somalia 1984.

References to management

References to access and restriction

References to over-fishing and depletion
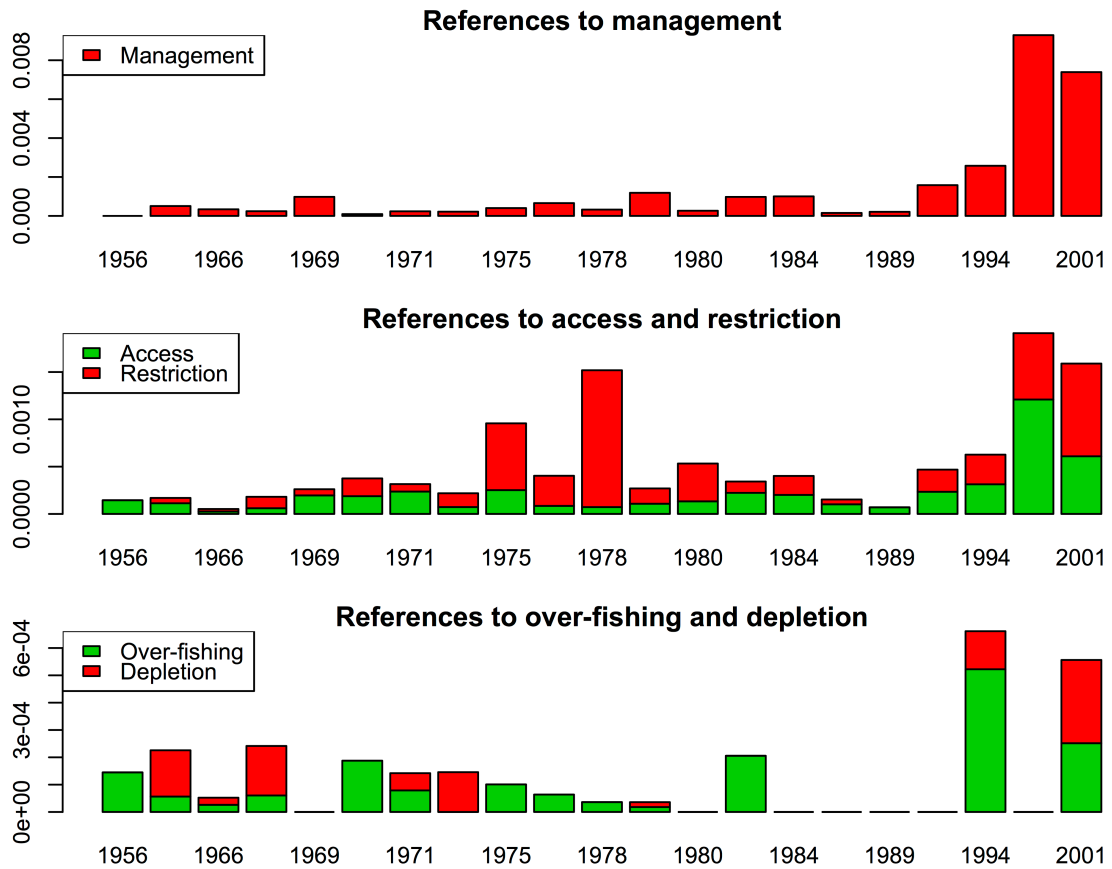
Figure 2.3: Word frequencies by year, relative to total document word counts for all World Bank fishing documents through 2001. **Top:** Any mention of fishery management in project documents was brief until 1992. **Middle:** Discussion of access rights started growing in 1967, peaked around 1978, and diminished until strong rise in 1992. This rise and fall suggests an active debate around commons issues, preceding the synthesis in the 1990s, in line with the narrative in section 3. **Bottom:** Based on word occurrences, there seems to have been a decreasing interest in over-fishing from 1956 to 1980; followed by sporadic panic, suggesting that the problem was ignored through the 1980s, but not resolved.

growth of an international open-access problem, but with little awareness of it at the time. Evidence of over-fishing is consistently ignored: "The fear that declining catch rates during 1962 to mid-1965 indicated depletion of the resource was allayed by the sharp rise in catch rates from mid-1965 to 1966" (WBP China 1967). The rise was temporary. Through the 1960s, there was no awareness of the open-access problem as such:

> It seems evident that the additional vessels introduced by Taiwan have had no influence on levels of catch and there is no reason to believe that the addition of a further 20 vessels would affect the abundance of stocks distributed over such a wide area. Japan has now ceased increasing its deep-sea tuna fleet and the foreseeable participation of other nations in the Indian Ocean tuna fishery is unlikely to have any significant impact on the stocks. (WBP China 1967)

However, a decade later, some international fishing regions had restricted fishing to brief seasons to try to maintain tuna stocks (WBP Panama 1975). Of the regions that mention tuna in their reports, tuna stocks in China, Ghana, South Korea, Mauritania, and Panama have since collapsed (according to the catch-based criteria used by Worm *et al.* (2006)). Catch per unit effort shows temporary increases, followed by collapse for China, as well as a general downward trend globally (see figure 2.4).

Evidence of project attempts to preempt over-fishing start in the late 1970s. The World Bank began to encourage the monitoring of fish populations under intense exploitation (WBP India 1977, WBP India 1978, WBA Philippines 1979). Simultaneously, however, some reports in the late 1970s offer rationales for why over-fishing should not be a concern (WBA Maldives 1979, WBA Philippines 1979). By 1992, monitoring was considered a necessary element of fisheries management; for example, one report advised that "careful monitoring of landings [i.e., catches] is desirable even though tuna stocks do not appear to be under pressure." (WBP Maldives 1992).

Two external milestones make a useful comparison. The northeast Atlantic cod fisheries, one of the most important European fisheries, peaked around 1968, followed by lower catches and a total collapse in 1992. World catches across all wild fish leveled off between 1968 and 1980. Starting in 1981, world catch rose again until 1986, and has since remained level. This helps explain concern for over-fishing in the 1970s, which lessened in the 1980s.
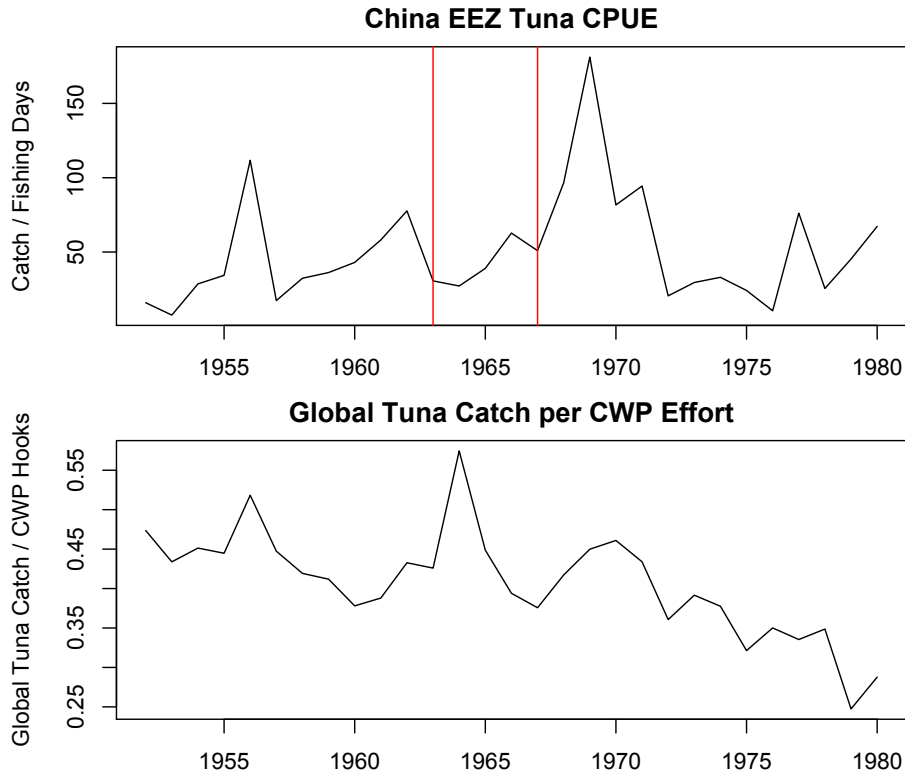
Figure 2.4: **Top:** Catch per unit effort (CPUE) for tuna within the exclusive economic zone of China between 1952 and 1980. Catch data is from Sea Around Us (2007) and effort data is from the Western & Central Pacific Fisheries Commission (WCPFC), translating all effort metrics into a single "effective fishing days" estimate. The red lines denote the years of the two China fishery projects, WBP China 1963 and WBP China 1967. While catch rates increased temporarily after each project, a collapse occurred within a few years. **Bottom:** Extrapolated CPUE for tuna globally, using WCPFC hook data as a proxy for global effort. These estimates do not include the 2.42% additional yearly increase in effective effort suggested by Watson *et al.* (2012), which would exacerbate the downward trend.

Concerns around access restrictions also shifted. Between the 1960s and mid-1970s the term "access" is almost never used in World Bank reports with respect to a management regime. The sole exception is a positive observation in WBP China 1967 that the global tuna stock "constitute a truly international resource accessible to all fishing countries." As the World Bank increased its focus on supporting private enterprise in agriculture during the late 1970s and 1980s, project reports expressed positive interest in access for fisheries, focusing on increasing the benefits to fishers of market access, through the building of roads and other infrastructure. The potential for unsustainable usage is absent from discussions of providing this kind of access.

Quotas were finally proposed as a management strategy in WBA Panama 1983, in fairly exploratory terms. Modern quotas are licenses which allow a ship to catch a certain tonnage of fish over the course of a season. In the last decade, ITQs (individual transferable quotas) have grown to be considered a fundamental component of management in many developed regions (Heal and Schlenker, 2008). In the World Bank corpus, quotas were first mention in WBP China 1963, expressed in the sense of a communist-style production quota. They were not mentioned in the 1980 report that tried to bring the problem of commons management to light. However, by WBR 1992, transferable catch quotas were considered a natural solution to the commons tragedy. In 1994, a sector report in Mauritania provides an extensive discussion of the implementation difficulties and complexities of quotas, which is further expanded in a 2001 Argentina report.

2.4 Equity in the World Bank commons

WBR 1980 describes the years leading up to the Berg Report, at a time when studies of the commons had identified a number of failings in Hardin's original theory. This study is inspired by Southeast Asian observations (its author, Donald Emmerson, is an Asia-Pacific scholar), but addresses fishing world-wide. WBR 1980 is largely a critique of the applicability of the theory of open-access resources to fisheries and its solutions to the tragedy of the commons: "[the open-access argument] is not sensitive enough to the contexts in which fishing occurs, to the different patterns of human behavior and motivation that, in

different cultures, influence the fishing act and its consequences." This sentiment echoes Ostrom (2010) emphasis on socio-ecological contexts as well as the concern for local community needs and expertise prominent during McNamara's tenure. While the concerns expressed by actors in this history echo issues codified by later authors, there are significant differences. The motivating concern in WBR 1980 is the poverty of fisherman.

A brief overview of fisheries management theory helps contextualize this report. Since the 1930s, "maximum sustainable yield" (MSY) has been a primary concern of fisheries management– the idea is to ensure that fish stocks are maintained at a level of maximum production of new fish to catch. Too high of a stock would leave little room for new fish; too low of a stock would lack an optimal number of reproductive age fish.[8]

Starting in the 1950s, fishery economists began to advocate for management to optimize economic efficiency. Gordon (1954) characterized the goal of "maximum economic yield" (MEY), which seeks to optimize fisheries at a lower level of exploitation than for MSY but at a higher profit. MEY differs from MSY both due to an integrated accounting of the cost of exploitation, and due to discounting potential future benefits from the fishery in a market that provides alternative investment opportunities. Scott (1955) added to Gordon's work by arguing that sole ownership of fishery resources was the most efficient institution for achieving MEY. According to WBR 1980, this movement was interpreted differently by many developing countries. They recognized a choice between MEY and MSY: governments could choose to maximize economic benefit, using their fisheries as drivers for development, or to maximize yield and thereby the number of poor fisheries who could be supported by a region's fishery. Many socialist countries made an explicit policy choice for the latter.

During the 1970s, the debate around fisheries shifted to focus on whose ends should be served by the restrictions involved in fishery management: whose property should the oceans become (Royce, 1988)? Economists in the developed world claimed that common property regimes, by at best supporting MSY, led to greater poverty. Sole ownership, on

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[8]In general, open-access yield is much lower than MSY. The level of fishing effort which produces MSY is optimal under zero discounting of future profits, while effort in an open-access fishery corresponds to an infinite level of discounting: there is no incentive in an open-access commons to preserve current stock for future yields (Conrad and Clark, 1987, chap. 2). Fishers will increase their exploitation and additional fishers will join the fishery until the marginal fisher earns zero profit.

the other hand, created the incentives for MEY and led to accelerated growth. WBR 1980 argues that this narrative ignores issues of wealth distribution. Economics saw the potential economic benefits as "dissipated" in a common property regime, while other groups saw these benefits as "shared" (WBR 1980). Developed countries argued for conservation and exclusion, while developing countries argued for distribution. These policies often integrated well with preexisting cultural norms to share benefits, in opposition to the "rational agent" assumptions typically made by in the economic analysis of open-access resources (e.g. Gordon, 1954).

Simultaneously, there was a growing recognition that the core of the fishery problem was industrial fishing. Clark (1973) notes "The most spectacular and threatening developments of today... can by no means be attributed to impoverished local fishermen. On the contrary it is the large, high-powered ships and the factory fleets of the wealthiest nations that are now the real danger."[9] Royce (1988) argued that open access should be preserved for non-commercial and subsistence use, and identified commercial exploitation as the cause of management problems. For these commercial entities, however, the management blueprint follows economic theory: "the rights should be owned, be transferable, and be divisable, so that sale or purchase of them would let a fishing business become efficient."

A decade after WBR 1980, WBR 1992 presents a different economic understanding of commons issues. In this report, the scope of policies for addressing commons problems consists of the range from "governmental restrictions on use of the resource to emergence of private collectives to allocate use rights." While government is not considered a necessary actor, access rights are. A lack of property rights is considered synonymous with a lack of restrictions.

WBR 1992 includes a short section on equity concerns that can arise from access-rights management schemes. These paragraphs read as highly skeptical of equity issues, while noting that restrictions "could change the proportion of small to large scale fishermen",

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[9]For example, this became a key issue in WBP Yemen 1980: "With unclear objectives left open to local interpretation by weak management, the corporation did initially focus heavily on its own non-project commercial catching and marketing activities which threatened to come into conflict with the interests of the small-scale fishermen." (WBA Yemen 1992)

and that "the resulting distribution of income from fishing could be highly skewed". In light of this, WBR 1992 recommends that "policy-makers who use fishery management as a development strategy may need to be sensitive to the occasional need to sacrifice economic efficiency in order to assure survival of economically disadvantaged groups, at least in the short run." The assumption is that once rights are firmly established, markets will efficiently ensure collective benefits.[10] This perspective could not be more at odds with WBR 1980.

During the 1970s, fishery management mainly aimed to either ensure high benefits to stakeholders or to ensure high yields. WBR 1992 reinterprets the aims of management as correcting a market externality. It argues that "fishery management may be a highly effective strategy to increase the contribution of the fishery sector to the net economic output of society, including the fishery sector itself." The World Bank had absorbed a keen awareness of commons problems, and concentrated on resolving them while disregarding the equity concerns that arose from its economic solutions.

The economic solution to the commons seems to have co-evolved with the drive for a free-market approach to fisheries. WBA Maldives/Yemen 1994 reviews a series of fishery projects in the Maldives and Yemen, aimed at supporting small-scale fishermen. In Yemen, the fisheries were explicitly opened to the free-market through the World Bank's intervention, replacing a state-run agency. In the Maldives, WBA Maldives/Yemen 1994 advises caution in doing the same only because "the pool of private entrepreneurs may be too small to quickly replace the State Trading Organization." These observations seem to be made without consideration that one likely consequence of increased free-market demand without governance of the commons is a tragedy of over-fishing. The report provides evidence that a state of over-fishing is being approached, but advises "investigation" and "monitoring", rather than management.

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[10]The section argues that any disenfranchised groups could be retrained or simply supported financially using of the new profits.

2.5 Panama's fishery projects

In the 1970s, equity was a primary concern, and by 1992, free-markets were the solution. Did this transition reflect a failure of previous management? The World Bank's early fishery development work in Panama offers additional insight into the years between 1971 and 1983, with three project reports and two audits. Projects in 1971 and 1975 aimed at expanding the shrimp fishery, with assurances that over-fishing was impossible but that careful management was nonetheless required. In fact, early documents mention a surprisingly proactive management style of the shrimp fishery, with existing government institutions and interventions aimed at ensuring MSY. In response to decreasing catch per vessel, the government placed a cap on allowable ships in 1968. By the time of the first fishery project, in 1971, the shrimp fishery was already believed to be fully exploited; the project aimed to replace old ships with new ships to stop catches from decreasing as ships aged and retired (WBP Panama 1971). At the same time, the loan included a provision to educate ship captains to increase the effectiveness of their exploitation, ignoring the increase in total exploitation that would result from this effort.[11]

Old ships did not retire as expected, but the project ships were nonetheless added to the fleet, depressing the catch per vessel. By 1977, catch had decreased so far that the government prohibited any fishing for two breeding months each year.

An audit of the first fishery project, WBA Panama 1979, critically argues that even if the old ships retired, the technologically-improved, newer ships would ensure an effective increase in pressure on the stock, without benefit since the fishery was already fully exploited. There were also significant problems with the enforcement of new standards, for example, to limit ship horsepower. Another audit, WBA Panama 1983, has long comments about the pervasive problem of over-fishing, which it claimed has plagued the fishing projects from their beginning but was given insufficient attention. By 1983, a fishery collapse had not yet occurred, but was expected.

The Panama government, however, may have held a different perspective. WBA Panama 1979 also includes a letter from National Bank of Panama (BNP), which adamantly denies

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[11]WBP Panama 1975 has ecology section, but it only concerns fish odors.

that the World Bank project contributed to over-fishing. The letter notes that the limit on ships was only increased slightly to accommodate project ships (the project contributed 26 ships, one ship retired, and before the project there were 23 fewer ships than government-imposed limit). There are two issues here. First, the BNP considered the natural limit to be the one placed by the government, rather than one based on catch per vessel. This is partly a question of whether fishing effort should be based on economic concerns or social decisions. Second, for the BNP, an increase in vessels without a corresponding increase in catch did not imply a failure– perhaps because they recognized the direct benefits to captains of the new ships. The World Bank specifically avoided employment programs organized through local institutions (Kapur *et al.*, 1997), but WBA Panama 1983 notes pressure from the BNP to make funding under the second project available to small-operation captains.

This short account suggests that in 1971 and 1975, the World Bank supported the management approaches taken by institutions in Panama, which aimed at equitably supporting the fishing community. Investment in new ships and education was intended to ease the burden on poor fishers, even if it could not raise their profits. By 1979, the World Bank had shifted its perspective, and in 1983 measured the success of all past fisheries projects by the single metric of economic returns.

2.6 Conclusion

The history told by Ostrom and others of the development of the concept of the commons appears to be the story that the current antithesis tells about itself. However, there was a previous antithesis, with a different story. That group, which was active within the World Bank during 1970s, focused on equity concerns, and the problems involved in the transfer of ownership entailed by the privatization of the commons.

Before 1992, the economic solution to commons tragedies was unclear, but not for a lack of management expertise. Common property theory was well understood, as well as the disruptive role of industrial fishers within traditional commons. Rather than identifying effective traditional practices, the early commons movement focused on the effects of economic intrusion. Traditional management formed a set of de facto owners, and the

transition to a property-oriented approach to development and management often meant disenfranchising these fishers.

Effective commons management hinges on the nature of over-fishing. If over-fishing is defined in comparison to an optimal profit or to the behavior of a sole proprietor, then many effective traditional management schemes fail. However, for many developing countries, the potential for sharing wealth was considered more important. Governments actively engaged with fishing communities in many regions, encouraging fair distribution and protecting groups from industrial fishing pressures, at the expense of some potential economic growth.

The problem of over-fishing appeared prominently with the peaking of the European cod fishery, in the same year that Hardin published *The Tragedy of the Commons*. During the 1970s years, the World Bank supported governments playing a role in management, and maintained a nuanced position on the utility of property rights and free-markets in fisheries. Issues of access and usage restrictions were discussed in fishery reports, but the emphasis was on supporting communities. With the Berg Report in 1981, privatization became the norm. Free-market pressures increased and over-fishing occurred, but these problems remained masked until the early 1990s as modern ships harvested ever wider regions.

During the 1980s and 1990s, as traditional management schemes collapsed under the new market demands, fishery economists developed new methods of management. A greater role for property rights– through quotas, for example– was proposed as a solution to over-fishing, though markets were also part of the problem. It would be another decade before management of the market and more context-specific approaches were recognized as necessary elements of fisheries policy, in light of an ever greater number of fishery collapses around the world.

Chapter 3

Marine Self-Organized Criticality
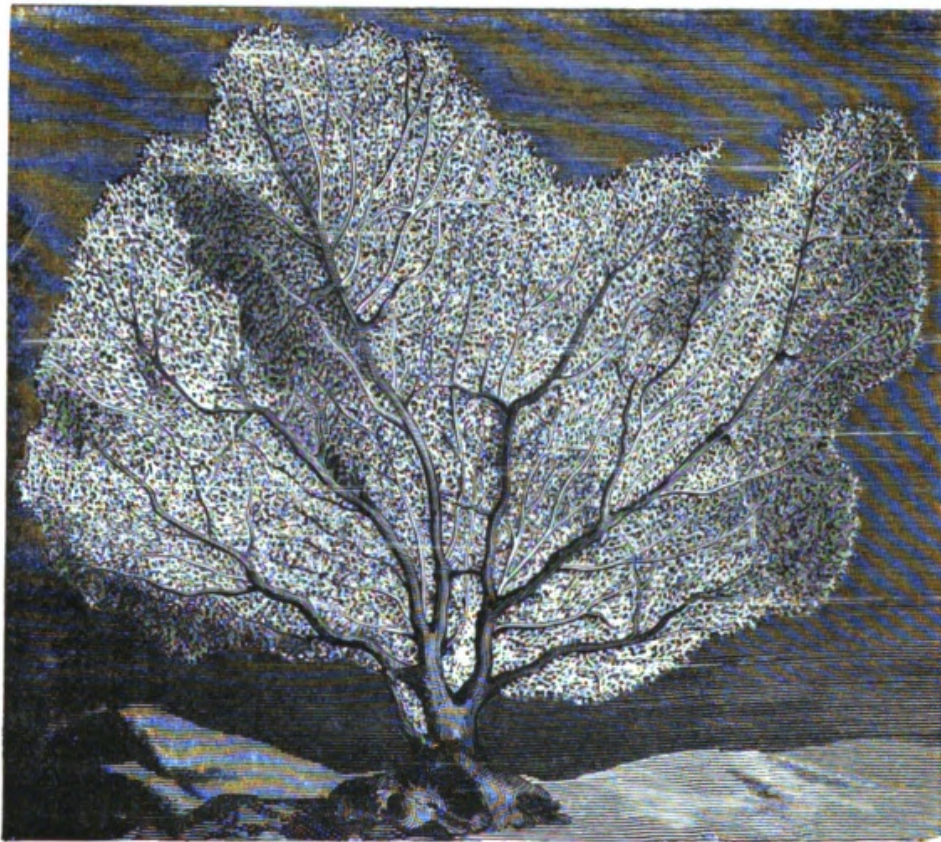
THE SEA-FAN.

Figure 3.1: Reproduced from Kirby (1884, p. 68).

AND JEFFREY SACHS.

Fine-scale, local interconnections are often the source of the larger spatial structures and dynamics that affect commons management. The class of models explored below in the context of marine ecology provide a mechanism for making that connection. While the implications of this model are more limited in their applicability than the model presented in the next chapter, its insights inform the model-building process.

Cross-scale emergence is a common feature in marine ecosystems. Emergence in an ecosystem describes the process by which "properties of the ecosystem at large spatial scales result from feedback interactions between components occurring at smaller scales" (van de Koppel *et al.*, 2005). Classic examples include fish schooling (Hemelrijk and Hildenbrandt, 2012), synchronized spawning in coral reefs (Harrison *et al.*, 1984), and the collapse of open-access fisheries. In each of these cases, micro-behaviors of individual agents produce dynamics that are usefully described at a macro-scale.

Emergence is a very general process, which can be produced by an unlimited range of mechanisms. This chapter considers one class of those mechanisms, called self-organized criticality. Self-organized criticality itself encompasses a broad range of behaviors and can be explored through qualitatively different models. However, it provides a level of specificity needed to identify useful insights into several forms of emergence in marine ecosystems.

3.1 Self-organized criticality

In the past two decades, the concept of self-organized criticality (SOC) has generated considerable interest, in fields ranging from physics to finance. The implications of self-organization and critical states have the potential to revolutionize our models of many natural and social systems and reshape our intuition. This paper reviews the fairly limited treatment that SOC has been given in marine ecology within the deeper literatures on complex systems and fractals in marine environments. It also identifies some under-explored potential that SOC has for informing research on conservation in marine ecosystems.

Self-organized criticality is a kind of emergent behavior found in a wide range of complex, spatial, and historical systems. These systems exhibit a critical state between chaos and

order, within which small perturbations can escalate to any size. Bak (1990) argues that

> the large temporal fluctuations, and the spatial self-similarity [exhibited by many
> systems] are two sides of the same coin: "self-organized criticality". The idea
> is that the systems operate persistently way out of equilibrium at or near a
> threshold of instability. The systems evolve automatically to this critical state
> without any fine-tuning of external fields; hence the criticality is self-organized.

Although the underlying mechanisms and structures that cause SOC are still unclear, most SOC systems exhibit a collection of interrelated characteristics. First, like all natural systems, these systems rely on structures in space or networks of connections, so their dynamics cannot be fully described with analytical expressions. Furthermore, the full state of the system is able to "build up" in time, historically and heterogeneously (Pascual and Guichard, 2005). As it builds, driven by some force, the system approaches a critical limit, beyond which it becomes locally and globally unstable. Instability leads to local collapses, which can avalanche by further destabilizing nearby regions. The distribution of the sizes of avalanches follows a power law, implying that there is no normal size of avalanches. As a result of this ongoing build-and-collapse dynamic, the SOC system maintains an emergent critical state. If the system is spatial, the critical state is characterized by fractal-shaped patches of order and disorder (Bak, 1990). Thus, the system exhibits heterogeneous, scale-independent features both in time and space.

The classic sand pile example, developed by Bak (1990), remains among the most intuitive, and a brief overview is informative.[1] In this computational model, grains of sand fall on a plane, forming a pile. Sometimes, a grain of sand falls on an unstable area of the pile's slope, causing an avalanche. A graph of the number of avalanches versus their size, measured in sand grains, conforms to a power law: for every doubling of avalanche size, the number of avalanches decreases by a consistent factor (see figure 3.2, a). This is a self-similar relationship, suggesting that the avalanches have no natural scale and very little predictability. The continuous build-up of instability and avalanching process naturally organizes the pile into a critical state, where the next sand grain could produce an avalanche

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[1]For other accessible examples, see Bak (1996) and Buchanan (2001).

System	Medium	Driving Frc.	Critical Var.	Collapse	Scale-Independence	Reference
Classic Examples						
sand piles	plane	sand drops	pile slope	avalanche	avalanche size	Bak (1990)
earthquakes	plane	tectonics	rock tension	earthquake	earthquake magnitude	Olami *et al.* (1992)
war	globe			conflict	casualties	Buchanan (2001)
academic papers	network				references	Buchanan (2001)
markets	line	competition		failures	fluctuation size	Stauffer and Sornette (1999)
Ecological Studies						
forest fires	plane	tree growth	density	forest fire	area burned	Malamud *et al.* (1998)
ecosystem variation	space	competition			variations by size	Jørgensen *et al.* (1998)
evolution	ecosystem	evolution		extinctions		Leroi (2000)
river braiding	plane	hydrology			spatial patterns	Sapozhnikov and Foufoula-Georgiou (1999)
Marine Ecosystems						
marine fossil record	ecosystem	evolution		extinctions		see below
brittlestar beds	time	predation			predator-prey dynamic	Aronson (1992)
microphytobenthos	space	competition			patch shapes	Seuront and Spilmont (2002)
colonial organisms	networks		branch ratio		branch pattern	Sánchez *et al.* (2004)
salt marshes	plane	wave action		erosion		van de Koppel *et al.* (2005)
bioluminescence	space	turbulence		flashing		Noever and Cronise (1994)
seagrasses	plane					van der Heide *et al.* (2010)

Table 3.1: Some classic, ecological, and marine SOC studies, and the characteristics that they have identified. SOC systems are identified by a spatial and historical medium, a driving force which causes the system to self-organize, a critical variable for which it is balanced between order and chaos, and a kind of collapse which can build into an avalanche. As a result of this dynamic, a variety of scale-independent and self-similar properties emerge; some authors begin by identifying these properties and follow up with models of the process; others start with models. Table cells are left blank where authors do not discuss those elements. Also see Turcotte and Rundle (2002) for additional examples.

of any size. Furthermore, graphs of the destabilized regions from an avalanche have fractal properties, implying that there is no natural size to these spatial structures (see figure 3.2, b).
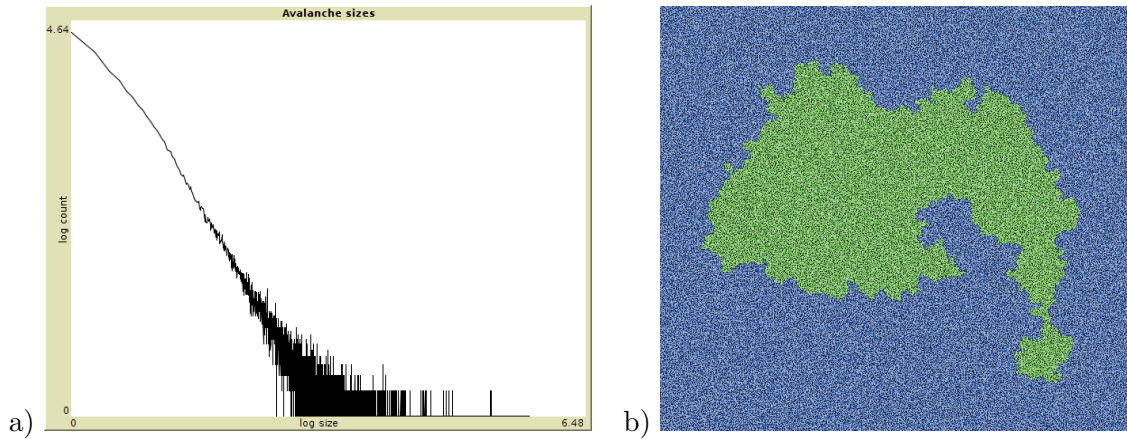
Figure 3.2: Example power law relationship and fractal clusters, using the NetLogo Sandpile model (Weintrop *et al.*, 2011, Wilensky, 1999). The power law in (a) applies to the sizes of avalanche from a single grain drop. An example region affected by an avalanche is shown in (b). The size and boundary of the avalanche region in (b) is not readily apparent from the distribution of unstable regions, shown in the lightest shade of blue.

Models of self-organized criticality have been proposed for earthquakes, forest fires, market fluctuations, human conflicts, and many other systems (see table 3.1). Many researchers studying the intersection of ecology and SOC appear to focus on identifying SOC features and extrapolating the implications of self-organizing forces. In particular, the power law is often taken to be the arbiter of SOC: where it is absent, SOC is shown to not apply, and where it acts, its characterization is often taken to imply a full SOC dynamic. If SOC is common, it has important implications for the role of history and local events, and the nature and predictability of catastrophic changes. As a tool for researchers, its main advantages include its departure from conventional (analytic and continuous) models, and the general applicability of its results.

Ecosystems are perfect candidates for SOC, because of their spatial complexity and nonlinearity, and their intense evolutionary forces and competitive limits. Jørgensen *et al.* (1998) suggests that "ecosystems strive towards moving as much as possible from thermodynamic equilibrium," placing them in the out-of-equilibrium domain of self-organized

criticality. Studies have identified SOC features in insect populations (Lockwood and Lockwood, 1997), pelican populations (Milne and others, 1997), forest canopies (Kizaki and Katori, 1999), river systems (Sapozhnikov and Foufoula-Georgiou, 1999), and many other areas.

Despite this work, there appear to be plenty of opportunities to integrate the insights from self-organized criticality more deeply into ecological studies. Theoretical work has focused on simplified ecosystem SOC modeling, while many empirical studies seem satisfied to identify power laws. In particular, the implications of scale-independence in ecosystems; the community-wide relationships suggested by SOC; and the use of SOC metrics to gauge ecosystem health warrant additional research, as described below.

3.2 Extinction in the marine fossil record

Self-organized criticality has long been suggested as an underlying cause of extinction in the fossil record. Evolution exhibits the "punctuated equilibria" characteristic of self-organized systems (Gould and Eldredge, 1977). This property of punctuated equilibria may also be scale-independent, applying both to ecological (micro-evolution) and geological (macro-evolutionary) timescales (Carson, 1975).

Bak and Sneppen (1993) proposed a simple self-organizing model of coevolution, in which incremental evolutionary mutations organize the biosphere into a critical state of trophic relationships, which then "collapse" into a cascade of extinctions and macro-evolutionary changes. In particular, the size of extinction events is predicted to follow a power law relationship, which suggests that species evolve into such a tightly knit complex for which one extinction can start a chain of extinctions that can grow to any size. As an implication, very large extinction events may result from this same dynamic, rather than an external catastrophe or qualitatively different process, if extinction events follow a power law distribution.

This work set off a long empirical debate. Solé *et al.* (1997) found a strong power law in family extinction records. However, other authors have suggested that this is an artifact of the statistical methods used (Kirchner and Weil, 1998, Alroy, 2008). Further

analysis showed that two distinct power law relationships may better explain the fossil record (Newman and Eble, 1999). Eble (1999) found no support for SOC in the timing of species origination for marine species, but did find evidence for SOC among land species.

Plotnick and Sepkoski Jr (2001) have attempted to move beyond this debate by applying some elements of SOC while discarding the rest. They propose a generalization of SOC properties called multifractal self-similarity:[2]

> We also believe that the underlying assumptions of SOC are incompatible with our understanding of the processes controlling macro-evolutionary patterns. Statistical analyses of the data sets are compatible, however, with the presence of multifractal self-similarity in both records, consistent with a hierarchical and multiplicative generating process.

It appears that some kind of deeper emergent structure underlies extinction– that it is not a simple matter of exogenous catastrophic events– but the nature of this structure remains unclear.

3.3 Scale-independence

Scale, applied to ecology, refers to both the spatiotemporal extent and resolution used in an analysis (Wiens, 1989). Unlike some contexts and models studied for self-organized criticality, marine environments have strong scale-dependence. Aronson (1992) emphasizes that "variables such as abundance and diversity often behave unpredictably at one level of resolution but produce predictable patterns at another". For example, fish species distribution in the Great Barrier Reef is fairly random at the scale of a single patch or atoll reef, but show predictable patterns at the scale of reef systems. Phytoplankton distribution shows a complicated set of patterns, dominated by local turbulence at the scale of kilometers, ecosystem effects on a wider scale, and oceanic flows on a still wider scale (Wiens, 1989).

Although some authors have argued for scale-dependence in opposition to the simplicity of SOC, ecological scales are neither surprising nor in opposition to the existence of

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[2]Multifractals are distributions of a variable that show some spatiotemporal self-similarity. They have also been proposed for spatial patterns of species abundance (Harte *et al.*, 1999)

scale-independent properties. Organisms work in very scale-dependent ways, because they have a natural scale, and have evolved to exploit dynamics at particular scales. On a fine enough scale, they can be considered a substrate for other organisms; at another scale, all but the autotrophs act as predators; and on a yet wider scale, most organisms are prey. This cycle repeats itself in a self-similar way at all scales. Azam *et al.* (1983) argues that the marine food web spans seven orders of magnitude, from cyanobacteria and other bacteria (the smallest of which are .2 μm) feeding nanoplanktonic flagellates, which feed microzoo-plankton (such as ciliates), which feed a hierarchy of zooplankton topped by the ocean's macrofauna.[3]

The biosphere as a whole can be scale-independent precisely because of this succession of scale-dependences. At any scale, species have filled all appropriate niches, often con-figuring themselves vis a vis other species into common arrangements (such as the preda-tor/prey dynamic, symbiotic relationships, and competition). The self-organized criticality of these relationships is evident in temporal studies of species (even though scale-dependence dominates spatial studies). The population dynamics of brittlestar beds show self-similar predator-prey dynamics on the scale of days, decades, and eons (Aronson, 1992). Tempo-ral self-similarities have also been identified in the effects of predators on gastropods and marine algae (refs in Aronson, 1992).

Spatial self-similarity is also present in a wide range of marine ecosystems. Studying the branching patterns in gorgonian corals, Sánchez *et al.* (2004) finds the fingerprints of a self-organized, self-similar critical state, which helps maintain the colony shape. Similar fractal properties are present in spatial patterns of the diverse and competitive ecosystems in intertidal sediment (Seuront and Spilmont, 2002). Fractals have been identified in a wide range of elements in the marine world: in the structures of coral reefs, marine snow, mussel beds, intertidal benthic communities, invertebrate and vertebrate behavior, species diversity, zooplankton, and phytoplankton (Seuront, 2009).

Simultaneously, the scale-independent process of evolution seems to abhor simplicity: every level of the biosphere is (fractally) filled with niches for other species. This is evident

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[3]The much larger phytoplankton compete with bacteria and support the conventional zooplankton trophic pyramid directly.

in the "patchiness" of marine ecosystems: the distribution of species in the marine biosphere is spatially heterogeneous at all scales (Downing, 1991). This patchiness is both a result of the critical self-organization of marine ecosystems, and beneficial to those ecosystems by providing living environments for species on all trophic levels (Seuront, 2009, pp. 31-32).

Power laws and scale-independence can also be usefully described as applying locally. Just as there are limits on all SOC systems (for example, sand pile avalanches cannot exceed the size of the sand pile), scale-invariant properties can be confined to a collection of scales without losing their validity.

3.4   Relationships in marine ecology

One contribution self-organized criticality models have that is missing from the ecological literature is the new relationships that it formally defines. Ecologists model ecosystems largely as collections of species and nutrients, and the relationships between them. Direct relationships between species generally fall into the classes of symbiosis, competition, and predation. Where nutrients are involved, species are commonly identified as producers, decomposers, or consumers. Most of these roles and relationships are trophic; less frequently, relationships describe how species support or undermine other life functions.

These relationships reflect an "organism bias" that remains prevalent in models of ecosystem interaction in marine ecology. In essence, this view is that a species plays the roles of its individual organisms (see figure 3.3). This is exemplified in the trophic map or food web, and chemical nutrient flow diagrams. The relationships described in these models are those observed at the organism level. The organism bias is a reflection of the implicit organism scaling problem above: the most natural scale for studying a species is the scale of its individual organisms. However, this need not be the only scale on which a species acts, as shown by the emergent effects of local changes in critical systems.

SOC in particular provides a suite of new relationships, based on species-wide and community-wide effects. These include:

Self-Organization Support when one species provides material or behavioral support for another's self-organization. For example, coralline algae supports coral recruitment
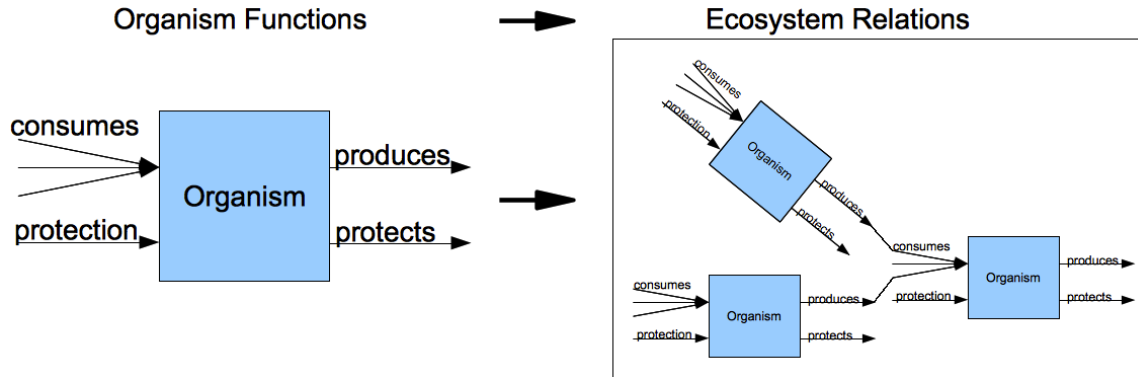
Figure 3.3: Diagram of the organism bias. Organisms have a collection of functions, which can thought of as instantaneous relationships between two organisms when they interact. The organism bias prescribes enduring relationships between two species based on the individual relationships found between their representative organisms.

(Harrington *et al.*, 2004), which is necessary for coral to self-organize into fractal forms.

Critical Value Support when the actions of one species affect the magnitude of the critical value about which another species is organizing. Salt-marsh vegetation organizes into a critical state which improves plant growth, but if the relevant critical value is too low, vegetation collapse leads to pervasive erosion (van de Koppel *et al.*, 2005).

Collapse Facilitation when one species plays a role in the collapse dynamic of another species. The effect of predation by fish and crabs on brittlestar beds supports its natural population collapse dynamic, which help its ecosystem maintain its critical population states (Aronson, 1992).

Critical Competition when the competition between species produces critical state spatiotemporal patterns. One result of competition between diatoms of several species is the creation of self-organized, fractal patch patterns in intertidal soils (Seuront and Spilmont, 2002).

In general, these are not relationships between individual organisms, and cannot be observed at the individual level; they appear only between species, with respect to their environments. With the exception of colonial animals, it is not the organisms themselves

that are self-organized, but their collective community or the ecosystem in which they are contained.

3.5 Contributions to conservation

Self-organized criticality has important implications for conservation and resilience. Complexity is omnipresent in ecosystems, making them unpredictable through nonlinearity, feedback, and chaos (Scheffer *et al.*, 2001). Our capacity to predict the effects of human interference on marine ecosystems underlies much of the work of marine conservation ecology. Conservation is also greatly concerned with resilience, because it both is a natural consequence of health and biodiversity, and counteracts the conditions that result in catastrophic shifts. However, SOC simultaneously undermines predictive capacity and supports resilience. If SOC is prevalent in marine ecosystems, it may suggest the need to reconsider complex uncertainty as a friend to conservation.

Healthy ecosystems are characterized by a high level of self-organization, suggesting that SOC metrics might be useful in measuring the overall health of an ecosystem. van der Heide *et al.* (2010) argue that the self-organized, spatial patterns of intertidal seagrasses are reliable indicators of the multiple stressors acting on them. Kolasa (2006) investigates the use of SOC power spectrum exponents as measures of self-organization and ecological integration. Lower exponents correspond to both ecosystem stability and species richness. Metrics of self-organization are significant because they reflect both ecosystem-wide dynamics, and signatures of local processes (Solé *et al.*, 1999).

Further buttressing this association, decreases in ecosystem complexity often correspond to decreases in environmental health. The overgrowth of macroalgae, a species that undermines the ability of coral reefs to self-organize, is a robust indicator of overall reef health (Bahartan *et al.*, 2010). Lohrer *et al.* (2004) find that a decrease in the complexity of marine fractal structures through thin sediment deposits causes a decrease in ecosystem biomass and diversity. The dangerous effects of invasive species can also be understood in this light: these species have not evolved to fill critical state niches, so their propagation decreases ecosystem complexity (Bax *et al.*, 2003).

Beyond reflecting ecosystem health, changes in the frequency relationships that characterize SOC systems can directly interfere with their functioning. Lohrer *et al.* (2004) notes,

> Most likely, species evolve to cope with minor disturbances that occur relatively often, particularly when those disturbances occur at frequencies greater than the generation time of the species, as this would be adaptive. However, with increasing disturbance intensity, the response of the community may shift from negligible to catastrophic, with an intermediate domain consisting of sub-lethal effects and/or mortalities of selected individuals and taxa.

A model of SOC behavior in ecosystems is developed in appendix C, as almost a direct discretization of a simple analytical fisheries model. It can produce two stable regimes, characterized by either high or low stock, or a chaotic region between the two, depending on the parameter choices. The chaotic regime is characterized by "boom and bust" dynamics, where the threshold for global fragility is below the point of population saturation. This model can help explain highly variable stocks among exploited species, since it is exactly this middle region where fisheries management might aim.

Self-organization is also an important factor in marine reserve design. Reserve design is greatly impacted by scaling effects of organisms (Wiens, 1989), which SOC informs. In addition, sites for marine reserves are currently selected under the assumption that physical features determine the distribution of species. However, new models suggest that large-scale and meta-population patterns are self-organized, challenging that assumption and the reserve boundaries based on it (Guichard *et al.*, 2004).

Based on these results, self-organized criticality has four major implications for marine conservation ecology. The research on SOC systems provides a framework for studying a class of emergence in marine ecosystems, and developing models that can describe multiple scales. Critical states point to the need for a new way of understanding and managing sustainability and catastrophic events, by working with and through complex uncertainty. Finally, self-organization is in no small way the very process of ecological restoration, and can be used as a metric for gauging the health of ecosystems. Each of these offers opportunities for future work.

Chapter 4

Management of the Distributed Fishery Commons
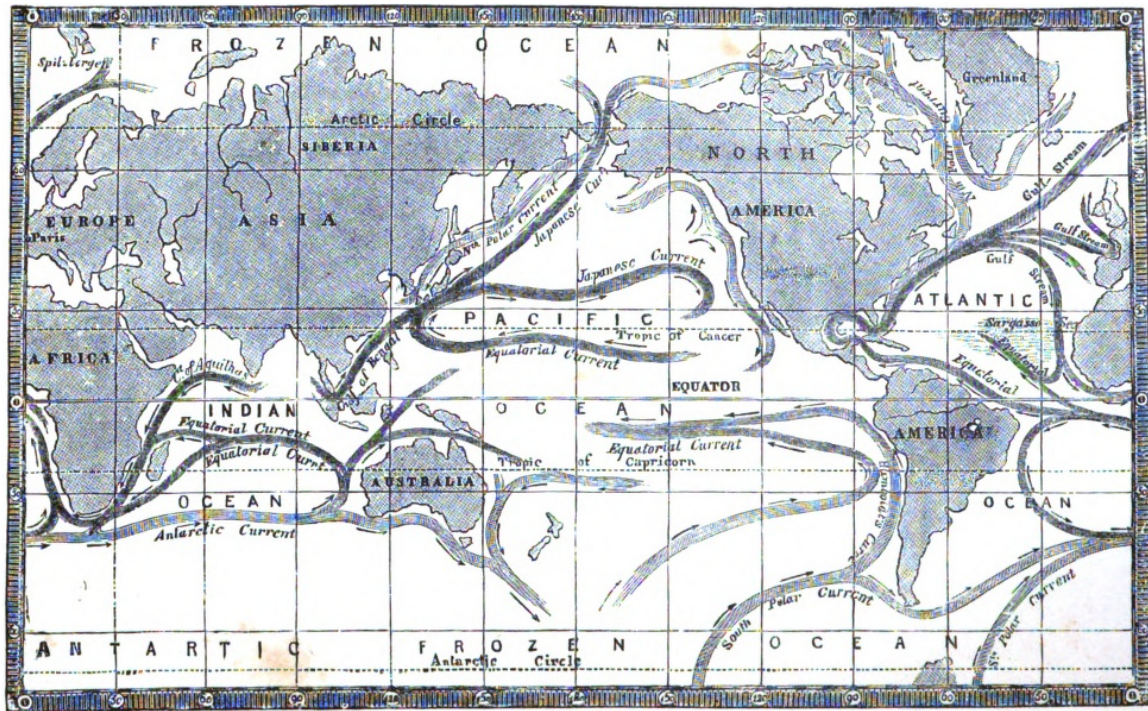
Figure 4.1: Reproduced from Kirby (1884, p. 15).

This chapter develops the central conceptual model of this dissertation. It is informed by both the broader understanding of the commons in the first chapter and of the process of emergence studied in the second. Future chapters provide an empirical foundation for aspects of the model, culminating into a computational model in the last chapter.

Fishery collapse is a global concern, affecting world food supply and economic prospects for fishing communities, and spilling across borders and across species. A growing set of policies are available to address these problems, including catch quotas, marine reserves, and community management. However, a variety of systemic problems continue to hamper their deployment and effectiveness, including those stemming from the open-access nature of some fisheries.

Wild fish stocks are common pool resources (CPRs), with high subtractability of use (rivalry) and high difficulty of excluding users (Ostrom and Ostrom, 1977). They are also plagued with many characteristics considered unfavorable to community management, often including entrenched stakeholders, mistrust of science, high variability and uncertainties in stock sizes, complex migration patterns, orientations toward short-term gains, questions of sovereignty, counterproductive subsides, incentives to misreport catches, and ambiguous spatial boundaries. Despite all these issues, some regions have had long histories of success (Pinho *et al.*, 2012). This paper considers commons that share these characteristics and asks how further success might be possible.

While some commons are "well-mixed", such as CO_2 in the atmosphere or a small pasture, in many commons the distance between users is a key parameter. For example, fish stocks, timber, groundwater, water and air quality, and other ecosystem services are exploited and impacted heterogeneously across space. Users and stakeholders also vary across the system's area. Modeling and thinking about these resources in a non-distributed and non-spatial way can significantly distort our understanding of the system and the choices within it (Durrett and Levin, 1994, Smith and Wilen, 2003).

I argue that understanding the successes and failures of fishery commons demands a cross-scale perspective. Most CPR research focuses on the local level, where cooperative management has the greatest potential. Contemporary CPR theory ignores many influencing factors from the outside world, or considers them "as 'given' or as a 'black box'",

independent of the activities of the local environment (Steins *et al.*, 2000). This perspective has limitations, particularly for fisheries where much of the crucial ecological activity takes place on a wider scale.

To support this investigation, this paper develops the concept of the "distributed commons" in the context of fisheries management. A distributed commons differs from the traditional commons model in that the boundaries of the system are imprecise and resource users are affected most by impacts located nearby. This conception brings to the fore the role of perspective and scale (including cross-scale aspects) in understanding the political economies around management. I sketch several consequences of this new model of the commons applied to fisheries management, along with the model's implications for the political economy surrounding local stakeholders, governing institutions, and their cross-scale interactions.

In the first section, the role and consequences of cross-scale issues are explored. Second, a general theory of distributed commons is developed, and certain consequences are explored. Finally, some consequences of this model are enumerated for fisheries management. In chapter 7, the intuitions from these theoretical models are developed into an agent-based model, to explore three dynamic situations.

4.1 A need for cross-scale management

Scale, applied to social-ecological systems, refers to the spaciotemporal, quantitative, and analytical extent and resolution used in a perspective or analysis (Wiens, 1989, Gibson *et al.*, 2000).[1] Scale has long been a central issue in ecology as explored in the last chapter. The dynamics, predictability, concerns, and opportunities differ depending on the scale of observation.

Traditional CPR research focuses on the local scale for several reasons. Management at

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[1]The term 'scale' is used in this paper mainly in reference to the spatial dimension, although temporal scales and institutional scales are also relevant. Importantly, discussions of "different scales" and "cross-scales" always refer to differences in the extent and resolution of a perspective or analysis within the same dimension. Thus, the distinction drawn by Cash *et al.* (2006) is not used and 'scale' and 'level' are used as synonyms in this paper.

the local scale provides the greatest potential for communication and cooperation, stakeholder engagement, application of local knowledge, and agile adaptation (Ostrom *et al.*, 1999). One branch of CPR research responds to the failures of management orchestrated by centralized institutions, which in many cases has resulted in unsustainable exploitation and undermined long-standing traditions of stewardship. A focus on the local scale also affirms the local perspective, in which the concerns of users and their relationships are paramount.

There is also an implicit claim that local analyses are applicable to larger scales and remote contexts. First, it is argued that higher level features have little bearing on the institutions and outcomes at a local level (Elster, 1989, Furubotn and Richter, 1991). Further, Keohane and Ostrom (1995) argue that important facts of commons management are scale-invariant, so conclusions from local CPR studies can inform problems of global governance.

This paper makes a case that scales do matter, and that it is a combination of cross-boundary issues and the relationships between the local and the global contexts that determine the scope of many commons issues. Cross-boundary issues arise when dynamics outside of a system affect those within it, such as when the sustainability of a region is undermined by fish harvests or pollution in neighboring regions. The emphasis on a local perspective is inappropriate for commons rife with cross-boundary issues. Schlager *et al.* (1994) argues that cross-boundary migration aggravates many CPR problems:

> (1) users are more likely to attribute flow declines to the behavior of users elsewhere in the system; (2) the users in any one location cannot control the flow even if they act collectively; (3) because no one group can control the flow and capture the benefits of collective action, users in any one location are less likely to provide benefits for users elsewhere in the system by restraining their own appropriation activities; and (4) coordinating activities with users in other locations raises transaction costs.

The complexity of stock dynamics makes it very difficult to determine the effects fishers have on each other. As a result, the rules adopted by fishers often do not address the appropriation externalities at the heart of CPR problems, because the source of these problems

extends beyond local fishing grounds. Leaving cross-boundary issues outside the scope of study undermines our ability to address the root of many CPR problems.[2]

Agents naturally act at many scales. Norberg *et al.* (2008) identifies the operational level, at which fishers make day-to-day decisions, as both smaller-scale and more pertinent than the collective choice ("local") level. Fishers regularly operate at more than one level, making operational decisions, helping debate collective choice rules, and providing input for regional rules. The multiple spatial and temporal scales upon which fishers act also affect the outcomes of management (Smith and Wilen, 2004). The cross-scale literature focuses on the relationship between fishing communities and the government or the market, but intermediate levels (such as middlemen) can also play crucial roles (Crona *et al.*, 2010). Those different levels can cause "cross-scale" issues.

Cross-scale issues are pervasive in complex social-ecological systems. Cash *et al.* (2006) identifies three common kinds of "scale challenges" in institutional responses to scales: "(1) the failure to recognize important scale and level interactions altogether, (2) the persistence of mismatches between levels and scales in human-environment systems, and (3) the failure to recognize heterogeneity in the way that scales are perceived and valued by different actors, even at the same level." In natural systems and their management, scale challenges come from both "top-down" and "bottom-up" mechanisms. Emergent patterns drive the dynamics of higher levels,[3] while intervention into regional dynamics is impossible without warping the functioning of lower levels.

The cross-scale impacts that government-scale policy has on the local scale are the best studied. Local fishing communities are affected by a wide range of outside factors, including regional environmental health, management policies, and market demands. In particular, national HDI, quota regimes, and the existence of protected areas have been identified as

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[2]Fishers are confronted with a wide variety of difficulties, including pervasive uncertainly (Schlager, 1994). I will not discuss issues, except as they pertain to scale and boundary issues. For example, a portion of fish stock variability is a consequence of fish movements, which is relevant to this exploration, while another part reflects underlying ecological dynamics, which is not.

[3]Emergence in an ecosystem describes the process by which "properties of the ecosystem at large spatial scales result from feedback interactions between components occurring at smaller scales" (van de Koppel *et al.*, 2005).

significant influences on the success of local management (Gutiérrez *et al.*, 2011). Higher level institutions can strengthen local ones, through state legitimization, enabling legislation, cultural revitalization, capacity building, and institution building (Berkes, 2002). However, centralization and higher level management also frequently undermine local institutions. Co-management, defined as an institutional arrangement between government and communities, addresses this need for collaborative rules at different scales, and will be discussed in the last section.

4.2   The distributed commons

A distributed commons is a kind of non-excludable, rivalrous resource, with localized impacts from resource users. The distinction between this and the traditional commons is most relevant when considering scale issues. At a sufficiently local scale, a distributed commons– for example, a fishing area– is simply a commons with an ambiguous boundary, cross-boundary effects, and an open community of users. As with any commons, anyone in the community can access it for individual benefits, with the potentials for aggregate externalities, on one hand, and collective or autocratic management, on another. However, classic commons theory presumes that both the scope of the resource and the community with the potential to access it are clearly and completely defined, and relatively homogeneous (Young, 1995).

These assumptions are modified for distributed commons. System boundaries are imprecise because there are no clear border lines, because the system is open with flows passing in and out, and because of uncertainty in the underlying state and dynamics. There is also no clear boundary defining a scope of the commons relevant to a given stakeholder, and different stakeholders may have very different experiences of the resources available and community using them.

Questions of storage, mobility, and property are at the heart of distributed commons, and new insights are available on these issues under this conceptual framework. Schlager *et al.* (1994) explore the distributed aspects of commons through their the attributes of mobility and storage. They categorize all commons based on the presence or absence of these
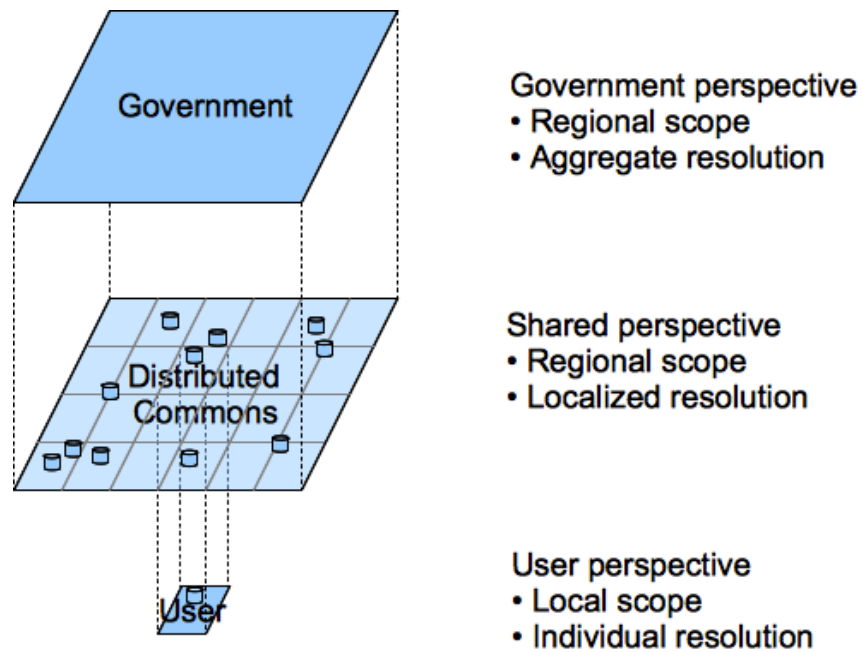
Figure 4.2: The distributed commons makes explicit the separate affects of scale and resolution. The governing institutions of a region operate across a widely distributed exploitation process, while an individual user observes other users spread out at distances from it. While the prior resolution of a government is aggregate, due to its concern for aggregate measures like taxes and law, it can cultivate a localized resolution. The user's perspective is inherently on a local scale, but by recognizing the regional scope of its problems, it can better address them.

attributes, but some distinctions they draw seem unsatisfactory. For example, fisheries are classified as having "mobile [resource] units" and "without storage", while groundwater basins have "stationary units" and "storage". Both of these contexts have varying degrees of both "mobile units" and "storage", and in very similar ways. Groundwater is far from stationary, although it seeps more slowly than fish swim. Furthermore, while water quantities are easier to measure and treat as stored, the spatial heterogeneity of geology imposes considerable uncertainty in many regions (Beckett and Webster, 1971).

For Schlager *et al.* (1994), storage must be specific to a given resource user, with their prototypical example being the well. However, even with groundwater, water left in a well can be extracted by other users over time (Pfeiffer and Lin, 2012). Conversely, although fish are highly mobile, the probability of fish varies over space, and is higher in regions where they are underexploited. Unexploited regions can become sources of population recruitment. The storage capacity of fish is isomorphic to that of groundwater: in both cases, resource units that are left unharvested diffuse and a portion of them are likely to be available later.

Property rights and "assignment problems", central questions and the direction for solutions for classic commons, become difficult to establish in full. One's capacity to lay claim to a parcel of the greater commons is easier than to ensure that activities beyond that parcel do not impact it. Because of uncertainty, "fishers are more likely to use time, location, and gear restrictions, as opposed to quotas" (Schlager, 1994). Location restrictions may be the most common rule used in community managed fisheries. Boundary rules, which attempt to restrict access to only local fishers, are a natural response to the spatiotemporal variation of distributed fisheries, but applying them to this context is difficult.

Sources and sinks are also important elements in distributed commons. Resource users can be thought of as sinks located at particular points. For renewable resources[4], production occurs across a region, but often with localized "hot spots". For example, marine reserves can act as sources of fish in neighboring fisheries, greatly increasing their sustainability (Gell and Roberts, 2003). For groundwater, recharge wells are gaining popularity as a management method, offsetting the growth of sinks by building more sources. A commons is overexploited when the sinks in a given region exceed the sources, and the degree of

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[4]Note that all CPRs with the potential for sustainability are renewable resources.

exploitation can vary across space.

The distributed commons has several conceptual advantages. It can help build an understanding of ecological issues, management issues, and the complexity between them. Strathmann *et al.* (2002) shows that many coastal species have very localized recruitment, suggesting that fish cannot be considered a single aggregate stock. This is supported by genetic and mating studies and has important implications for resilience (Strathmann *et al.*, 2002, Norberg *et al.*, 2008). The spatial distribution of degraded ecosystem regions can have important consequences for the capacity of the ecosystem as a whole to recover from shocks (Hughes *et al.*, 2005). Moreover, patterns of degradation are important determinants of whether SES systems experience regime shifts, reflecting the large-scale consequences that small-scale activities can have (Elmhirst *et al.*, 2009).

While CPR theory typically applies unitary conceptions of the system's state and the resource-using community, these abstractions are often problematic (Carlsson and Berkes, 2005). In contrast, the distributed commons model starts from a disaggregated view of the resource and its users, and thereby helps situate management options. For example, the necessary scope of management depends on the properties of the underlying resource. Management policies need to apply to larger or smaller regions based on the characteristics of the resource system that determine the degree of cross-boundary effects that users experience. Management of tuna requires international conventions since tuna are so mobile, while forest management may require only small buffer zones.

Empirical CPR studies find that heterogeneity in preferences and capabilities typically undermines cooperative governance (Johnson and Libecap, 1982). When heterogeneity in preferences is combined with a recognition of heterogeneity in membership– that not all users need to be in partnerships with all other users– new cooperative potential can be identified.

Distributed commons also make explicit the actors in the two-level games that characterize governance of global commons (Putnam 1988, Evans *et al.* 1993). Two-level games involve the relationship between domestic constituents and international negotiations. The users of traditional commons under the regimes constructed through international relations remain nationals or institutions of one state or another, these commons on an international

scale are also examples of a classic two-level game (Young, 1995). An equivalent dynamic occurs between regions in a single country, where the preferences of users of a distributed commons are necessary determinants of the national policy regimes that arise.

4.3 A model of the distributed commons

Diagram 2-a of figure 4.3 is a visual representation of a concrete model of a distributed commons. In contrast to a classic commons, where it is exactly the capacity for multiple users to simultaneously use a resource that is its defining feature and the source of its problems, the distributed commons is conceptually segmented into parcels under exploitation by at most one user or a single agent that represents a cooperative community. These parcels may fluidly shift in size and physical location, but it is not their simultaneous use that causes problems. Instead, the tragedy of the commons results from the compounded drain of many localized sinks. The compounding effects come from the cross-boundary movement of critical elements between the parcels. These elements may be the actual resource (e.g., fish and other mobile stocks), a necessary input (e.g., water for agriculture), or ecosystem services which rely on a wider domain (e.g., bee pollination services). The resulting tragedy may only apply to a region, or may be felt across the entire distributed commons.

For many real commons, this makes natural sense. The space that a boat physically occupies is inherently excludable– two boats cannot occupy the same space.[5] For ground-water use, each well occupies a distinct column, and property rights typically keep some distance between them, but too many wells across a region cause overuse.

The diagrams in figure 4.3 explore some basic classes of management strategies available for commons. The boxed figures refer to classical commons, where three potential institutional approaches are as follows.

Unmanaged commons (1-a) An unmanaged commons, with multiple users exploiting it simultaneously.

Managed commons (1-b) The exploitation of all users of the resource is limited through

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[5]Also, boats speeds cannot exceed 3×10^8 m/s.
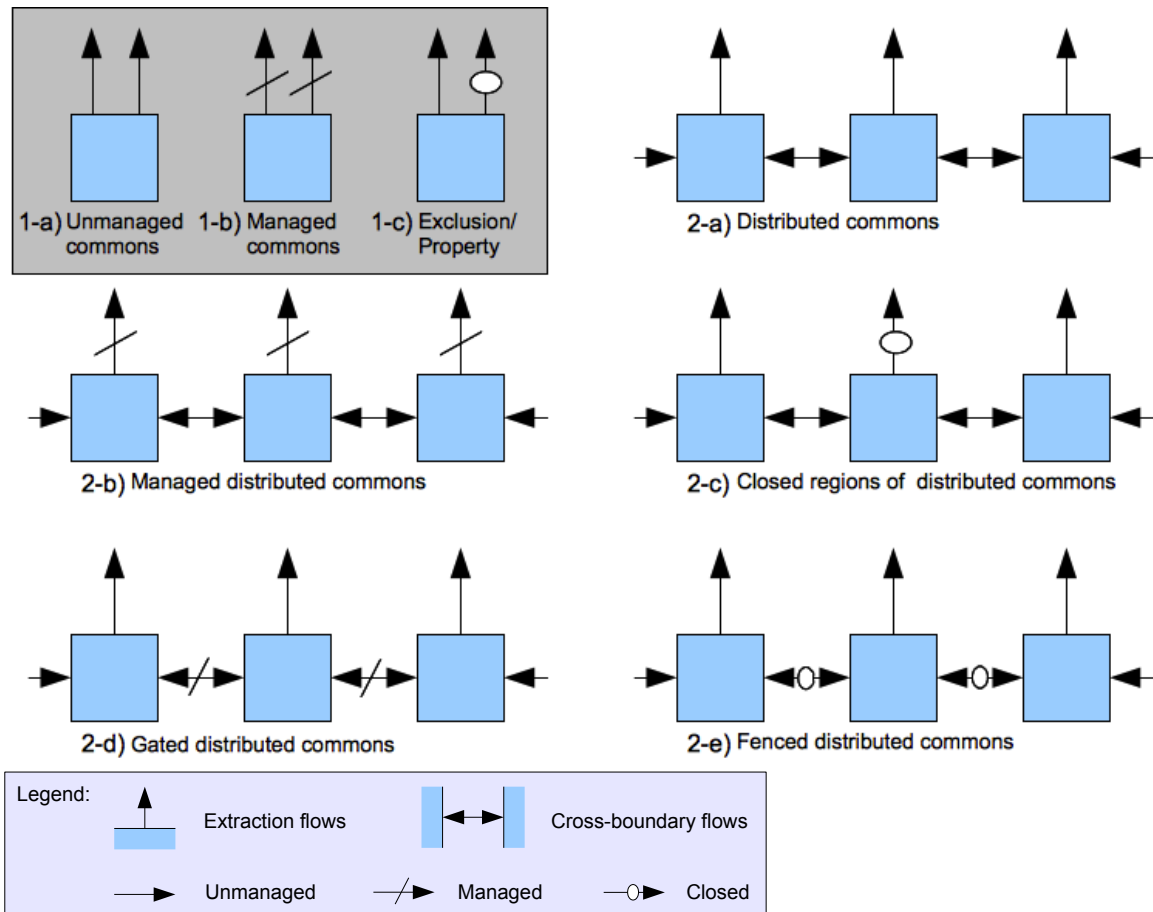
Figure 4.3: Classes of governance options for classical and distributed commons. The boxed schemas (1-a through 1-c) denote classic commons. These diagrams are meant to introduce a visual language of symbols used in the remaining diagrams, where the same approaches are applied in distributed fashion. The remaining diagrams show a general model for the distributed commons (2-a) and management options for it (2-b through 2-e). See the text for descriptions.

rules, such as quotas. The managed commons requires active monitoring and communication.

Exclusion/property (1-c) The tragedy of the commons is resolved by enforcing property rights, excluding all but a single agent or self-managing community of users.

In distributed commons (2-a), rather than imagining multiple users of one area, each area is exploited by at most user. With strong cross-boundary flows and without communication, the tragedy of the commons is almost inescapable. For distributed commons, the same abstract approaches yield more management options, due to the greater complexity of the model.

Managed distributed commons (2-b) All users within a region are managed, through limiting extraction, monitoring, and communication.

Closed regions of distributed commons (2-c) Some regions can be closed, providing sources to offset sinks elsewhere. This is equivalent to having protected areas for ecosystems or recharge wells for groundwater.

Gated distributed commons (2-d) The flow of materials between parcels is monitored and impeded, or equivalently, the material sizes of each parcel are managed. Management of the material sizes, which is mainly appropriate where the flowing material is not the same as the resource being extracted, can involve continuously changing the spatial boundaries.

Fenced distributed commons (2-e) The flow of materials between parcels can be shut off, or equivalently, their material contents can be fixed, so that each parcel or distinct regions acts like classic (non-distributed) commons with a single user.

An example of managing the material sizes of regions as a "gated distributed commons" is as follows. If each user in the commons harvests the resource to the same level, none are benefited by diffusive inflows from their neighbors, since they lose the same amount (in expectation) to their neighbors. However, by harvesting more, a user could create a gradient under which they would benefit from additional diffusion. If this amount of diffusion could

be modeled or measured, it can be exactly removed from the material size of such a user's region and added to their neighbors. This removes the incentive to over-extract in order to implicitly harvest from one's neighbors.

The diagram shows the distributed commons modeled on along a line, appropriate for a coast or a river, but a grid or network would better represent spatial regions. A network could also be used to represent more complicated interacting resource systems.

4.4 Cones of Depression

Useful resolutions of a distributed common could have a vast range, from small-scale fishing locales to large-scale fishing regions across the global commons (e.g., Berkes *et al.*, 2006). The extent of cross-boundary effects is a key factor in choosing a resolution for representing a distributed resource. Cross-boundary effects arising from a random walk of fish or the diffusion dynamics of groundwater can be modeled using the diffusion equation:

$$\frac{\partial u}{\partial t} - \alpha \nabla^2 u = g(u, x, t) - f(u, x, t)$$

for a resource distribution in space and time, $u(\vec{x}, t)$, where α is related to the rate of diffusion, $g(u, x, t)$ is the rate of resource growth (e.g., $g(u, x, t) = ru(1 - \frac{u}{K})$ for logistic growth), and $f(u, x, t)$ is the rate of resource extraction (e.g., $f(u, x, t) = -c\delta(x)$ for a point extractor). Examples of the consequences of a set of extraction regimes are shown in figure 4.4.[6]

The magnitude of impacts over space decays roughly according to a spatial exponential decay rate constant: D in $e^{-|x|/D}$, where D is related to α, the diffusion constant.[7] Point extraction produces an exponentially decreasing impact, with a length scale D, described in the hydrological literature as a "cone of depression". In other words, the mobility of resources sets the natural length scale for describing impacts across distributed resource. This result will be used in the next section to explore a spatial game on the commons.

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[6]For $g(u, x, t) = ru(1 - \frac{u}{K})$, analytical solutions have only been discovered in the special case of traveling waves (Kaliappan, 1984).

[7]This is clearest when considering the steady-state ($\frac{\partial u}{\partial t} = 0$) and $g(u, x, t) - f(u, x, t) = u(x)$. Then $u(x) = Ae^{-x/\sqrt{\alpha}}\mathbf{1}\{x > 0\}$ for the boundary condition that $u(0) = A$.

Methods used to study groundwater levels provide a simplified approach for some of the interaction scenarios that are possible. Wells have well-known compounding effects, where the cones of depression for individual wells sum in areas affected by multiple wells. We can use this approach of compounding cones to understand different exploitation scenarios and the sustainable yields that can be achieved under them. Figure 4.4 combines stock level graphs with yield graphs for the central stakeholder under four scenarios.

The most important consequences of the investigations in figure 4.4 are the changes in maximum yield, extreme yield, and potential for depletion. In diagram B, it is shown that a single user on a distributed commons can expect greater sustainable yields, as well as greater yields under exhaustive exploitation, in the presence of inward resource flows from either side. Diagram C shows the case of a local user in a region with broad depletion, which in fisheries might be produced by external industrial fishing. In this case, little additional exploitation potential is available. Diagram D shows the cone of depletion that occurs between users. As their use increases, the cross-border benefits from the depleted region between them diminishes.

The exact levels of stock and catch corresponding to the scenarios in 4.4 depend on the growth model and parameters. See appendix D for examples.

4.5 Optimal harvesting under diffusion

Optimization under spatial diffusion has recieved considerable attention recently. Typically, these discussions identify optimal control for social planners Brown and Roughgarden (1997), Costello and Polasky (2008), Brock and Xepapadeas (2008), and in some cases compare these outcomes to behavior under open-access (Conrad, 1999). As here, a key question is how to study the effect of marine protection within such a context (Conrad and Smith, 2012). The result here is most similar to Brock and Xepapadeas (2010), who compare cite-specific private optimization to social optimization under diffusion. While the spatial context used in Brock and Xepapadeas (2010) is more sophisticated, only the myopic behavior is considered. We are interested in agents optimizing under a diffusive environment with other agents.
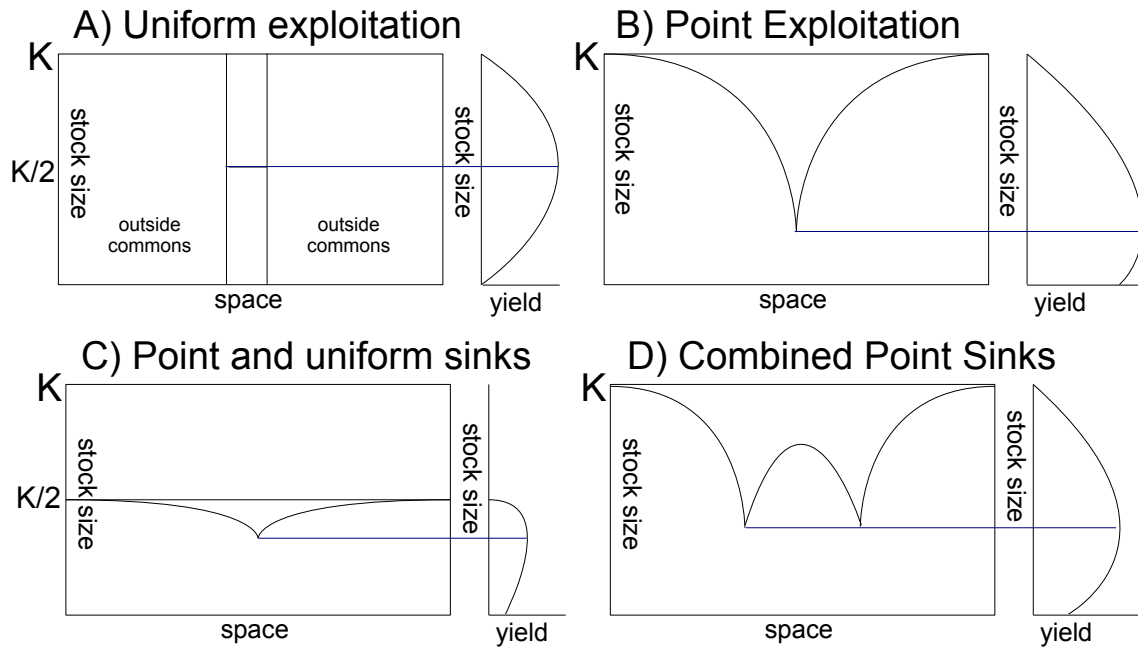
Figure 4.4: Examples of distributed commons exploitation at maximum sustainable yield, and the associated yield graphs. For each diagram, the box on the left shows resource levels across space, as they would be in a sustainable steady-state. To the right is the surplus available to a point resource user across levels of the resource, with decreasing levels down. The peak of this right graph is the maximum sustainable yield (MSY), shown left. (A) shows conventional uniform exploitation, with MSY at a stock size of half the carrying capacity (under a logistic-style growth model). Only the stock in the center are is available, and no stock flows in from outside of the commons. (B) adds the benefits of diffusion, showing exploitation at a single point sink: the total MSY is greater, since stocks are flowing in from neighboring regions; the MSY stock may be higher or lower depending on characteristics of the resource. (C) shows a point sink under conditions where the total stock is already uniformly exploited: now both the maximum yield and the stock at that yield are much lower. (D) shows the effect of multiple point sinks, with an intermediate region of depleted stocks. In the extreme, the region between the two point sinks would be devoid of resources.

Consider two neighboring regions, with diffusion rate γ between them, as shown in figure 4.5. The stock levels are X a Y, growth rates are $F(X)$ and $F(Y)$, and harvesting levels are A and B.

$$\dot{X} = F(X) - A + \gamma(Y - X)$$
$$\dot{Y} = F(Y) - B + \gamma(X - Y)$$
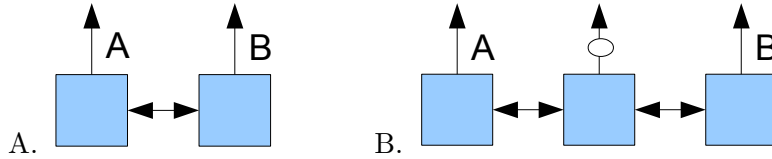
Figure 4.5: Two simple distributed commons with two agents. **(A)** Extraction occurs from two regions with diffusion between them. **(B)** The same case as (A), with enforced separation between the two regions using a marine protected area.

The optimal steady-state extraction is determined by the Hamiltonian with discount rate r:

$$H = e^{-rt}p(A + B) + \lambda_1 \left(F(X) - A + \gamma(Y - X)\right) + \lambda_2 \left(F(Y) - B + \gamma(X - Y)\right)$$

The result is that the optimal level of A and B are where $F'(X^*) = r$ and $F'(Y^*) = r$, identical to the result without diffusion.

Now consider two agents, with agent A extracting from region X and agent B extracting from region Y. We wish to find the level of harvest for these symmetric agents that represents a Nash equilibrium. Let the right agent's extraction as assumed, producing a steady-state stock level Y^*. Now the Hamiltonian for agent A is,

$$H = e^{-rt}pA + \lambda \left(F(X) - A + \gamma(Y^* - X)\right)$$

The optimal level is $F(X^*) = r + \gamma$, and by symmetry, $F(Y^*) = r + \gamma$ and this is a Nash equilibrium. This higher slope is at a lower level of stock, and a correspondingly lower sustainable harvest, than the economically optimal level, and it is driven by the perceived losses to the other agent. It also produces an range between sole ownership and open-access:

(sole ownership) $X_1 > X^* > X_\infty$ (open-access)

When $\gamma = 0$, each agent has sole ownership of their stock. As $\gamma \to \infty$, the Nash equilibrium converges to the open-access level, equivalent to infinite discounting.

Finally, if a protected area is positioned between the two, it decreases the effective diffusion between then, thereby allowing the agents to achieve higher Nash equilibria.

4.6   The distributed prisoners dilemma

The analysis above only applies to simple diffusion, but often there are additional fishery-wide effects of decreasing stocks. As an example of the potential consequences resulting from these interactions, we consider a prisoner's dilemma situation (Rapoport, 1965), with two fishery users.

The general payout matrix for any prisoner's dilemma game is

		Player 1	
		Coop	Expl
Player 2	Coop	M, M	H, T
	Expl	T, H	L, L

The game is played once with known payoffs. If both users cooperate by harvesting sustainably (Coop), they get modest returns (M). If one tries sustainable harvesting while the other excessively exploits the resource (Expl), the returns to the cooperative harvester are very low (T) while those to the exploiter are high (H). If both exploit to their full potential, they get equal low returns (L). For this model to produce a dominant strategy that results in L, L– that is, for this to represent a true prisoner's dilemma– the payouts must be such that $H > M > L > T$.

We define the fishery prisoners dilemma to have a payout for each user of the form

$$v_i = c_i - \mathbf{1}\{c_i + c_{-i} > S\}\left(\frac{c_i + c_{-i} - S}{2} + P\right)$$

where $\mathbf{1}\{\cdot\}$ is the indicator function (1 if $\cdot$ is true, else 0), c_i is the catch that user i aims for, S is the stock, and P is a penalty for causing a fishery collapse. For our analysis, let

$S = 100$, c_i be either at a cooperative level, set to 40, or an exploitive level, at 80, and let $P = 20$. The resulting payouts under this function are,

Player 1

		Coop	Expl
Player 2	Coop	40, 40	50, 10 (*)
	Expl	10, 50 (*)	30, 30 (*)

where (*) denotes a fishery collapse. This model satisfies the prisoner's dilemma criteria and rational agents will both select high levels of exploitation, causing lower total and individual payouts as well as fishery collapse. This payout function produces intuitive results for a wide range of parameter values. Figure 4.6 shows the consequences of varying levels of exploitation by one user against a constant level or equal level by the other.
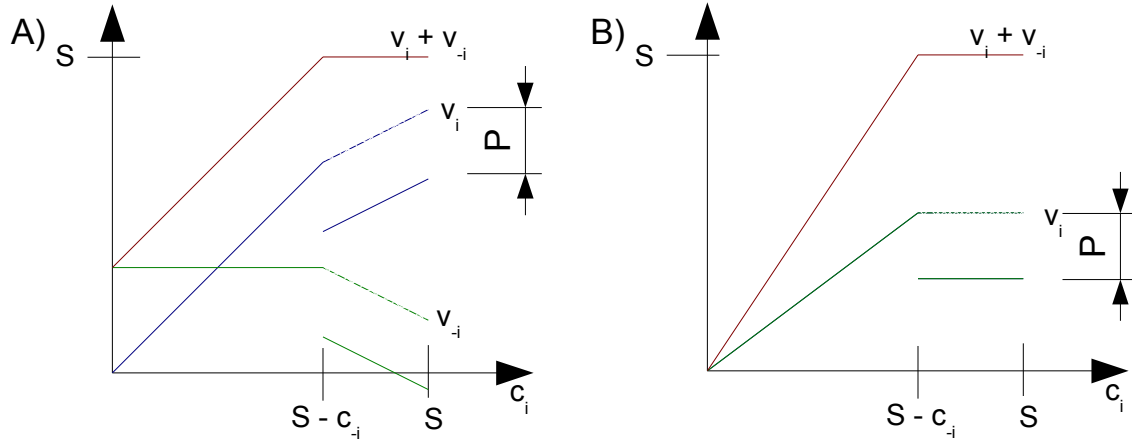
Figure 4.6: (A) shows the mechanics of the fishery prisoners dilemma payouts, varying the level of exploitation (c_i) against a constant exploitation by user $-i$. Up to $S - c_{-i}$, increases in exploitation lead to corresponding increases in value. After a level of $S - c_{-i}$, increases in exploitation are still beneficial, but the limited stock is shared. Also after this point, a penalty, P, drives down both payoffs. (B) shows a similar payout graph, if both players choose the same action.

Next consider if there is a distance d between the two, causing their compounding effects to decrease. We modify the payout equation by adding an exponentially decreasing portion of the impact of the other user, as a function of d, the distance normalized by D:

$$v_i = c_i - \mathbf{1}\{c_i + c_{-i}e^{-d} > S\}\left(\frac{c_i + c_{-i}e^{-d} - S}{2} + P\right)$$

As d increases, eventually choosing fishing sustainably under the assumption that the other user is exploiting will no longer result in the penalty of a fishery collapse. At this point, the exploitive strategy will no longer be dominant (since it is dominated by cooperation in the case where the other user is exploitive). At a further distance, cooperation will no longer be socially optimal, since the cross-user impacts will be smaller and the ecosystem use will be sustainable even when both users are exploitive. These regions are diagrammed in figure 4.7.
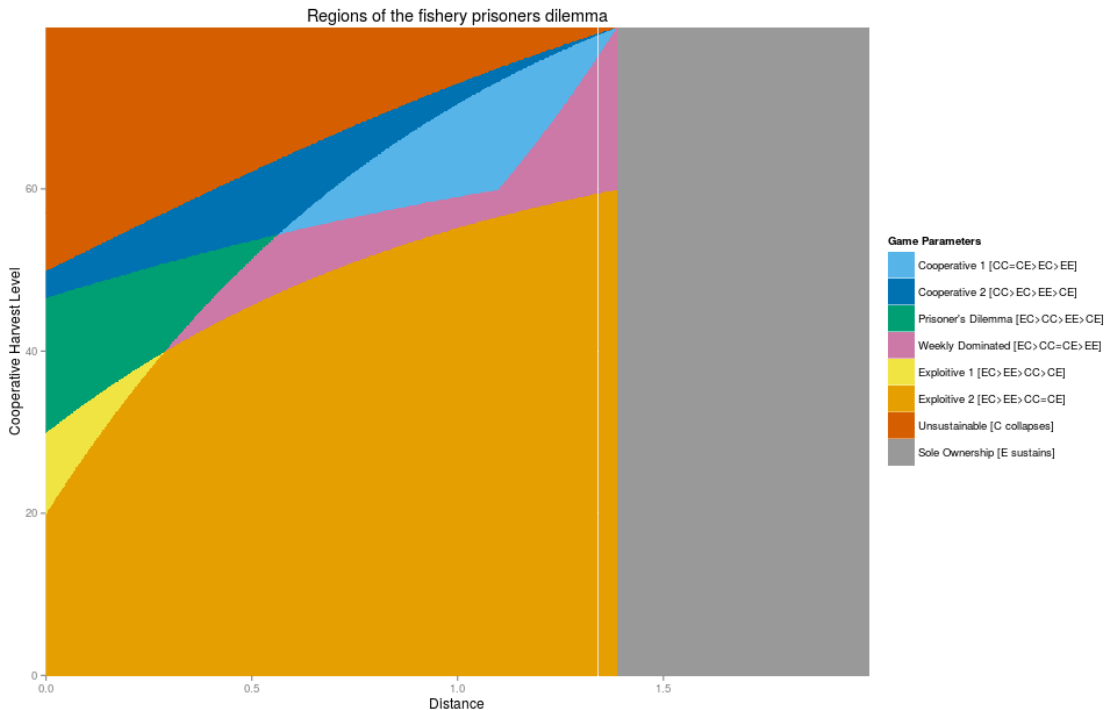
Figure 4.7: Regions for the spatial prisoner's dilemma game, based on the distance between users, d, and the "cooperative" level of exploitation, c_{Coop}. Each region is characterized by the relationships between the payouts for each pair of actions (CC = cooperative/cooperative, CE = cooperative/exploitive, and so on). At large enough values of d, even exploitive levels of use are sustainable, so cooperation is no longer socially optimal. For values of interest, producing a prisoner's dilemma for $d = 0$, as d increases, a new region appears where exploitation is only weakly dominant.

Above, c_{-i} can be treated as the combined catch of all other resource users in a simple extension of the model. In either case, recognition of the distance between users here exposes some new opportunities to avoid the tragedy of the commons.

4.7   Fishery governance for a distributed commons

Local fishing communities bear huge costs from fishery collapse. While many communities historically developed commons management practices to maintain their resources sustainably, a combination of government policies, market changes, industrial entrants, and environmental cross-border effects have undermined these regimes. This section investigates some of the ways that the distributed commons perspective illuminates responses that local fishing communities have to these difficult issues. How can local communities encourage overarching regulatory regimes that then support their ability to self-organize sustainable fisheries?

Addressing this question requires understanding the perspective of both governments and users within local communities. An example of conflict between these perspectives revolves around "outsiders", entities whose cross-border effects impact the commons. One source of outsiders is the international commercial fishing industry which over the last century has had an increasing impact on coastal fisheries in Africa (Alder and Sumaila, 2004). The government and local communities may have very different perceptions on the nature of outsiders and their consequences. For example, some agents that are "outsiders" for the fishery may actually be sanctioned by the government, such as international industrial fishers invited to develop fishery resources. Other outsiders may be national institutions, such as the agricultural or industrial sectors, producing impacts on the fishery.

The remainder of this discussion applies a focus on a particular stakeholder. For a fishing community that desires sustainable management (and understands the institutional needs by virtue of historical traditions, learning about other successes, or outside facilitation), that desire can be said to have a "nucleus"– a core group with the capacity to deliberate on their situation. The scale on which the nucleus operates defines the local scale. At that local scale and in the region of the central stakeholder, an array of other users coexist, cooperate, and compete. Other members of the distributed commons are at various distances from that nucleus: some use a fishing region which only partially overlaps; others have closer affiliations with outside nuclei.

In the interests of maximizing well-being or tax revenue, the government's first priority for a renewable resource is to maximizing the total sustainable or economic yield under

some discount rate. Depending on the characteristics of the fish species, this requires a combination of gear restrictions (to protect vital life stages or supporting species), ITQs (or other restrictions on total catch), and marine reserves (to protect vital habitats). Often, however, only a portion of the necessary practices are implemented, reflecting the political economy around entrenched stakeholders and poor communication between fishers and management groups.

The management concerns from the perspective of the nucleus are different from the government's. These include a reliable livelihood, the elimination of outside drains, and support for self-management. Scientific information could also inform these goals, but the current distrust that many fishers have for the scientific community obstructs this avenue.

Two questions are critical for the nucleus. The first is what changes in the conditions of access and usage would best support a better fishing regime. A wide range of institutional regimes coexist within the vast distributed commons that is global marine and coastal fisheries. These include catch restriction regimes (such as quotas and gear restrictions), spatial restriction regimes (such as MPAs, use rights, and exclusive economic zones), community management regimes (which variously focus on decision-making and conflict-resolving practices), and co-management regimes (which apply combinations of the three other types). Cooperative community institutions may themselves impose catch restrictions and spatial restrictions, or be embedded into a larger region where such restrictions exist. In addition, these regimes may include governance architectures and boundary institutions that specify the roles of stakeholders, scientists, economists, special interest groups (including environmental NGOs), the national bureaucracy and international community.

The research on commons suggests the best options that an overarching governance body has for encouraging community self-organization: for example, it can close the commons (through a quota system), or enforce boundaries on the commons. A more inclusive approach asks how government can support effective local institutions. Several answers are available in the literature:

- Pinkerton (1989) notes that governments can support local institutions through data gathering, protection from environmental damage, and enforcement.

- Gutiérrez *et al.* (2011) identifies empirically the management characteristics that lead

to effective co-management of fisheries, including leadership, quotas, cohesion, and protected areas.

- Gibson (2000) shows that the capacity for, and laws supporting, local property rights and rule-making plays an important role in conservation.

- Ostrom (2009) identifies the size of the resource system, its productivity, its predictability, the mobility of its units, the number of users, leadership, norms and social capital, knowledge, importance of the resource, and the ability to create rules as important factors determining the capacity of users to self-organize.

- The insight that social inequality results in poorer ecosystem management (Klooster, 2000, Holland *et al.*, 2009) may also be explored through this lens.

For classic commons, these many factors determine whether the expected benefits of managing the commons exceed the perceived costs. For a distributed commons, that result can differ in space, and the factors determine the potential scope of such management regimes.

Communities need to identify zones with leadership, equality, good communication, and the whole host of factors which support community management. Within the regional institutions, they need rights, legitimacy, and support from groups with a wider scope of information. Wilson (1982) notes that "gear conflicts or other forms of physical interference [arise] because fishermen often find it advantageous to fish very close to one another." This fact helps the establishment of local management regimes: the repeated encounters in prime fishing spots are conducive to cooperation (Schlager, 1994).

The second issue confronting the nucleus concerns methods for encouraging these changes. I will not address tactical options, which might include campaigning, lobbying, or mobilization of scientific support. Zürn (1993) argues for the importance of domestic actions in regime formation (cited in Young, 1995). The consequences of this cross-level power relation is studied in the literature on two-level games (Putnam, 1988).

In addition, the spatially heterogeneous nature of the commons provides an powerful incentive to devolve power to local authorities. Local users not only have greater knowledge of the environmental dynamics around them, but have a capacity to recognize the local social dynamics as well. Similarly, regional authorities are better equipped to recognize

regional level issues. This suggests a potential not only for a mutually beneficial information exchange, but for local users to extract rent (or better, drive power shifts) from their knowledge.

Berkes (2002) notes that "the balance of evidence from the commons literature of the past few decades is that neither purely local-level management nor purely higher level management works well by itself." Co-management, the institutionalized cooperation between regional and local governance, is a solution for handling cross-scale issues (Berkes, 2006). Co-management has the potential to mitigate the weaknesses of the management techniques used by different levels of institutions (Pomeroy and Berkes, 1997).

The distributed commons provides a natural perspective for a co-management regime, for both the government and local communities– but with different resolutions. For users within local communities, the agents of the distributed common are individual fishers. For the government, the agents are communities and commercial fishers. Many users can be encapsulated behind a single market, a single revenue stream, and aggregate metrics of well-being. At either resolution, cross-boundary and cross-scale issues continue to motivate a non-unitary vision of the commons.

By applying a model of distributed commons at local, regional, and international levels, the diverse needs and issues of fisheries management become clearer. The relevant users, resources, preferences, and impacts are all non-unitary and vary over space. Recognizing these variations can help identify places with greater potential for sustainable governance, and produce the cross-border effects which will support other regions in becoming sustainable as well.

4.8 Conclusions

The distributed commons model offers a new perspective on the commons and the potential for managing them. Many commons are not well-mixed, unitary resources. First, the community of resource users differs over space, opening the possibility of finding zones of cooperation. Second, the mixing process itself operates on a distance-scale and a time-scale that is observable to those resource users. As a result, cross-boundary flows become a

central source of conflict, and the tragedy of the commons is often mediated through them.

This chapter constructs a conceptual model of the distributed commons that remains general enough to describe a wide range of systems, but concrete enough to provide new insights. The insights fall into three main categories. First, the model explicitly captures a broader range of management options than the traditional commons model, where protected areas and cross-boundary flows can become a target for management. Second, the impacts of resource users upon each other decrease over space in an exponential fashion. The actual resources experienced by users are partly a consequence of diffusion from neighboring regions. As a result, isolated users will experience a greater extraction potential than users of a non-spatial resource; but when two users are nearby, they will cause a zone of depletion in the region between them. Third, incorporating this exponential decay into a classical prisoner's dilemma model of the commons opens up new opportunities. Most importantly, at intermediate distances, unsustainable harvesting is no longer a strictly dominant strategy.

These insights have implications for local commons management. The cross-boundary flows that produce a tragedy of the commons in the distributed commons demand co-management at both the local and regional levels. Local managers can seek zones of cooperation and leadership, and ensure support from the regional management institutions that protect, inform, and legitimize their ability to cooperate.

Chapter 7 will extend the results here through a concrete, computational model of the distributed commons. The implications of cross-boundary flows and distributed agents in commons tragedies have an enormous potential to change the ways that we perceive, study, and manage global and local commons.

Part II

Empirical Analysis

Chapter 5

Inferring Spatial Fisheries

THIS WORK WAS DEVELOPED IN 2012 IN COLLABORATION WITH KIMBERLY LAI, AND WITH THANKS TO DR. UPMANU LALL.

This chapter provides an empirical method for constructing a spatial perspective on fisheries using the most widely available fisheries data, catch records. The logistic model studied here takes a central role in the next chapter. It also includes concretely one of the fundamental claims of the Distributed Commons model: that communities at different spatial locations can affect each other through the spatial medium of fish.

The spatial dynamics of fish stocks are a growing concern in fisheries management. Fish resources often display a "patchiness", resulting from complex spatial arrangements. As spatial management becomes more complex, planners need to know what environments fish rely on during different stages of their lives. Estimates of stock abundance that do not account for spatial heterogeneity may result in over-estimation and error (Anderson and Seijo, 2011, ch. 10).

However, spatial information about fish stocks is difficult to collect. Individual fish may have widely varying spatial habits, and standard tagging methods are expensive and time-consuming. This study provides a method of estimating the "spatial reliance" of fish stocks, understood as the connection between stock recruitment and individual regions, using fish catch and spatial plankton information.

We analyze the spatial reliance of anchoveta *Engraulis ringens* near Peru. Anchoveta are
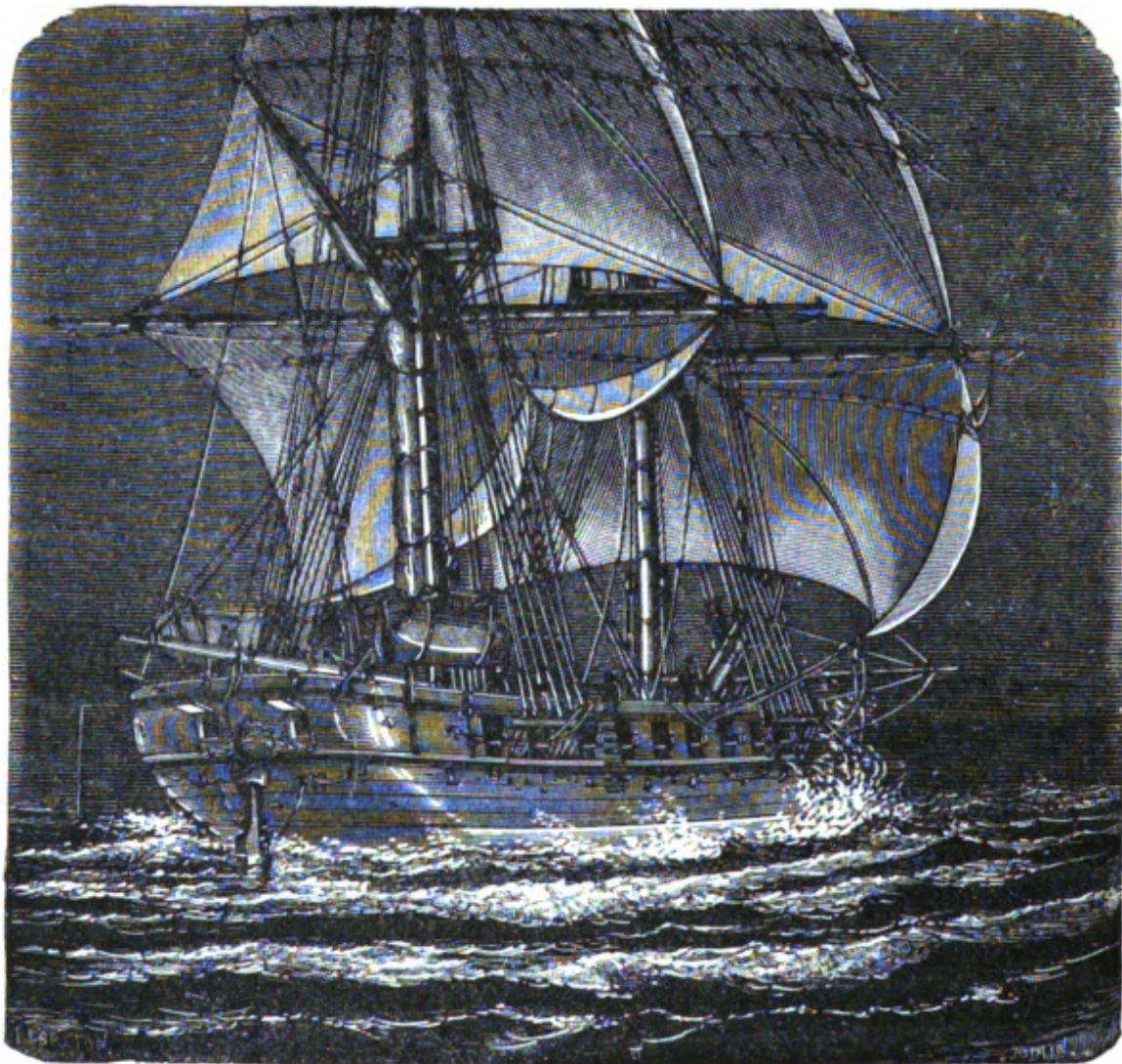
THE SHINING SEA.

Figure 5.1: Reproduced from Kirby (1884, p. 44).

the most heavily exploited fish in the world, but have widely chaotic dynamics driven partly by ENSO cycle (Iwamoto *et al.*, 2010). They consume primarily zooplankton (Espinoza and Bertrand, 2008), so this study can take advantage of a long historical dataset of plankton measurements performed by the Instituto del Mar del Peru.

5.1 Literature review

An extensive body of work on plankton and anchoveta dynamics was collected in the books *The Peruvian Anchoveta and Its Upwelling Ecosystem: Three Decades of Change and The Peruvian Upwelling Ecosystem: Dynamics and Interactions* (Pauly, 1989). Multiple approaches were used for estimating anchoveta stock.[1] The first approach involved collecting information on each step of catch-landing-processing to estimate under-reporting (Castillo and Mendo, 1987). Other studies took a more standard stock-assessment approach with spawning, stock, and egg production data (Pauly and Soriano, 1987), growth models (Palomares and Muck, 1987) and length-weight catch composition (Tsukayama and Palomares, 1987). The papers collected in these volumes also studied plankton seasonal variation (Carrasco and Lozano, 1987), the physical dynamics of the region (Brainard and McLain, 1987, Bakun, 1987), and bioeconomic modeling (Aguero, 1987).

Spatially, anchoveta occurs mainly within 80 km of the shore. The effects of El Niño on anchoveta are well known (e.g., Cushing, 1981), with their range limited by warmer and less saline El Niño currents (Iwamoto *et al.*, 2010). Bertrand *et al.* (2004) find that the spatial distribution of anchoveta changes with stock sizes, with both geographic range and density decreasing when the abundance is low.

Scientific understanding of plankton-fish dynamics has evolved over time. Originally, ecological modeling focused on modeling the physical system, including hydrodynamics such as temperature and nutrients (Denman, 1976, Weber *et al.*, 1986). It was found that such hydrophysical features were relevant only on a macro scale (dozens of kilometers or more) and on a micro scale (100 meters or less); distribution on an intermediate scale is uncorrelated to hydrodynamics. To model these distributions, researchers looked for patterns

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[1]Each relevant chapter is given a separate reference in this paragraph.

based on predator-prey distribution and developed phytoplankton-zooplankton interaction models (Scheffer, 1991, Malchow, 1994). Several papers introduced local chaos based on seasonal oscillation of parameters (Doveri *et al.*, 1993, Scheffer *et al.*, 1997), multiple interacting plankton species (Ascioti *et al.*, 1993), or diffusion of nutrient gradients (Pascual, 1993). Another branch of ecological modeling took off in the late 1990s: instead of trying to model physical relations, papers applied statistical approaches (Punt and Hilborn, 1997, Chen *et al.*, 2003). Other studies have incorporated logistic growth models (Pyo and Lee, 2003).

More recently, focus has shifted to spatial and hierarchical Bayesian techniques. Smith *et al.* (2009) develop a general model of spatial ecology and develop a method for identifying the parameters of their model from observed data. Our approach here uses a lumped model, but relaxes assumptions on the relationship between effort and catch, using a fitted distribution.

Bayesian approaches have re-integrated physical assumptions to develop multilevel priors. For example, Hiruki-Raring developed predictors such as krill density, sea ice, and seal dive characteristics in her paper on fur seal foraging and pup growth related to sea ice and prey abundance (Hiruki-Raring *et al.*, 2012). When it is not possible to calculate the joint posterior distribution, Markov chain Monte Carlo simulations are used (Mäntyniemi, 2001).

Our analysis builds on the statistical and Bayesian approaches. We apply a hidden Markov framework, informed by a logistic growth model. Combining this model with a Kriging estimate plankton concentrations throughout our region, we develop an estimate of the spatial reliance of anchoveta.

5.2 Data

This analysis draws on two key data sets. Plankton data comes from NOAA's World Ocean Plankton Database and was collected by Instituto del Mar del Peru (IMARPE). IMARPE was created in 1964 and started its plankton collection project the same year with the goal of better understanding the impacts of ENSO on the marine resources. The IMARPE casts use a Hensen net (300 m) to collect zooplankton and a 75 m surface tow net for phytoplankton.

The survey spanned the Peruvian coast and all seasons between 1972 and 2005. It includes 16,099 geolocated casts for each of phytoplankton and zooplankton (ml/m^3).
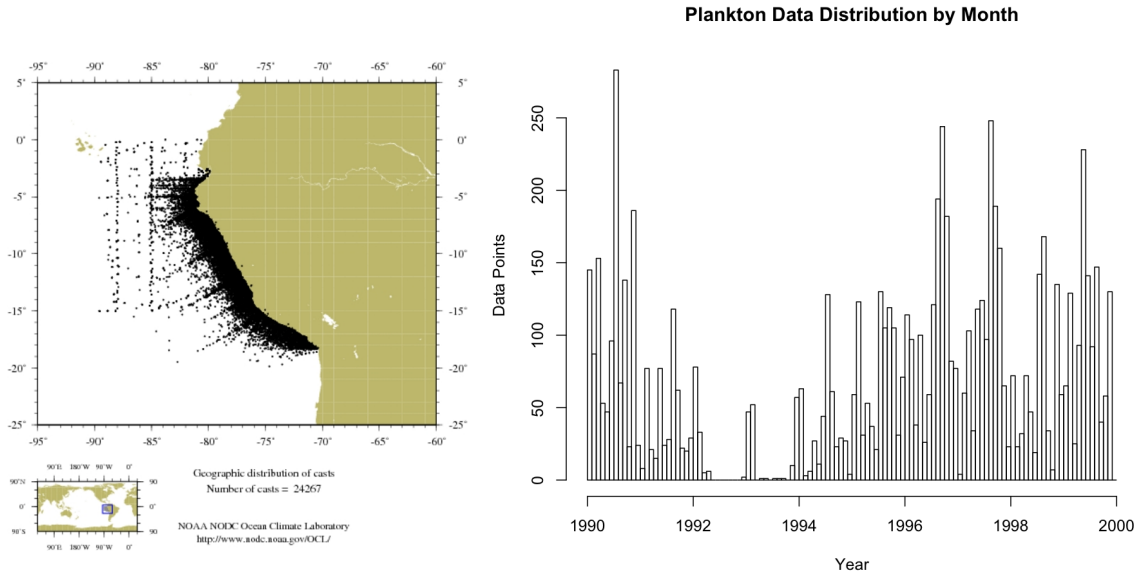
(a) *Location of plankton samples from IMARPE, 1972-2005.*

(b) *Distribution of Plankton casts by month, 1990-2000.*

Figure 5.2: Plankton casts in space and time.

The second data set comes from the International Research Institute (IRI) Eastern Pacific Pelagic Fisheries project and was collected by the Fishmeal Exporters Organization (FEO). It contains weekly anchoveta landings in tons in three ports (Paita, Chimbote, and Pisco, Peru) as well as total anchoveta landings for Peru for the years 1993 - 1999.

In the analysis, the entire collection of plankton data is used to produce a Kriging variogram, defined in the next section. In the second phase of this analysis, where plankton concentrations are related to fish recruitment, the plankton data is divided into two segments. The 6,012 plankton casts taken before 1990 are used to construct a backdrop of plankton concentrations, while the 7,926 casts between 1990 and 2000 provide the variation for identifying plankton's effects on fish populations. See supplemental section E.1.1 for a discussion of the selection of these data.

(a) *Anchoveta FEO ports from Google Earth.*

(b) *Weekly catch data to the Pisco, Chimbote, and Paita ports, and total catch for Peru.*
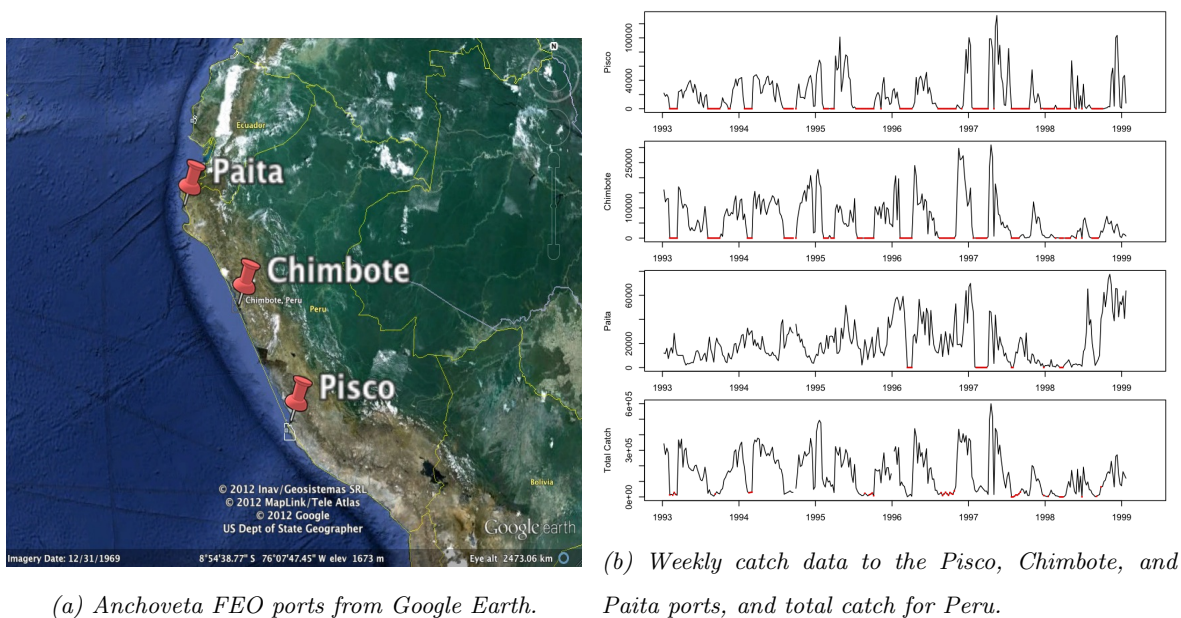
Figure 5.3: Weekly catch records for three ports in Chile.

5.3 Methodology

5.3.1 Cleaning the plankton data

The phytoplankton and zooplankton data span a wide range of values. The boxplot in figure 5.4 shows only the values above 0, on a log scale (59 phytoplankton and 89 zooplankton zero values are excluded from the boxplot, but included in the analyzed data). We removed four data values recorded for Feb. 29 of non-leap years and the zooplankton outlier (a value of 855 ml/m^3, compared to the next highest value at 66 ml/m^3).

To handle the long tailed distribution of values, the plankton data is analyzed in ranks. That is, throughout the values used to analyze plankton concentrations are the ordered rank of each value, with the rankings determined separately for phytoplankton and zooplankton.

5.3.2 Gridding the plankton data

Kriging is a method of using estimates of the spatial covariance of sampled data to construct gridded estimates of the values throughout a region. The sampled data is first used to estimate the semivariance between points at different spatial distances, displayed in a

Boxplots of Measured Values
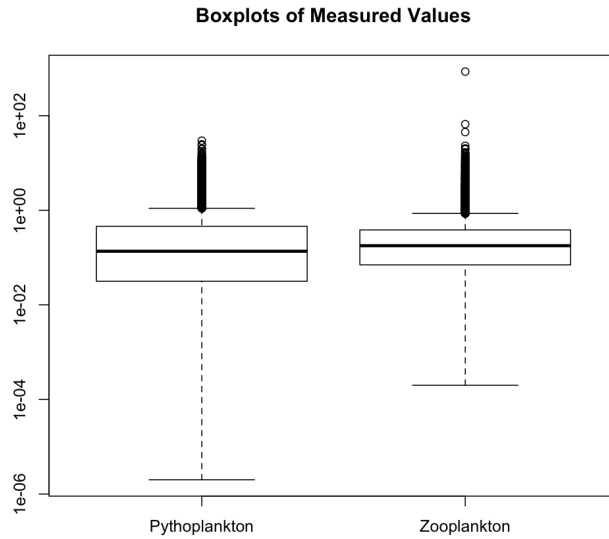
Figure 5.4: Measured phytoplankton and zooplankton concentrations. Samples with concentration measurements of zero are not shown. The large number of outliers suggests very long tails. The high zooplankton concentration outlier is dropped for the analysis.
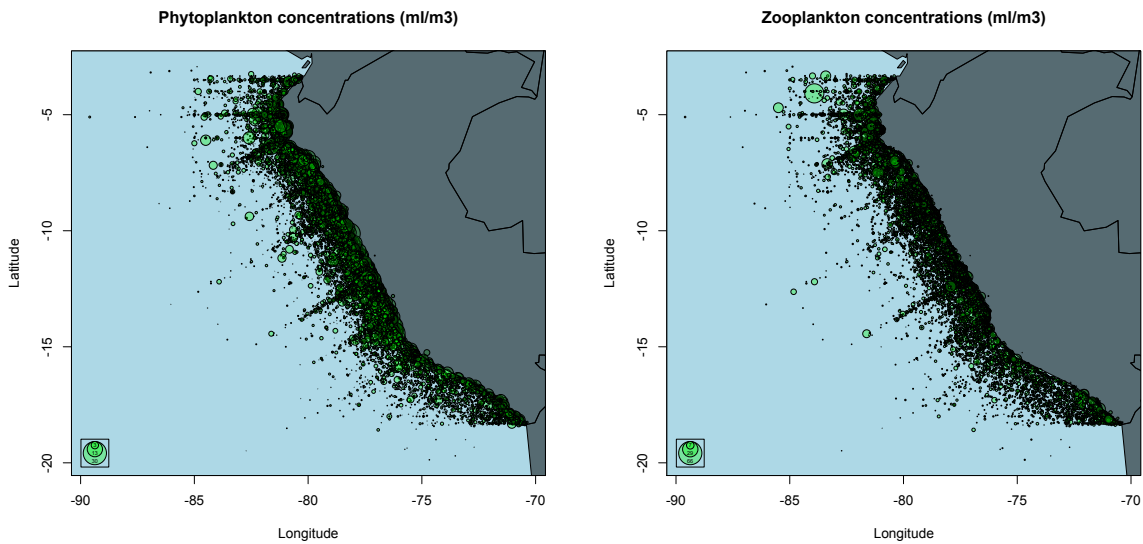
(a) Phytoplankton bubble plot   *(b) Zooplankton bubble plot*

Figure 5.5: Phytoplankton and zooplankton observed densities. Each bubble shows a single cast, with the size proportional to the concentration level. The data points are aligned along axes of latitude and longitude, although the axes are not equally scaled.

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variogram. The semivariance increases as distance between sampled points increases, reaching a maximum value where covariance between points decays to zero. An analytical model is fit to the variogram, consisting of a smooth increase in semivariance, starting from a lower value called the nugget and reaching a higher value called the sill, at which point it is constant. In the analysis below, a spherical model is used for this smooth increase (Pebesma, 2004).

We use Kriging to fill in the spatial data holes for phytoplankton and zooplankton. We treat the plankton variation is isotropic, and since Peru is near the equator that degrees of longitude and latitude are nearly equivalent. The assumption that latitude and longitude degrees can be treated as equivalent appears justified (see supplemental section E.1.2), however there is a clear difference between variation along the shore and away from the shore that is not accounted for. Separate variograms were estimated for phytoplankton and zooplankton, as ranks, and each month in the data set. The data is divided by month to minimize the amount of temporal evolution across the region. A spherical model was fit to each month's variogram (see figure 5.6), and the median values for the nugget and sill parameters across all months were used to construct a single variogram model to be applied to all time.
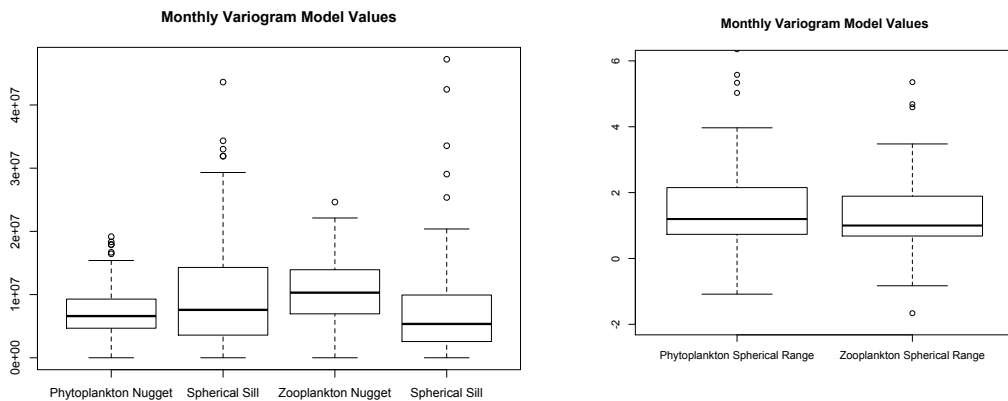
(a) Sill parameters for the variogram models

(b) Range parameters for the variogram models

Figure 5.6: Boxplots of the parameters for the variogram models. Each point represents a month between 1972 and 1990. The range parameters are in units of degrees.

Figure 5.7 shows the variogram monthly semivariance data points, with our model in blue, the mean semivariances in red and a locfit line in green.
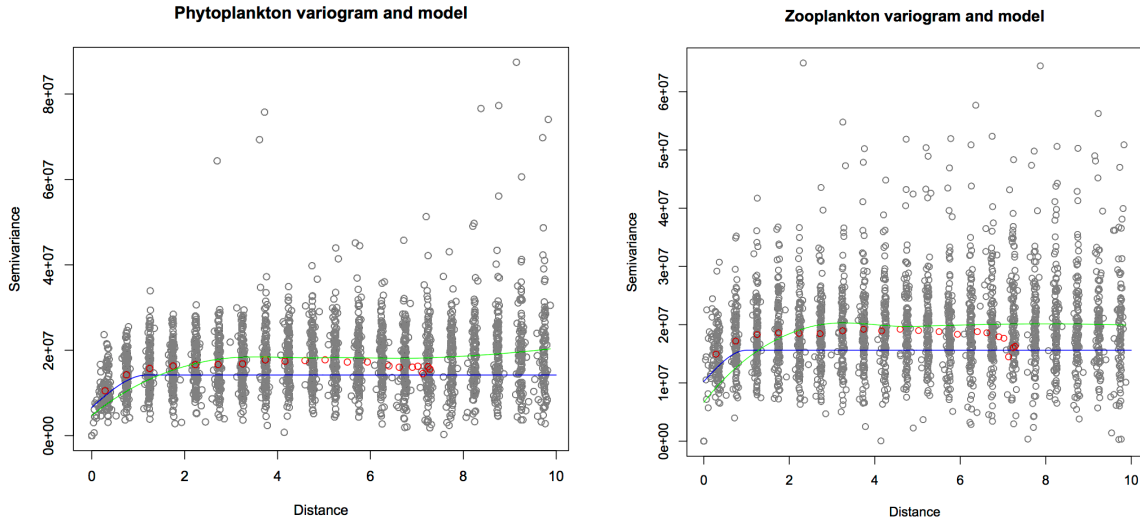
(a) *Phytoplankton variogram model verification*   (b) *Zooplankton variogram model verification*

Figure 5.7: Phytoplankton and zooplankton variogram model verification. Each point (black) represents the semivariance within a span of distances, estimated for a given month. The blue line is the variogram model used in the analysis, applying median values for each parameter. Green shows the locfit (local regression) of the data points, suggesting that a longer decorrelation distance may be appropriate. Red points show the semivariance for the entire dataset, which is higher due to loss of temporal correlation.

The resulting Kriging graphs provide good spatial coverage and temporal variability. Two sample weeks for phytoplankton and zooplankton are shown in figure 5.8.

5.3.3   Modeling fish stocks with a hidden Markov model

Next a hidden Markov model and expectation-maximization are used to estimate anchoveta stocks based on port landings. Hidden Markov models (HMMs) assume a dependency between an observed time-series and the time-series of a hidden state of the system variable (Leroux, 1992). In this case, the state of the system is the stock, and the observed time-series is the port landings.

In the model shown in figure 5.9, the observed catch (C_t) is informed both by a hidden stock variable (S_t) and by information denoting which of the ports are active ($P_{i,t}$). $P_{i,t}$

(a) *Week 1: Phytoplankton krige*

(b) *Week 151: Phytoplankton krige*

(c) *Week 1: Zooplankton krige*

(d) *Week 151: Zooplankton krige*
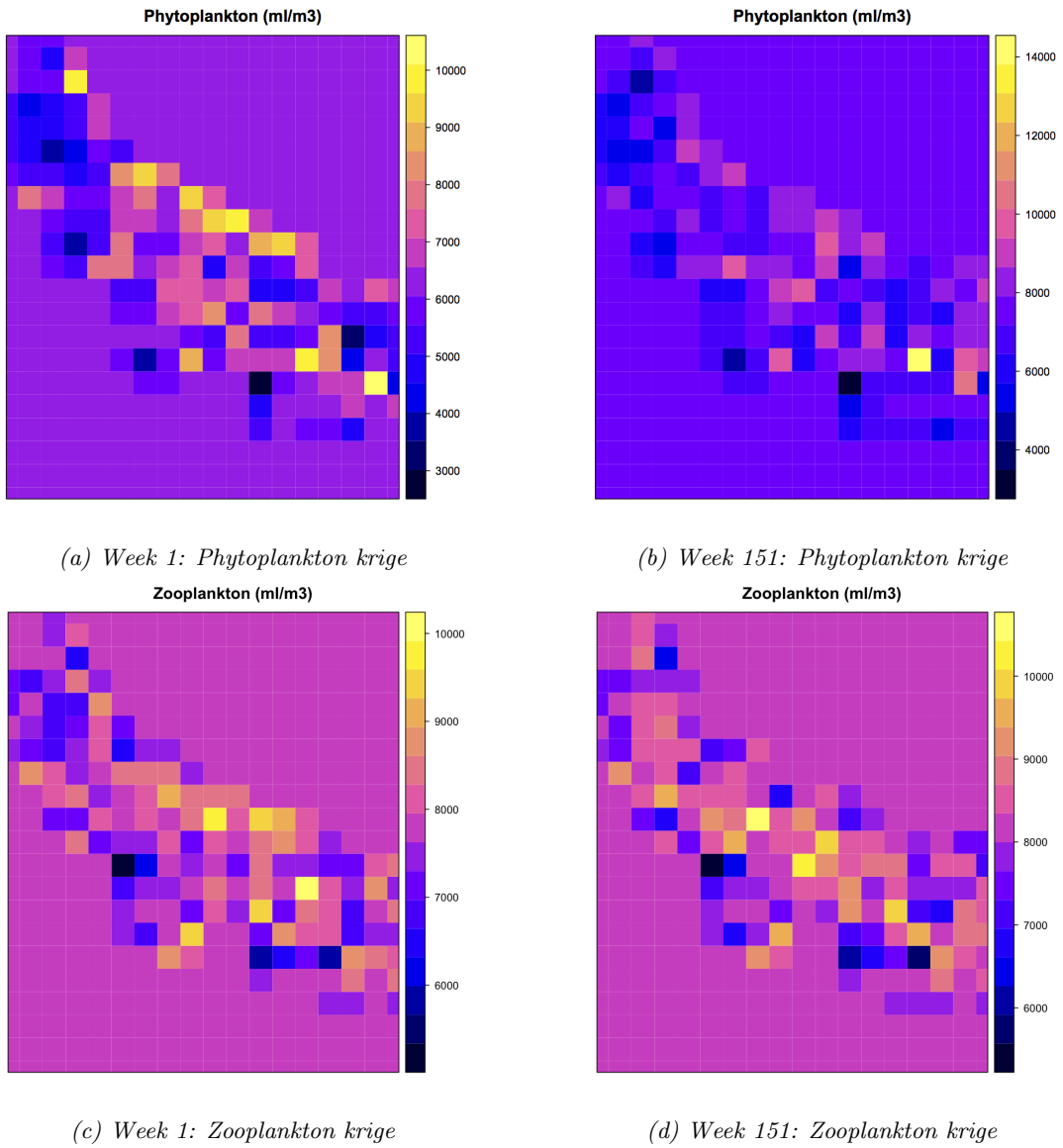
Figure 5.8: Kriging estimates for gridded phytoplankton and zooplankton, for various weeks.
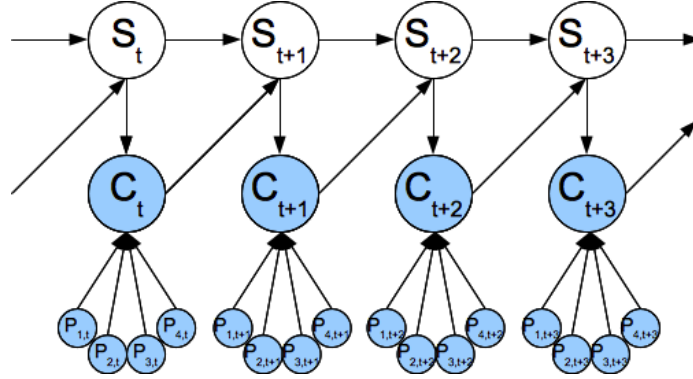
Figure 5.9: Hidden Markov Model for modeling fish stocks.

for $i \in \{1, 2, 3\}$ is 1 if the port i reported a catch greater than 0 in period t; $P_{4,t}$ is 1 if the country-wide total catch minus the sum of the available ports is greater than 0, representing the remaining undisaggregated ports. The time periods when $P_{i,t}$ is 0 are marked in red in figure 5.3.

We assume that growth follows a logistic model,

$$S_{t+1} = S_t + rS_t \left(1 - \frac{S_t}{K}\right) - C_t + \epsilon_t$$
$$= \alpha S_t + \beta S_t^2 - C_t + \epsilon_t$$

for some unknown parameters, S_0, α, and β.

The model is estimated by relating catch to the "effective stock", the stock available to those ports that are open in a given week. We also assume that the catch in each time period is drawn from a probability distribution characterized by the ratio of catch to stock level. The effective stock is

$$\bar{S}_t = S_t \sum_i \gamma_i P_{i,t}$$

where γ_i is the fractional influence of each port, with the constraint that $\sum_{i=1}^{4} \gamma_i = 1$. Therefore, we assume that there exists a probability distribution function f such that $P(C_t|\bar{S}_t) \sim f(\frac{C_t}{\bar{S}_t})$. The assumption is that the ratio of catch to effective stock is a static distribution to be estimated. This assumption is valid if the effort did not significantly change over the time period and if there are no density-dependent effects in anchoveta

schooling behavior.

The aim of the estimation-maximization procedure is to simultaneously estimate this function f, using a kernel density, and determine the model parameter estimates, S_0, α, β, and $\{\gamma_1, \ldots, \gamma_4\}$. The resulting values are those which produce a maximum likelihood under the following expression,

$$\max_{S_0, \alpha, \beta, \{\gamma_i\}} \sum_t \log f\left(\frac{C_t}{S_t(S_0, \alpha, \beta)\left(\sum_i \gamma_i P_{i,t}\right)}\right)$$

The specific steps are enumerated in supplemental section E.1.4.

The predicted stock is shown in figure 5.10.
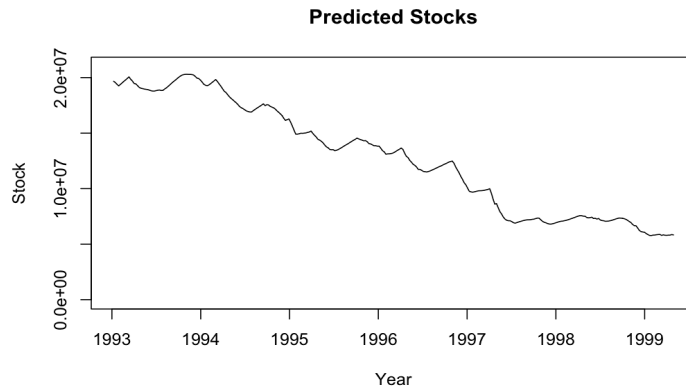
Figure 5.10: Predicted total stock aggregated across the Chilean coast.

From the total stock, we can also estimate the stock growth ($G_t = S_{t+1} - S_t + C_t$), as shown in figure 5.11. Plankton is not used in this model, but we now correlate the stock growth with plankton across space in the next section.

5.3.4 Average correlations

Anchoveta eat zooplankton, so we expect to see a correlation between anchoveta growth and average levels of plankton. That correlation could be delayed, based on the maturation and spawning time of the anchoveta. The average plankton concentrations over time give an indication of their variability (see figure 5.12).

We correlate the predicted anchoveta stock growth against measured average plankton concentrations, delayed from 0 to 200 weeks (see figure 5.14). Autocorrelation in the plankton is corrected for (see appendix E.1.3). Delayed weeks correspond to earlier life stages

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Changes in Stocks
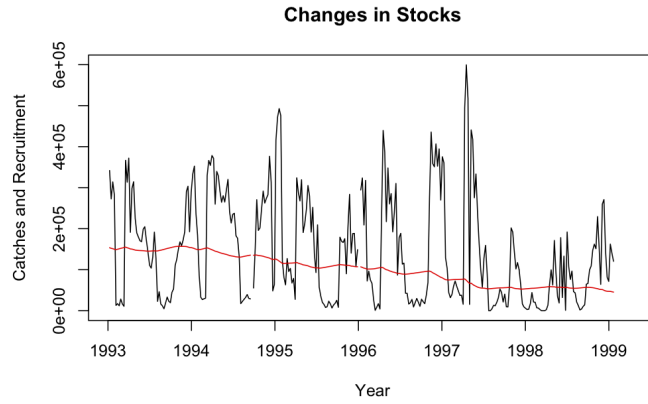
Figure 5.11: Change in predicted stock across the Chilean coast. A long-term decline in recruitment (red) extends over the period despite varying catch (black).

Average Phytoplankton and Zooplankton Concentrations
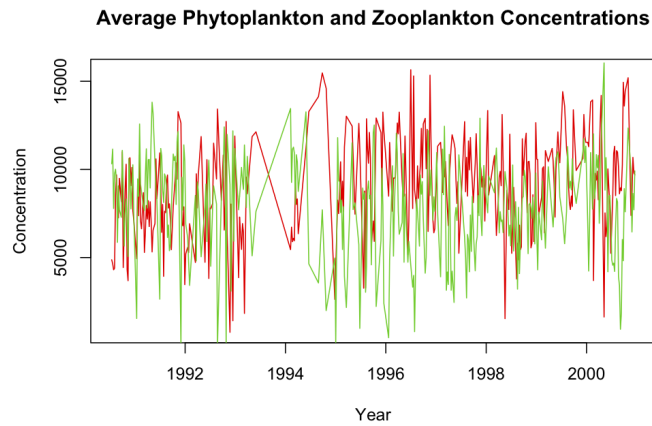
Figure 5.12: Average phytoplankton (red) and zooplankton (green) concentrations over time, from observations, showing their wide variability.

and the preceding generation. Growth occurs when fish are recruited to the fishery, at 5-6 months of age. Correlations at a delay greater than 5-6 months describe the previous generation, which was spawned 1 - 3 years before this (Iwamoto *et al.*, 2010). Figure 5.13 shows the time-line preceding recruitment. We also include some leading weeks, as a falsification test.
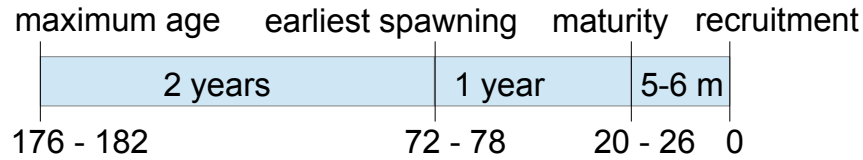
Figure 5.13: Time-line for delayed weeks in subsequent figures. The top row describes life events: the maximum and earliest birth times for the previous generation, and the time of spawning and recruitment. The middle row shows the durations of each stage, and the bottom row shows weeks before 0 (the week of recruitment).
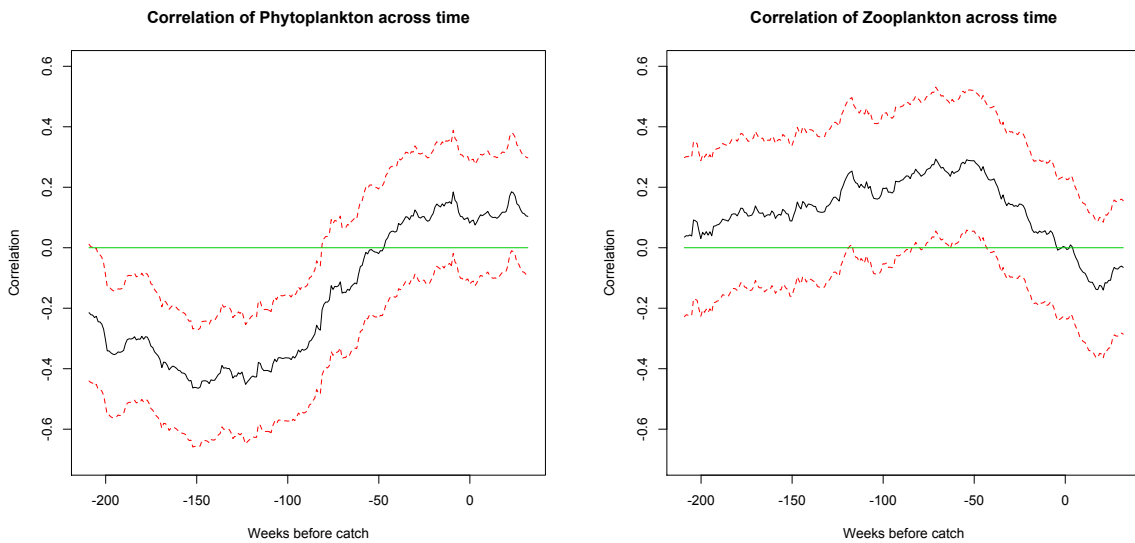
(a) Correlation between growth and phytoplankton   *(b) Correlation between growth and zooplankton*

Figure 5.14: Correlation between anchoveta growth and phytoplankton and zooplankton concentrations, with 95% confidence intervals. Each location along the x-axis is a delay between the concentrations and the growth, implying that high phytoplankton is disadvantageous to growth 150 weeks before recruitment, but zooplankton correlates positively (but unsignificantly) with growth at 50 weeks before recruitment.

Phytoplankton abundance in the preceding two to four years correlates negatively with

anchoveta stock growth. Weeks with high phytoplankton might support competing species, making it difficult for young anchoveta to grow. Zooplankton abundance correlates positively with anchoveta growth, with 95% statistical significance in the 45 to 85 weeks preceding recruitment. This reflects the reliance of anchoveta on zooplankton, particularly during the spawning period of the generation preceding the recruited generation.

5.3.5 Spatial correlation

Finally, we perform the delayed correlation between plankton and stock growth for each grid cell. We display a measure of the combined correlation, $\rho_{phyto} + \rho_{zoo} - \rho_{phyto}\rho_{zoo}$, where ρ_{phyto} is the absolute value of the correlation between phytoplankton and stock growth and ρ_{zoo} is the absolute value of the correlation between zooplankton and stock growth. This describes when either phytoplankton or zooplankton are correlated with stock growth. Where the 95% limits for the correlation of phytoplankton or zooplankton include 0, accounting for autocorrelation, that phytoplankton or zooplankton correlation is set to 0. This gives an indication of what regions anchoveta, and their food chain, draw upon for various stages of their lives.

Qualitatively, the following results are suggested by figure 5.15:

- The region of significant correlations expands across the period of delays, to its largest extents when the delays are small.

- In the period 2-3 years before recruitment, only scattered regions show significance, possibly clustered near the three ports.

- Around 1.5-2 years before recruitment, a zone of significance emerges off the southern coast of Peru.

- Around 0.5-1 year before recruitment, the spatial dependence becomes smoother over the whole of Peru.
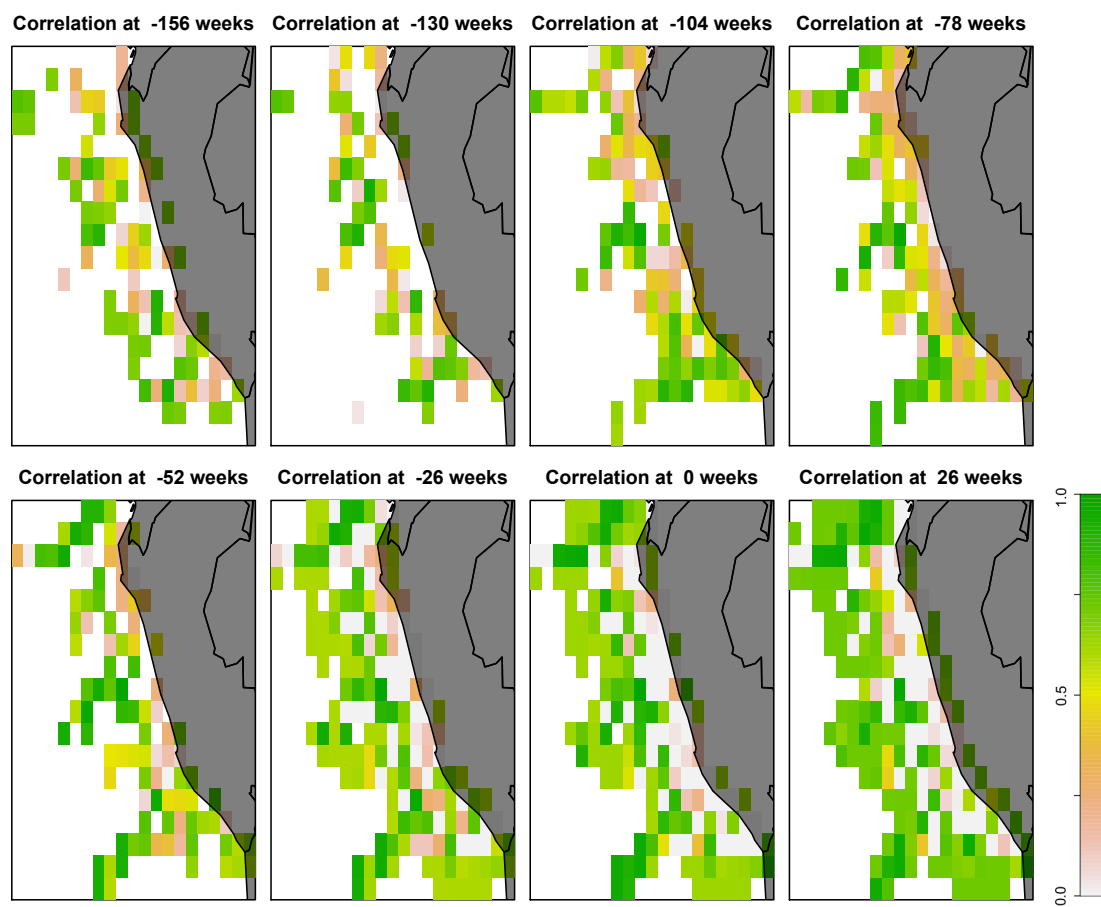
Figure 5.15: Spatial maps of the correlation between stocks and plankton, across time. The correlation values are the average of the correlation between growth and delayed stocks for phytoplankton and zooplankton.

5.4   Conclusions

It was found that stock growth correlates positively with the abundance of zooplankton in the period from 45 to 85 weeks prior to recruitment (95% statistical significance). The correlation varies considerably in space.

The varying correlations in space and time suggest that anchoveta stocks continue to shift over their lifespan, at odds with traditional "static" adult population models (e.g., Anderson and Seijo, 2011, ch. 10). These techniques provide a basis for using targeted catch restrictions to support specific cohorts, depending on the spatial dependence during that period. Additionally, the state of plankton can be used to inform the future growth of the anchoveta stock.

These results suggest a need for further research to better understand the dynamics of the plankton-anchoveta relationship. The Kriging approach helps interpolated sampled plankton across the region, allowing correlations with predicted stocks in space and time. Further improvements to the Kriging results could be accomplished by adding predictors to the variograms in our Kriging model, such as latitude, distance from shore, and the meridional Atmospheric Circulation Index, which has been associated elsewhere with increases in Anchoveta population (Kliashtorin, 2001). Closer study of the correlations between phytoplankton and zooplankton could also help to shed light on the reasons for the time-lagged negative correlation between phytoplankton and anchoveta.

Chapter 6

Global Benefits of Marine Protection
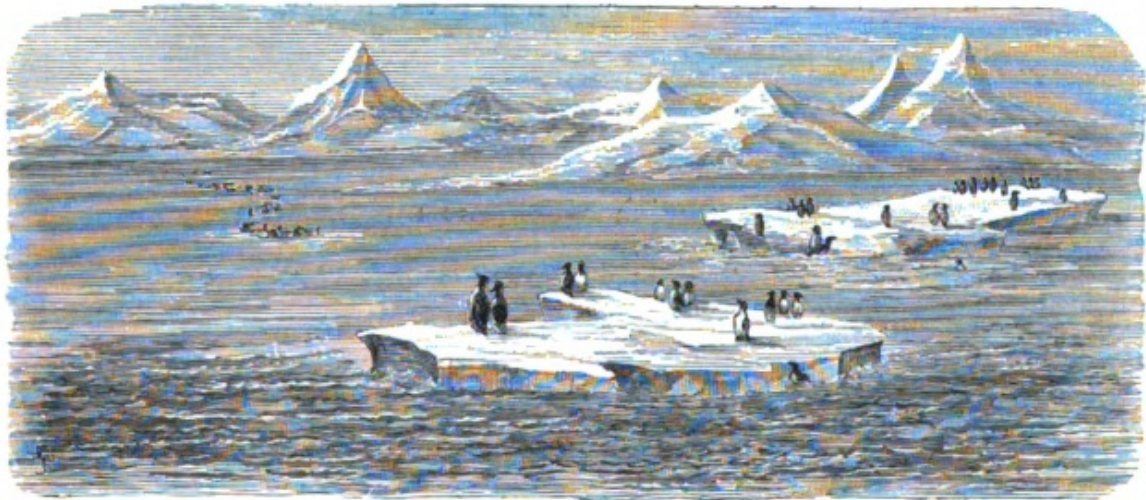
ISLAND OF PENGUINS.

Figure 6.1: Reproduced from Kirby (1884, p. 229).

THIS WORK WAS DEVELOPED BETWEEN 2012 AND 2015 IN COLLABORATION WITH GEOFFREY HEAL.

This chapter analyses the central outcome of marine spatial planning, the marine protected area. Significantly, the full expression of its effects requires a spatial and cross-boundary perspective, as developed in previous chapters. The results produced here further

inform calibration of the computational, spatial model of the Distributed Commons.

Marine Protected Areas (MPAs) are regions within which human activity is restricted in order to preserve ecosystems or historical and cultural features. MPAs, and particularly their strictest form, the no-take reserve, can be very powerful tools for conservation. Within many MPA boundaries, fish biomass, density, average size, and species richness increase dramatically (Lester *et al.*, 2009). MPAs also have the potential to benefit adjacent fisheries. These effects have been studied as "benefits over boundaries", in which increased fish biomass diffuses into exploited areas (Gell and Roberts, 2003), and as "savings accounts", able to buffer fluctuations and support resilience (Grafton *et al.*, 2010).

Theoretical models (Polacheck, 1990, Holland and Brazee, 1996, Tuck and Possingham, 2000, Pezzey *et al.*, 2000, Sanchirico and Wilen, 2001a, Neubert, 2003) and computational models (Walters *et al.*, 1999) have explored the potential for MPAs to benefit exploited fisheries, but previous empirical studies have focused only on specific regions and species (e.g., Sluka *et al.*, 1997, Murawski *et al.*, 2000, Roberts *et al.*, 2001, McClanahan, 2010). This study is the first global analysis of the benefits of MPAs for harvested fish stocks. We exploit temporal and spatial variation in protected areas as predictors of stock surplus, as new areas are designated as MPAs within the management boundaries of exploited stocks. The designation of new MPAs amounts to a natural experiment whose consequences we study.

The first section describes our dataset, which combines global databases of MPAs, stock assessments, catch time-series, and sea surface temperatures. In the second section, we perform a set of treatment-and-control analyses, by comparing stock assessments in regions that had a concentrated growth of protected areas to those which did not. The third section constructs a flexible model with stock-specific parameters, based on the logistic growth equation. We treat MPAs as mechanisms for enhancing the growth rate parameter of a stock, and explore a number of model formulations. The fourth section studies a regression tree of the stock-specific MPA effects, as a way of identifying which stock and region characteristics drive positive results. The fifth section estimates the economic benefits of MPAs, and compares these to country-specific costs for maintaining the protection. The sixth section develops a model that accounts for non-linearity in the effects of MPAs and

	All	Marine	No Take
Available:	4259	2489	1493
Non-zero:	4255	2489	381
Median (km^2)	8	4.4	.7
Mean (km^2):	1166	1171	203
Total (km^2):	5.0e6	2.9e6	3.0e5
% of Ocean:	*1.4%*	0.81%	0.08%
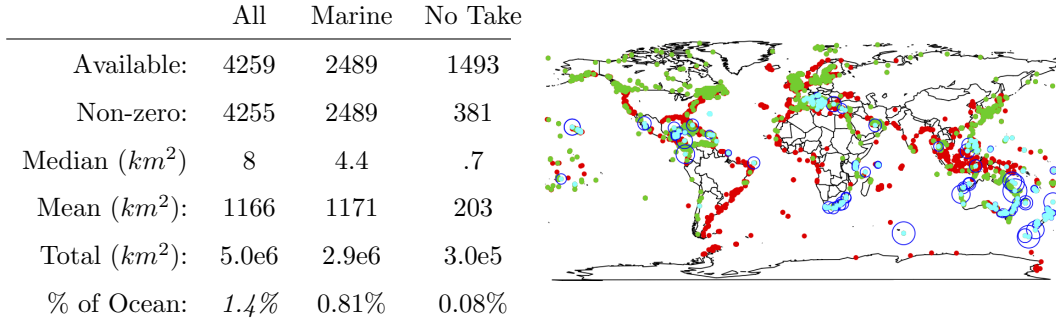
Figure 6.2: **Left**: Summary statistics for the MPA Global database. All entries are include in the left column, while only entries that distinguish marine area from coastal area, and no-take area are included in the middle and right columns. The "% of Ocean" row shows the portion of the full ocean area included in the MPAs. However, the denominator for calculating this value in the "All" column is inaccurate, since the total protected area sometimes includes non-marine area. **Right**: Map of MPAs in the MPA Global database. Red dots denote MPAs with only total protected area information available; green dots denote MPAs with non-zero marine areas (corresponding to the "Marine" column on left); and blue dots denote MPAs with non-zero no take areas (corresponding to the "No Take" column). Blue circles around blue dots show the relative sizes of the no take MPAs (but are not to-scale with the geography).

stock-size dependent effects.

6.1 Data

As described below, our main results rely on the MPA Global database (Wood, 2007) and the RAM Legacy Stock Assessment database (Ricard *et al.*, 2011). Economic results additionally use data from Sea Around Us (Sea Around Us, 2007), and our localization result applies maps from the AquaMaps project (Kaschner *et al.*, 2007).

The MPA Global database includes 4383 MPAs, along with their designation date, location, IUCN category, and area (often specifying the marine portion and no-take portion).[1] Figures 6.2 and 6.3 summarize this data.

We use stock assessments from the RAM Legacy Stock Assessment Database. The RAM database contains 336 assessments, with an average of 5 time-series variables across

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[1]MPA regions can include segments of land, such as islands and shoreline. The marine area of an MPA excludes this area.

Number of MPAs designated

Total Area of MPAs designated
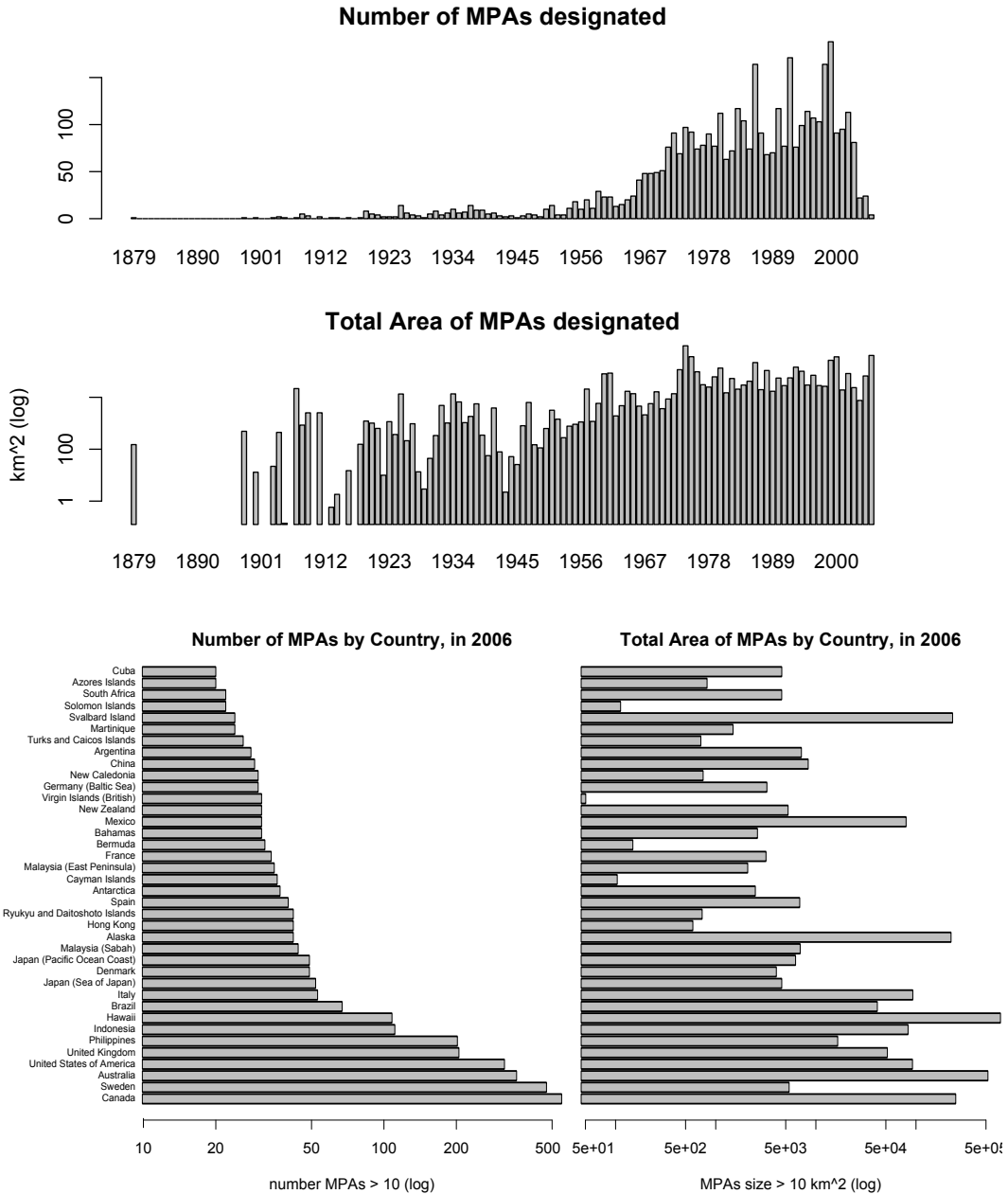
Figure 6.3: **Top**: Total MPAs and total MPA area (on a log scale) designated by year. More than 50 new MPAs have been designated yearly since 1970, and over 1000 km² of ocean has been protected yearly since 1956. **Bottom**: MPAs and total MPA area by region. Regions designations are from the Sea Around Us project, which divides some countries into smaller regions (for example, Japan is divided into two coasts). Countries have taken very different approaches to designating protected areas, and the regions with the greatest number of MPAs do not always have the greatest area protected. For example, while Sweden has the second most protected areas, it is ranked 27th by total area protected.

46 years per assessment. The assessments vary considerably in the variables they include, but of these, 285 assessments contain yearly values which can be interpreted in terms of metric tons (MT) for both the stock and yearly catch, for an average of 36 years. These are the assessments that will be analyzed below. The assessments are biased toward developed countries, with 42% from the U.S. another 40% from other OECD countries. Half of the remaining are from international organizations, and the remaining half from poorer countries. Unassessed fisheries may differ significantly from assessed ones (Costello *et al.*, 2012), and we do not attempt to resolve this uncertainty.

Throughout this paper, we will study the effects of MPAs on the yearly surplus or recruitment (which we treat as equivalent), defined as the change in stock level from one year to the next, plus the catch in that year:

$$R_{it} = S_{it} - S_{i,t-1} + C_{i,t-1}$$

Catch includes discards where the data is available. This measure reflects the total productivity of the stock, whether it is appropriated by fishers or not.

We approximate bounding regions for these assessments, using FAO and country management region shapefiles (see appendix F.1). Of the assessment regions in the RAM database, 274 had additional MPAs designated within them during the years when stocks were assessed. Figure 6.4 shows the spatial distribution of these assessments.
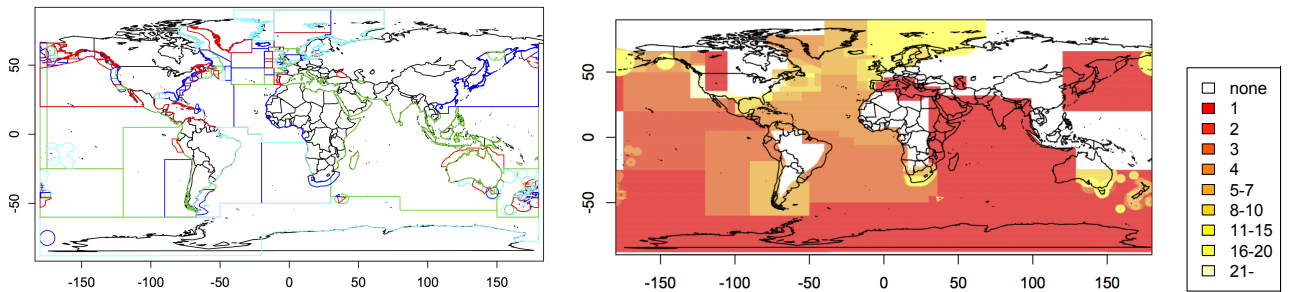
Figure 6.4: **Left**: Geocoded assessment regions from the RAM database. **Right**: Heat map of number of assessments used in the analysis.

We combine MPAs within assessment regions to generate a consistent measure of protection which we use across most of the analyses below. This is the portion of an assessed

region under various forms of MPA designation, as it varies in time.

$$\frac{MPA_{it}}{Area_i} = \frac{1}{Area_i} \sum_{\substack{j \text{ for} \\ Centroid_j \in Region_i \\ \text{and } t_j < t}} MPA_j$$

where MPA_{it} is the total protected area in assessment region i and year t. Assessment region i encompasses an area $Area_i$. The sum on the right includes an individual MPA j if its centroid is within the assessment region and it was designated before year t.

In addition, catches, landed values, and shelf areas by country and distinct sub-country marine regions were collected from the Sea Around Us Project (Sea Around Us, 2007).

Temperature provides a key control. Environmental variability, through temperatures and indices based on them (such as ENSO, NAO, and AMO) have been shown to strongly predict recruitment (Keyl and Wolff, 2008). Changes in sea surface temperatures (SSTs) can have effects throughout the food web, although the effects can vary by species and region. Along the west coast of South America, warm temperatures can signal a shutting-off of nutrient-rich cold water upwelling due to El Niño conditions, resulting in decreased productivity and increased mortality for many stocks. In other stocks, such as scallops in the same area, warmer temperatures increase growth rates (UCAR, 1994). Long-term temperature changes are also shifting stock ranges producing more complicated impacts (Cheung *et al.*, 2013).

We calculate the average SST anomaly for each region and year, using the Extended Reconstructed SST dataset from NOAA (Smith *et al.*, 2008). Estimates that account for SSTs include temperature anomalies from the two years preceding the measured stock growth, further capturing some cohort effects.

Of the fish assessed in the RAM database, 121 species have fish distribution maps available in the AquaMaps database (Kaschner *et al.*, 2007). These maps are show in appendix figure B.1. Fish distributions offer a way to refine the effective protection available to each species.

SSTs are averaged over assessment regions and fish population distributions are summed over circular regions representing each MPAs. Since both SSTs and AquaMaps data are provided on a grid, we generate values across regions in an analogous manner. The average

temperature in a region A is approximated by the average value for all grid centers within the region:

$$T_A = \frac{1}{|A|} \iint_{x,y \in A} T(x,y)\, dx\, dy$$
$$= \frac{\sum_{x,y \in A} T(x,y)}{\sum_{x,y \in A} 1}$$

For fish distributions within an MPA, the effective protection provided by an MPA covering the area A for a fish with distribution $D(x,y) \leq 1 \,\forall\, x,y$ is,

$$D_A = \iint_{x,y \in A} D(x,y)\, dx\, dy$$
$$= |A| \frac{\sum_{x,y \in A} D(x,y)}{\sum_{x,y \in A} 1}$$

If the region is too small to encompass any grid centers, the closest grid point is used ($T_A = T(x,y)$ and $D_A = |A| D(x,y)$).

6.2 Treatment and control analysis

We first divide assessments into "treated" and "control" assessments. This is similar to BACI (Before-after-control-impact) approaches which compare changes in reserves to changes in similar control regions.

Treated regions are those which had a period of very little change in their MPA portion, followed by a short period in which proportionally large areas were protected, followed by another stable period. Control regions are those for which, over the entire timeseries, the protected portion never increased beyond a threshold. Formally, a region is categorized as a treated region if there exist points in time t_0, t_1, t_2, t_3, such that $t_0 < t_1 < t_2 < t_3$ and

$$\frac{MPA(t_1) - MPA(t_0)}{Area} < \mu$$
$$\frac{MPA(t_2) - MPA(t_1)}{Area} > \nu$$
$$\frac{MPA(t_3) - MPA(t_2)}{Area} < \mu$$
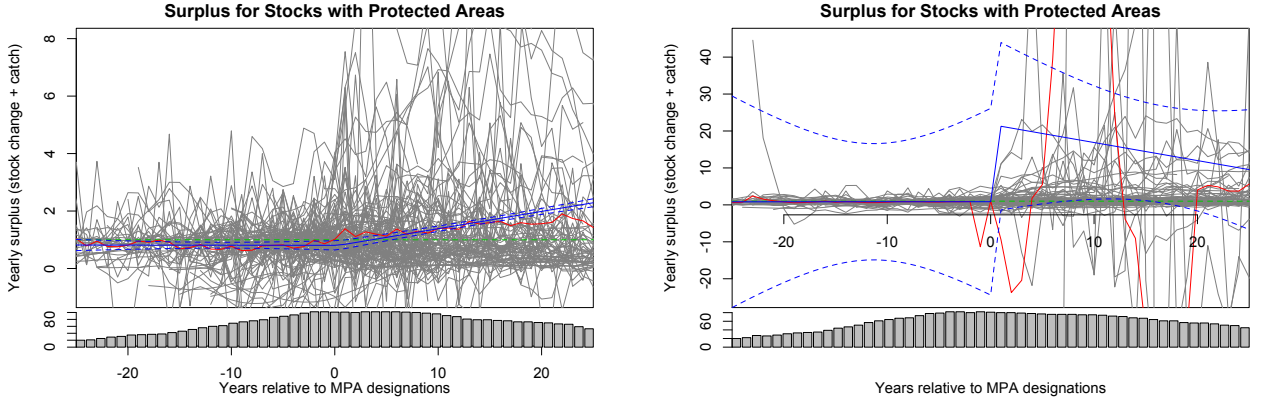
Figure 6.5: Evolution in time of stock surpluses, for "treated" stocks (left) and "control" stocks (right). The histogram below each graph shows the number of assessments available for each year. All stocks are normalized to a pre-treatment average of 1, with control regions applying their matched treatment period (green line). The red line shows the average stock surplus in each year. In the left graph, the blue shows estimate and confidence average surplus and post-treatment trend.

and a control region if

$$\frac{MPA(t_3) - MPA(t_0)}{Area} < 2\mu$$

The values for t_0, t_1, t_2, and t_3 are determined on per-region basis, selected for the ranges that produce the longest baseline and treatment periods, by maximizing $(t_1 - t_0)(t_3 - t_2)$.

We use $\mu = 0.15\%$ and $\nu = 0.5\%$, which produces 102 treated regions and 132 control regions.[2] Treatment regions are matched to control regions as described in appendix F.2, using a combination of stock dynamics and location. We assume that the creation of reserves in the treated areas does not also affect effort in the control areas. Normalized time-series of the surplus of these two collections show a strong effect (see figure 6.5).

A difference-in-difference regression supports this result. The model is as follows:

$$\frac{R_{it}}{Baseline_i} = \beta_0 + \beta_1 T_{it} + \beta_2 P_{it} + \beta_3 T_{it} P_{it}$$

$R_{it} = S_{it} - S_{i,t-1} + C_{i,t-1}$ is the surplus for stock i in year t. For each observation, T_{it} is 1 if the stock is treated and 0 otherwise, and P_{it} is 1 if the time is after t_2 and 0 otherwise. Control stocks use the same year for t_2 as their matched treated stock.

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[2]These values are somewhat arbitrary, and chosen to maximize the number of treated and control regions while still providing separation between them.

	No Temp.	With Temp.
Treated (β_1)	0.14	0.15
	(0.13)	(0.13)
Post (β_2)	0.53	0.52
	(0.38)	(0.37)
Treated $\cdot$ Post (β_3)	0.34	0.30
	(0.32)	(0.31)
Year	yes	yes
Temperatures	no	yes
R^2	0.02	0.02
Adj. R^2	0.01	0.01
Num. obs.	5558	5558

$^{***}p < 0.001$, $^{**}p < 0.01$, $^{*}p < 0.05$

Table 6.1: Two difference-in-difference regression models of the effect of MPA designation. The left and right models are identical, except for additional temperature controls in the right model.

The values are normalized as in figure 6.5, by dividing by the average surplus over the pre-treatment period (from t_0 to t_1), with time periods for control stocks again taken from their matched treated stocks. This result is shown in table 6.1.

Treated stocks are slightly better off prior to treatment, and all stocks improve later in their assessments. However, treated stocks show an 30% improvement in surplus, relative to matched control regions. None of these coefficients are significant under robust standard errors clustered at the assessment level, but provide suggestive relationships. The next section uses a more flexible, biologically-motivated model to capture endogenous growth.

6.3 Logistic results

We use the logistic growth model to estimate the growth of each fish stock over time, given stock and harvest levels. While logistic growth models are unnaturally simplistic, they are appropriate in situations when age class and ecosystem information is missing. The

Gompertz model is also reasonable in these context, and an analogous analysis is done for it in appendix F.8.

Under logistic growth,

$$R_{it} = rS_{i,t-1}\left(1 - \frac{S_{i,t-1}}{K}\right) = rS_{i,t-1} - \frac{r}{K}S_{i,t-1}^2$$

We hypothesize that the effect of MPAs is a function of the portion of the total assessed region that is protected. In the main model specification (reported below), the protected portion produces additional surplus through changes to the logistic growth rate, r. The ratio $\frac{r}{K}$ is assumed to remain constant. This assumption suggests that increases in r are balanced by proportional increases in K, and is supported by the data.[3] An increase in r represents more productivity for a given stock level, as a consequence of supporting environmental and ecosystem services.

Protected areas provide an immediate benefit to fishers by providing a greater yearly surplus production. If the new growth rate is expressed as a factor increase over the old value, $r_1 = ar_0$, then the new maximum sustainable yield by the square of the increase:

$$MSY = arS_{MSY} + \frac{r}{K}S_{MSY}^2$$
$$S_{MSY} = a\frac{K}{2}$$
$$MSY = a^2\frac{r_0 K}{4}$$

By combining the stock assessment time series with the area protection history, the marginal benefit of protecting an additional portion of stock's range can be estimated directly. We use a linear model, estimated by least squares regression with robust standard errors clustered at the assessment level:

$$R_{it} = \left(\gamma_i + \beta\frac{MPA_{it}}{Area_i} + \zeta_{1,i}T_{i,t-1} + \zeta_{2,i}T_{i,t-2}\right)S_{i,t-1} + \kappa_i S_{i,t-1}^2 + \delta_i + \epsilon_{it}$$

Logistic growth is represented here through coefficients estimated on $S_{i,t-1}$ and $S_{i,t-1}^2$, and estimated independently for each stock. The marginal change in growth rate, β, is

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[3]Models that include a term $\frac{MPA_{it}}{Area_i}S_{it}^2$ find that the corresponding coefficient is not significantly different from zero.

the parameter of interest, and $MPA_{i,t-1}$ is the area protected within the stock assessment region, $Area_i$. The two previous years' average temperature anomalies are included ($T_{i,t-1}$ and $T_{i,t-2}$) as controls on the growth rate.[4] δ_i is a stock fixed effect. ϵ_{it} is a normally distributed error term. In addition, we include time as stock-specific trends ($\theta_i t$) and year fixed-effects (ϕ_t) in some formulations.

The results are shown in table 6.2, for total protected areas, and for marine-only portions of protected areas, where the data is available.

The numerical values of these results are difficult to interpret, since each stock has a distinct baseline growth rate. The distribution of growth rates is shown in figure 6.6. For the histogram, we estimate growth rates after accounting for the global MPA effect, constraining them to be positive. The median growth rate is 0.34. According to the fifth column model (including trends and year fixed effects), increasing the protected area portion by 10% would increase this median growth rate by 50% and the MSY by 125%. The distribution of these percentage increase in growth rate per percent increase in protected area is also shown in the lower-right graph of figure 6.6.

MPAs often include non-marine regions, such as islands and shoreline, and the marine-only regressions only include the portion of each MPA that is specified as marine. Results for marine-only MPAs tend to be less significant but numerically greater. This is expected, since these regressions include only the 60% of the globally designated protected area which specify the marine portion of their area (see figure 6.2). The observations are at the assessment-year level, so the number of observations does not change between the total and marine-only MPA data, even though less MPA area and fewer MPAs are counted in the marine-only results. These results suggest that marine protection is more beneficial to fish stocks than protection of their adjoining land.

We also consider models that allow for changes in K, a relative change in r, phasing in of the effects of a time horizon (appendix F.12), and inclusion of only IUCN categorized protected areas (appendix F.5).

The spatial distribution of positive and negative estimated benefits from MPAs is displayed in figure 6.7.

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[4]This modification allows growth rates to vary systematically, in line with Carson *et al.* (2009).

	No Temp.	Timeless	Stock Trends	Year FE	Trends-Year
Data: MPA variables represent Total MPA areas					
β	0.7872	0.9369*	1.7077***	1.0124**	1.6860***
	(0.7530)	(0.5179)	(0.3685)	(0.4486)	(0.3731)
R^2	0.5280	0.6042	0.6375	0.6076	0.6398
Adj. R^2	0.4844	0.5391	0.5636	0.5402	0.5635
Data: MPA variables represent Marine-only MPA areas					
β	1.1983	2.1367	10.4473	2.9363	10.2263
	(1.9896)	(4.5583)	(6.7117)	(4.4884)	(6.6549)
R^2	0.5277	0.6040	0.6366	0.6076	0.6390
Adj. R^2	0.4841	0.5389	0.5625	0.5401	0.5625
Temperatures	*no*	*yes*	*yes*	*yes*	*yes*
Stock trends	*no*	*no*	*yes*	*no*	*yes*
Year FE	*no*	*no*	*no*	*yes*	*yes*
Num. obs.	10093	10093	10093	10093	10093

***$p < 0.01$, **$p < 0.05$, *$p < 0.1$

Table 6.2: The estimated marginal change to the growth rate for protecting a portion of an assessment. To capture this entire benefit, 100% of the region would need to be protected, a value which is outside of the range of our data. The first column has no temperature or time regressor, while all other columns include temperature controls. The second column includes temperatures, but not time. The middle third includes assessment-specific trends. The fourth column has a fixed effect for each year. The fifth column includes both year fixed effects and stock trends.
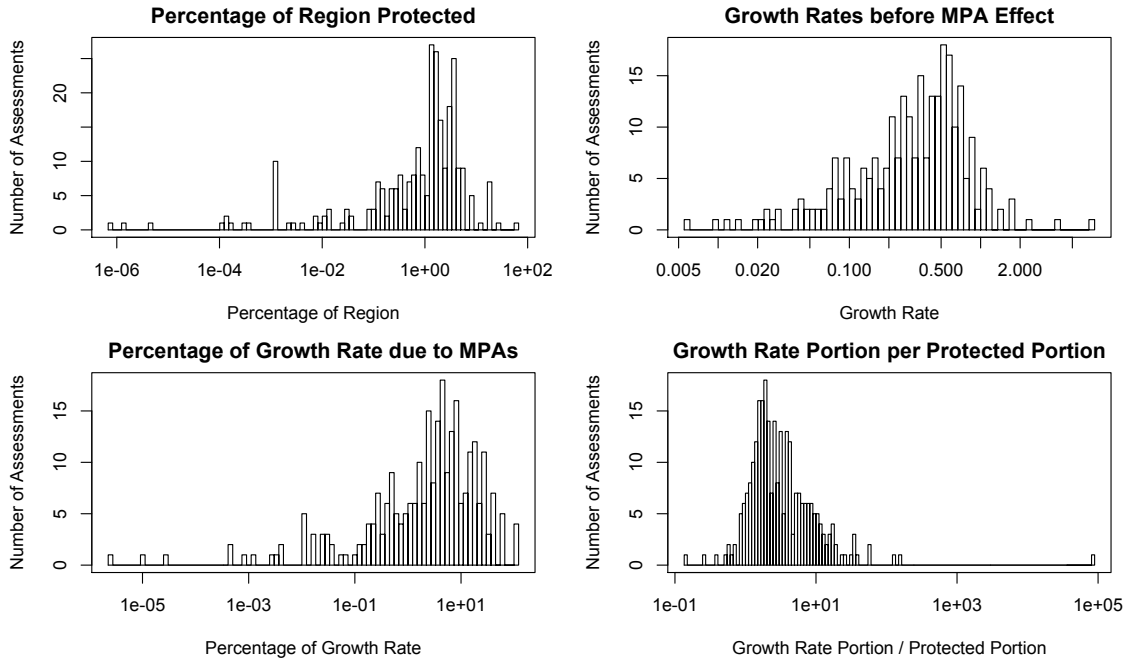
Figure 6.6: **Upper-left:** A histogram of the protected portion of each assessed area. The middle 50% have protected portions between .3% to 2.8%. **Upper-right:** A histogram of the growth rates, before accounting for protected areas, according to the fourth column (time FE) model and for growth rates estimated as non-negative. **Lower-left:** Percentage of the observed total growth attributable to protected areas in 2006. **Lower-right:** Relationship between marginal increases in protected areas and increases in growth rates.
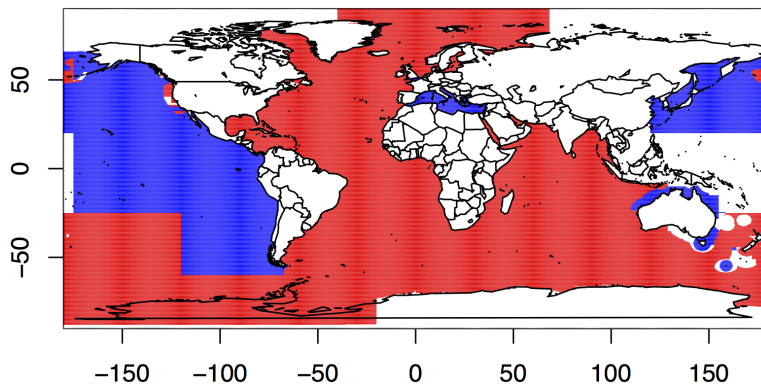
Figure 6.7: Regions with positive and negative estimated effects from MPAs. Blue regions have most assessed stocks showing positive benefits; in red regions most assessed stocks showing negative benefits.

Stock assessments for exploited fisheries estimate only the portion of the fish population available to fishers. As such, this estimate only describes the benefit of protected areas to exploited populations, either due to cross-boundary recruitment or fish movement. Most assessed regions have a small fraction of their area protected (the median protected portion is 1.4%; see figure 6.6), but the impact on the fish population outside of this region is detectable.

6.3.1 Simulation experiments

These estimates rely on variation in the MPA area over time. In general, the timing of protection designation is endogenous to the greater socio-ecological fisheries system. Moreover, some past analyses of the impacts of management suffer from identification problems that conflate the role of catch restrictions with other forms of management (Bromley, 2009, Nowlis and Van Benthem, 2012). This sections considers the potential for the measured effect of MPAs reported here to represent other forms of management.

First, we argue that much of the variation in our independent variable is exogenous. The exact year of designation is difficult for members of the system to predict, given the multiple scales at which decisions need to be made and the often delicate discussions between stakeholders that can take many years. The protected areas that we study are also rarely designed to support exploited fisheries productivity, since evidence of this process has only emerged in the last decade.

Forms of fishery management that operate through catch restrictions (e.g., ITQs, season restrictions, and horsepower limits) are captured by changes in the observed catch. Gear restrictions which can allow catch to be more selective, however, are not accounted for and could explain some of the benefits here attributed to MPAs if these restrictions were implemented simultaneously. There is also an error-in-variables problem, since the measure used of protected area may not correspond to the amount of area that is effectively protected, and this will downward bias our results. A great number of additional factors are missing from this model: environmental forcings, species interaction, and other anthropogenic effects. However, these are unlikely to correlate either strongly or positively with MPA designation.

To investigate the potential for this estimate to falsely identify other changes in man-

agement, we perform a series of experiments. Each experiment simulates a stock under growth and harvesting model. Initially, each stock is fished at an unsustainable level, for 20 years. In year 20, optimal catch management is implemented: first, all fishing is closed to allow the stock to recover; once the stock is within 10% of MSY, the sustainable harest level is applied.

For each experiment, we estimate four statistical models that include effects from MPAs. The true effect of the MPA under these simulations is 0, while estimates other than 0 are a reflection of conflating management effects with MPA effects. The estimated models either have a step increase of 1% in MPA coverage starting in year 20, with and without time trends (**Step, Time Trend** and **Step, No Time**), or a linear increase in MPA coverage starting in year 20 (**Increasing, Time Trend** and **Increasing, No Time**).

The six experiments are describe below:

Logistic (no error): A logistic growth function is used, with to confirm that the estimate on the MPA change is zero.

Logistic (10% error): As above, but with 10% model error on the simulation.

Age-structured (no error): An age-structured model from Branch (2009).

Age-structured (10% error): As above, with 10% model error.

Age + Gear (no error): As above, but including also a gear restruction which ensures that no fish under 4 years of age, which do not yet have high fecundity, are caught.

Age + Gear (10% error): As above, with 10% model error.

The range of the MPA coefficient under simulations with model error is estimated from 1000 Monte Carlo runs. The results are shown in figure 6.8, and show insignificant and negative effects for all MPA coefficients. This supports the claim that the MPA estimate is not picking up other management, even if it is correlated with the implementation of other management practices.
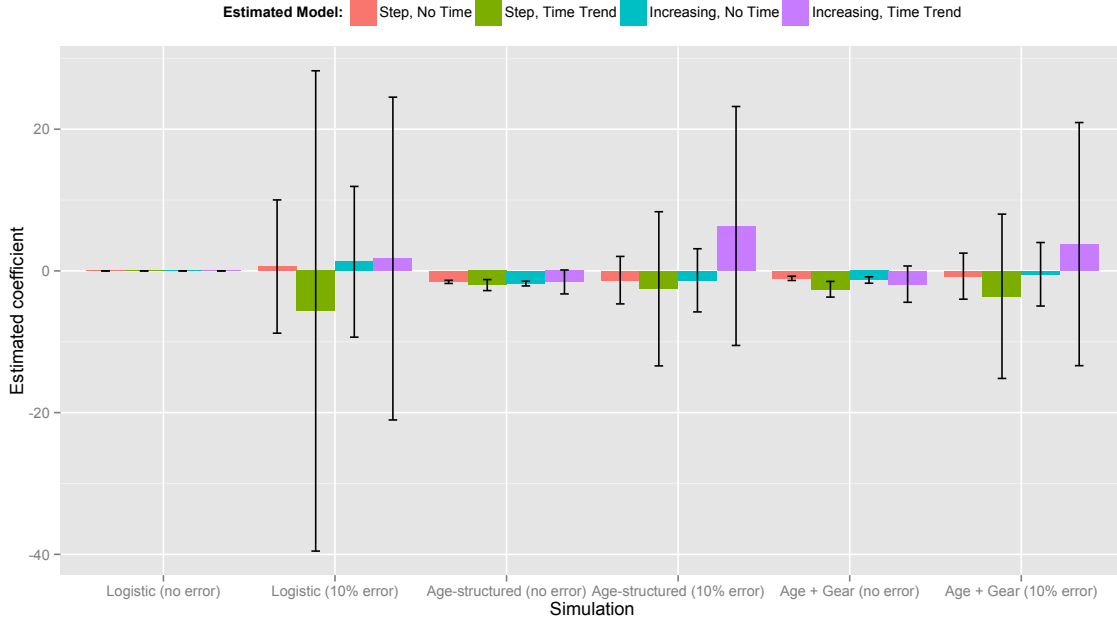
Figure 6.8: Estimates of MPA effect for null models. For each simulation described in the text, and under each model, the estimated effect of MPAs in a model with no effect from MPAs is 0 or negative.

6.4 Regression tree analysis

To determine what factors most influence the effectiveness of MPAs, we estimate the MPA effect for each assessment individually, and build a regression tree against a wide variety of assessment characteristics, shown in table 6.3. These include both region and fish features. The assessment-specific values for β use the multiple years available for each assessment, with a linear time trend.[5]

Regression trees identify the natural divisions that associate input and output parameters (Breiman *et al.*, 1984). Each decision branch of the tree is a division in the data or a subset of the data, and is characterized by an average coefficient estimate and a number of observations. Branches are chosen sequentially, based on whichever parameter and

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[5]The regression model for each stock is

$$R_t = \left(\gamma + \beta \frac{MPA_t}{Area} + \zeta_1 T_{t-1} + \zeta_2 T_{t-2}\right) S_{t-1} + \kappa S_{t-1}^2 + \delta + \theta t + \epsilon_t$$

split produces the greatest reduction in the sum of squared residuals. The highest-level node, representing the average β coefficient across all data when estimated independently, is negative.

Predictors of assessment MPA effect coefficients
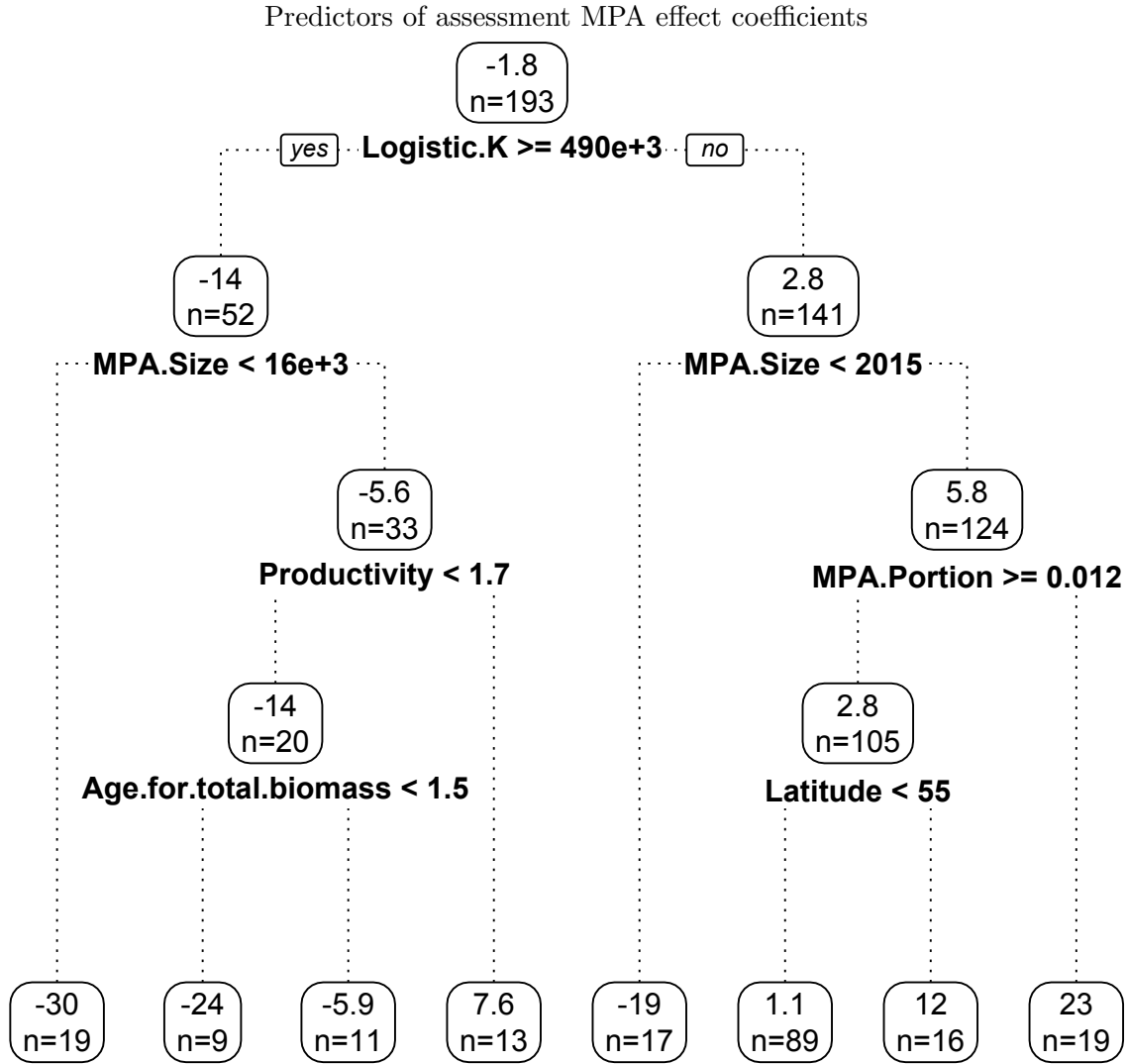
Figure 6.9: A regression tree to explain stock-specific estimates of the benefits of MPAs, according to the logistic model. Each box shows the mean value of the β coefficient for all assessments matching the given constraints, and the number of such assessments. The variables are defined in table 6.3.

The first branch of the regression tree is determined by the potential size of the stock, which also relates to the size of the assessment region. Large stocks (left branch) typically are not benefited by protected areas unless the protected areas are large and the stock

Parameter	Description	Notes
Region.Size	Assessed region size	in km^2
MPA.Size	Protected area size	in km^2
MPA.Portion	Portion of assessed region protected	
Latitude, Longitude	Latitude and Longitude	of the assessment region centroid
Productivity	Productivity	as the maximum recorded catch, divided by the region size
Logistic.r	Logistic growth rate (r)	estimated
Logistic.K	Logistic carrying capacity (K)	estimated
Country	The country, or multinational	
REC.AGE.yr,	Age of recruits and recruited biomass	in yr, as used for estimating stocks
Age.for.total.biomass		(by individuals, N = 210; by biomass, N = 182)
MSY.MT	Maximum sustainable yield	in MT (N = 193)
F.AGE.yr.yr	Age for computing mortality	in yr (N = 157)
Habitat.Habitat	Habitat	pelagic; demersal; diadromous; wholly freshwater (N = 142)
SSB.SEX.sex	Sex of spawners	for estimating stocks (N = 129)
SSB.AGE.yr	Minimum age of spawners	in yr (N = 128)
M.1dT, M.1dyr	Natural mortality	per year (N = 126)
A50.yr	Age at 50% maturity	in yr (N = 122)

Table 6.3: The variables used to in the regression tree construction to predict effect sizes, as estimated by the logistic equation.

is very productive. For smaller stocks with large MPA areas, all branches show positive effects from MPAs, with the largest effects from large MPAs within even larger assessment regions. High latitude stocks also show large benefits from MPAs, probably driven successful management in Alaska and the North Sea.

A few missing parameters are notable. In particular, the country is not a top-level predictor, although latitude may be a proxy for it. Age only shows up on the lower left branch (age for estimating biomass), suggesting that both long- and short-lived species benefit. Finally, habitat type is not a top-level predictor.

6.5 Economic analysis

One of the clearest impacts of successful fisheries management is on fisher income, and MPAs may be an effective method for boosting incomes (Worm *et al.*, 2009). In principle, this expected surplus can be compared to the cost of protecting an area, to determine if the benefits of protection outweigh the costs.

In the following, we treat the estimated point values from the model including trends and year fixed effects in section 6.3 as valid across all countries. This is a poor approximation, and the following results should be taken as only suggestive of actual country benefits.

To translate our estimate of effects into economic benefits, we use the Sea Around Us landed catch values. The additional value attributable to protecting a given area of ocean is,

$$V(MPA) = \sum_k p_k \beta \frac{MPA}{EEZ} \bar{S}_k$$

where $\bar{S}_k$ is the steady-state stock for species k, p_k is the landing price per MT for that species, and EEZ is the exclusive economic zone area for the given country. The steady-state stock is unavailable for most species, so we use recorded catch as a lower bound on the stock and produce a conservative estimate of these benefits.

Balmford *et al.* (2004) uses a survey of the costs of 83 globally representative marine reserves to estimate an empirical model of the reserve maintenance. They include both a simple model, based only on the reserve size, with an R^2 of 0.79, as well as a model that includes more predictors and achieves an R^2 of 0.90. Both models show economies of scale, where larger MPAs cost less per unit area than smaller ones.

This relationship describes the added value of marine protection in terms of potential landings. This is not equivalent to fisher incomes, since we have no information on effort or fisher costs. It is furthermore an underestimate of the value of these landings to society as a whole.

To estimate the threshold at which benefits exceed costs, we apply the simple model from Balmford *et al.*:

$$\log_{10} \frac{cost}{MPA} = 5.02 - 0.8 \log_{10}(MPA)$$

where MPA is measured in km^2 and *cost* is measured in year 2000 U.S. dollars. The landed value of catch from Sea Around Us, $p_k \bar{C}_k$ is also in year 2000 dollars, and the average of catch is taken over the years 1997 - 2006.

We combine these to produce a profit function, and find its root, which is the point at which estimated economic benefits exceed estimated costs. Of 234 countries and large marine regions, 71 have enough area protected to have a net economic benefit, 141 currently

have a net loss, and 22 generate too little rent for any amount of protected area to generate surplus sufficient for an economic gain. The mean portion of a country's EEZ that needs to be protected to generate an economic surplus is 8.5%, while the current protected region, averaged by country as a portion of EEZs, is 2%. A table of these countries and their key parameters is listed in appendix F.11.

This estimate assumes that a single MPA is created for each country. However, most countries have a large number of small MPAs. Balmford *et al.* provide an refinement of the expression above that accounts for distance to a populated area (DP) and purchasing power parity (PPP), which we use to calculate a total cost for the actual MPA configuration of a country,

$$\log_{10} \frac{cost}{MPA} = 5.62 - 0.72 \log_{10}(MPA) - 0.002DP - 0.3PPP$$

where DP is measured in km.

As an example, compare Alaska and Australia, two large and productive regions (see table 6.4). Both have a large share of their EEZs under protection, and because of the high value of their marine products, this amount is far greater than the breakeven portion for which potential benefits exceed costs. Since the average landed value in Australia is 40% of the landed value in Alaska, the breakeven point for Australia is correspondingly larger, although it is still less than .1% of its total EEZ. The economic benefit in Australia is estimated to be less than Alaska's, despite the larger protected area. In addition, the much larger number of MPAs in Australia (356 compared to 42) causes its total estimated cost of maintaining them to be higher. With both effects, Alaska is estimated to gain three times its costs, while Australia's MPAs cost three times their benefit. All of these effects, however, are much smaller than the economic benefit to Australia of just the tourism of one protected region, the Great Barrier Reef Marine park, estimated at $5.7 billion in 2012 (Deloitte Access Economics, 2013).

Country	EEZ (km^2)	Landed Val. ($m)	Breakeven	Current	Cost ($m)	Benefit ($m)
Alaska (USA)	3 770 021	1398	0.000 26	0.059 39	26	94
Australia	6 362 934	544	0.000 97	0.082 19	160	51

Table 6.4: Estimated cost and benefit components for Australia and Alaska.
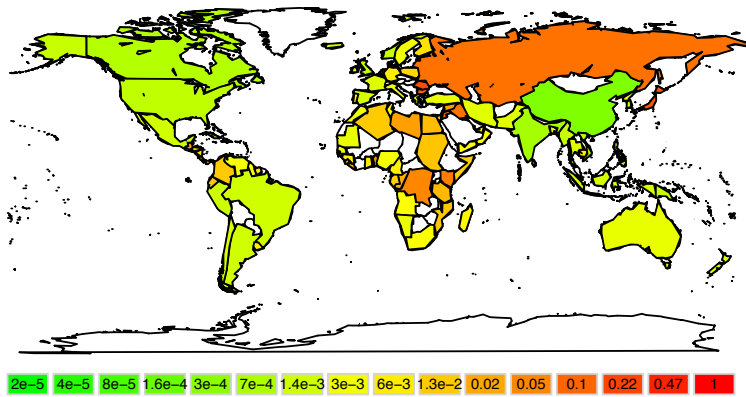
(a) Portion of a country's EEZ required for benefits to exceed costs: green countries require less than .1%; yellow countries require less than 1%; and orange countries require less than 10%.
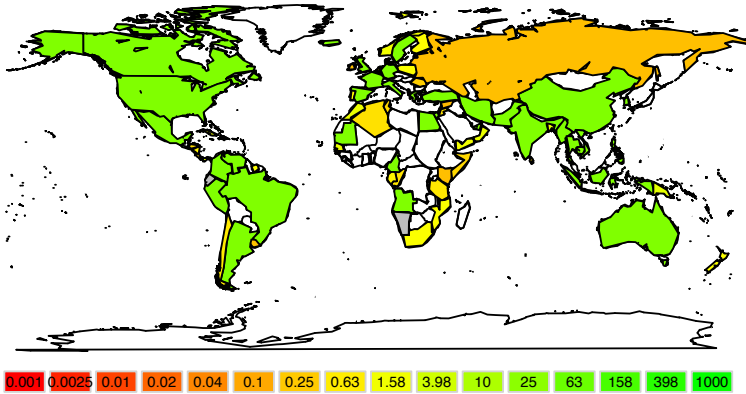
(b) Current status of total MPA area, relative to the breakeven point: green denotes countries which have more than the necessary share of protection; orange and red for those below that threshold.
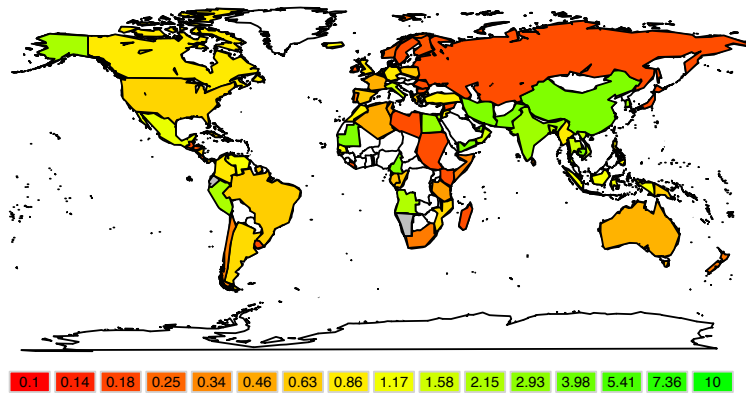
(c) Current status of MPA approximated costs and benefits: green denotes countries for which economic benefits exceed costs; orange and red for those below that threshold.

Figure 6.10

Globally, the estimated costs of all MPAs, under Balmford's assumptions, is 1.45 billion USD per year. Using the assumptions above, the additional landed value attributable to MPAs is 2.18 billion USD per year. This estimate is conservative, since catch is used as a lower bound on the stock. The relationship between catch and stock is widely variable (see appendix F.11.1), but the average ratio in our dataset since 2000 is 0.18. Using this to estimate stocks globally, the added value is 12 billion USD per year. This suggests that out of a total landed value within EEZs of 51 billion USD in 2005, existing MPAs may be responsible for approximately 23% of catch (see table 6.5).

Regoin	Costs ($m)	Cons. Value ($m)	Scaled Value ($m)	Landed ($m)
North America	434	1118 (13%)	6212 (75%)	8258 (16%)
South America	64	60 (1%)	335 (7%)	4261 (8%)
Europe	503	466 (6%)	2589 (38%)	6713 (13%)
Africa	41	99 (3%)	554 (18%)	2989 (5%)
Asia	174	281 (1%)	1563 (6%)	24071 (46%)
Oceania	226	143 (3%)	799 (17%)	4485 (8%)
Open Ocean	6	12 (2%)	66 (12%)	516 (1%)
Total	1451	2181 (4%)	12121 (23%)	51297 (100%)

Table 6.5: Regional costs and added value for nationally designated MPAs. Costs are estimated using the three-term expression from Balmford *et al.* (2004). Conservative (Cons.) values use catch as a proxy for stock; Scaled values are conservative values scaled by the estimated stock-to-catch ratio. Percents for each are the percent of the total landed value (Landed). Percents for the landed value are as a fraction of the total global landed value from national EEZs.

6.6 Nonlinearity in responses

To explore the effects of non-linearity in the strength of the response, we add quadratic terms and terms that allows the effect to vary with stock-size. The derivation of this model is described in multiple steps.

By including the predictor $\left(\frac{MPA_i}{Area_i}\right)^2$, we find that the marginal benefit of MPAs decreases with total protected area (see appendix F.10), but the decrease is slight ($< 2\%$) for

	Timeless	Stock Trends	Year FE	Trends and Year
# MPA S_t	-0.0007	-0.0027^{**}	-0.0006	-0.0028^{**}
	(0.0004)	(0.0010)	(0.0004)	(0.0010)
Sum MPA^2 / Area$^2 S_t$	16.6424^{***}	18.5588^{***}	16.6583^{***}	18.1516^{***}
	(3.6807)	(3.2159)	(3.7054)	(3.2042)
R^2	0.6072	0.6422	0.6099	0.6448
Adj. R^2	0.5425	0.5691	0.5428	0.5695
Num. obs.	10093	10093	10093	10093

$^{***}p < 0.001$, $^{**}p < 0.01$, $^{*}p < 0.05$

Table 6.6: Estimate of the nonlinear effects of individual MPA areas, represented as coefficients on the number and sum of squared areas. Additional MPAs have a negative effect, unless they have a large enough area. This break-even point is 1.2% for the Trends and Year model.

most of the observations.

However, the benefit is increasing in the size of individual MPAs. Let the effect of an individual MPA with area MPA_j within an assessment of area $Area_i$ be $\beta_0 + \beta_1 \frac{MPA_j}{Area_i} + \beta_2 \frac{MPA_j^2}{Area_i^2}$. Then the effect of a collection of MPAs is $\beta_0 N + \beta_1 \frac{\sum MPA_j}{Area_i} + \beta_2 \frac{\sum MPA_j^2}{Area_i^2}$. First, we find that $\frac{\sum MPA_j}{Area_i}$ and $\frac{\sum MPA_j^2}{Area_i^2}$ are significantly colinear (see appendix F.10.1), but that the sum of square areas provides a closer fit to data, so we drop the linear term.

Fitting the model with these changes shows that the largest MPAs produce the largest benefits, as also supported by the regression tree. Small MPAs do not produce a clear benefit. In fact, MPAs less than 1.25% of an assessed area are estimated as having a negative effect, although the significance of that result is low. This is at odds with Halpern (2003), who suggests that relative impacts are independent of reserve size.[6]

We further add a relative stock-size effect. Whereas the previous coefficients of interest above are interacted with current stock size to estimate growth rates, we now add cor-

---

[6]Halpern (2003) studies the effects of protection on fish density and biomass and other benefits within reserves and finds these benefits are constant per unit area. The assumptions of this paper differ from those of Halpern both in that we consider a separate form for the impact, and study that impact for populations adjacent to reserves.

responding coefficients that are not interacted with the stock size, but are scaled by the observed mean stock. These terms boost the stock surplus above its previous predicted growth rate when the stock is low, and depress it when it is high.

This forms our final specification:

$$R_{it} = \left(\gamma_i + \beta_0 MPN_{it} + \beta_2 \frac{\sum_j MPA_{ijt}^2}{Area_i^2} + \zeta_{1,i}T_{i,t-1} + \zeta_{2,i}T_{i,t-2} \right) S_{i,t-1} +$$
$$\kappa_i S_{i,t-1}^2 + \alpha_0 MPN_{it}\bar{S}_i + \alpha_2 \frac{\sum_j MPA_{ijt}^2}{Area_i^2}\bar{S}_i + \theta_i t + \phi_t + \delta_i + \epsilon_{it}$$

where MPN_{it} is the number of MPAs in assessment region i in year t and $\bar{S}_i$ is the average stock level over the entire assessment period.

While the carrying capacity, K_i, might be a more natural choices to use in place of $\bar{S}_i$ to scale relative effects, this specification better incorporates directly observed values. In addition, $\bar{S}_i$ and the estimate of $K_i = \frac{\gamma_i}{\kappa_i}$ are closely correlated (95% CI 0.943 to 0.964).

The coefficients from this model are cleanly estimated, but their impacts are more difficult to interpret. For both the number of MPAs and their squared sizes, the coefficients on the current stock level are of similar magnitude to the coefficients on the average stock, but of opposite signs. When the stock is at its mean level, the combined coefficient on MPA number is negative, implying that more small MPAs depress growth. At the same time, the coefficient on squared area is positive, so that a large enough protected area will counterbalance this effect.

The benefits of MPAs, according to this model, are also stock-size dependent. Depressed stocks receive a larger benefit, even producing surplus increase for all MPA sizes when stocks fall below 76% of the historical biomass. One consequence of this stock-dependency is that there exist qualitatively different dynamics in different regions of the MPA state space, as shown in figure 6.11.

As before, the coefficients for marine-only areas are less precisely estimated. As expected, the marginal effect of a squared-MPA-portion at the average stock size ($S_t = \bar{S}$) is greater for marine areas than for total areas (20 compared to 16). In the discussion that follows, we will use the coefficients estimated for MPA total area.

Two parameters provide a clearer interpretation of this model's results, which we call

	Timeless	Stock Trends	Year FE	Trends and Year
# MPA S_t	−0.0024*	−0.0039***	−0.0025*	−0.0040***
	(0.0010)	(0.0008)	(0.0011)	(0.0008)
Sum MPA^2 / Area$^2 S_t$	−33.0278**	−35.8660***	−34.5726**	−37.5532***
	(10.9829)	(7.9614)	(11.7827)	(9.1776)
# MPA $\bar{S}$	0.0016**	0.0031**	0.0016**	0.0030**
	(0.0006)	(0.0010)	(0.0006)	(0.0010)
Sum MPA^2 / Area$^2 \bar{S}$	48.5862***	52.6993***	50.0724***	54.0134***
	(7.4335)	(7.9802)	(8.1518)	(9.1374)
R^2	0.6101	0.6450	0.6130	0.6475
Adj. R^2	0.5458	0.5724	0.5463	0.5727
Num. obs.	10093	10093	10093	10093
# Marine S_t	−0.0419***	−0.0521***	−0.0391***	−0.0492***
	(0.0089)	(0.0113)	(0.0095)	(0.0133)
Sum Marine^2 / Area$^2 S_t$	−29.1042***	−21.3140**	−30.2583***	−22.9769*
	(8.0467)	(7.8458)	(8.8440)	(9.2439)
# Marine $\bar{S}$	0.0045	−0.0133	0.0056	−0.0153
	(0.0066)	(0.0307)	(0.0068)	(0.0319)
Sum Marine^2 / Area$^2 \bar{S}$	43.1634***	42.2305***	43.7379***	43.1593***
	(7.5955)	(8.2189)	(7.9078)	(8.9348)
R^2	0.6560	0.6697	0.6635	0.6772
Adj. R^2	0.5663	0.5626	0.5655	0.5618
Num. obs.	2917	2917	2917	2917

***$p < 0.001$, **$p < 0.01$, *$p < 0.05$

Table 6.7: Results from the nonlinear, stock-dependent model. For both the number of MPAs and the sum of squared areas, values interacted with the current stock and with the mean stock size are of opposite signs, driving a counter-balancing stock-dependent effect.
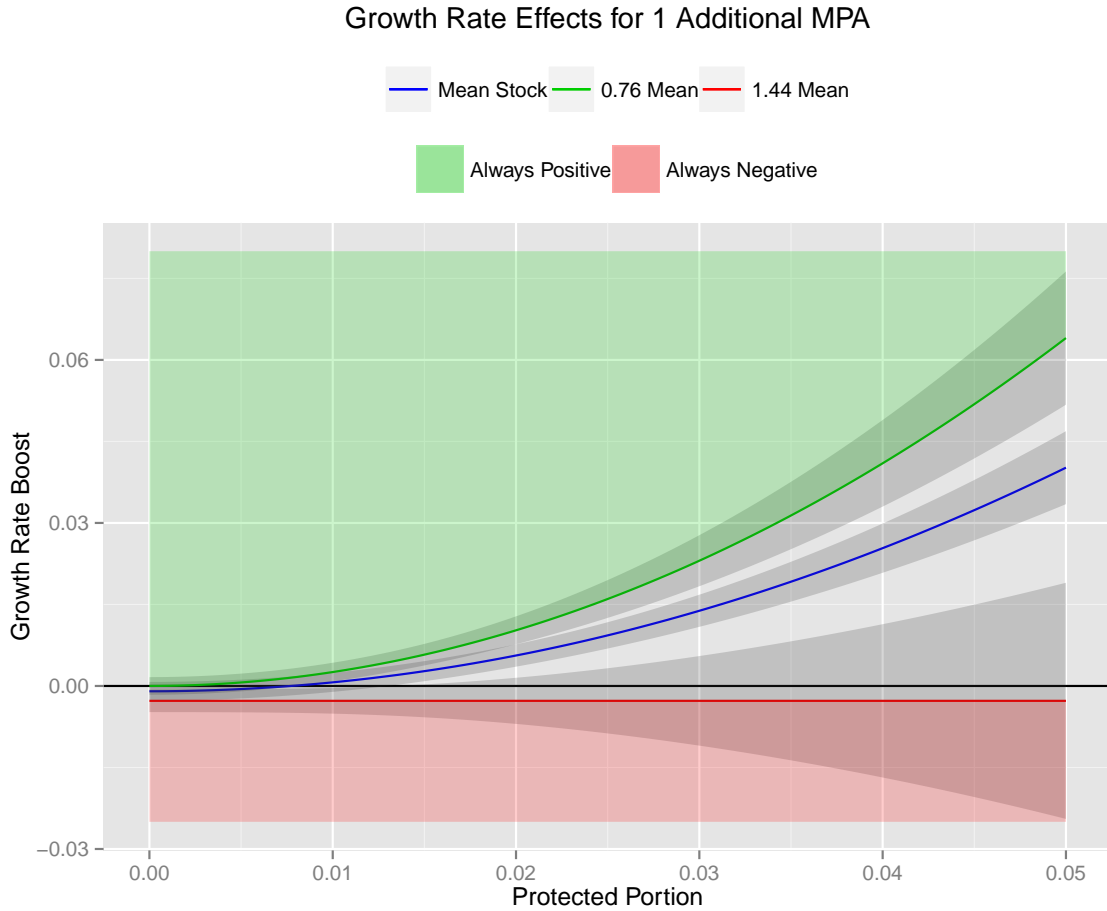
Figure 6.11: Parameter regions for the stock-dependent effects of a single MPA. Stocks at the mean stock level (blue) experience a small decrease in growth rates up to a protected area portion of .8%. However, stocks less than 76% of their mean biomass always experience additional growth, and stocks greater than 144% of the mean biomass always experience depressed growth.

the surplus parameter and the stability parameter. First, we define,

$$A = \alpha_0 MPN + \alpha_2 \frac{\sum_j MPA_j^2}{Area^2}$$

$$B = \beta_0 MPN + \beta_2 \frac{\sum_j MPA_j^2}{Area^2}$$

A is the stock-independent effect of MPAs (the coefficients for $\bar{S}_i$), as a function of MPA distribution, and B is the stock-dependent effect (the coefficients for $S_{i,t-1}$. The value $A + B$ is the surplus parameter, and estimates the steady-state growth boost from protection for any given combination of MPAs, when $S_{i,t} = \bar{S}_i$. The value $2A + B$ is a summary statistic that captures the added stability provided to the region by its protected areas. The derivation of this "stability" parameter will be described in section 6.6.3. Figure 6.12 provides a comparison of these summary parameters for MPA total and marine areas.
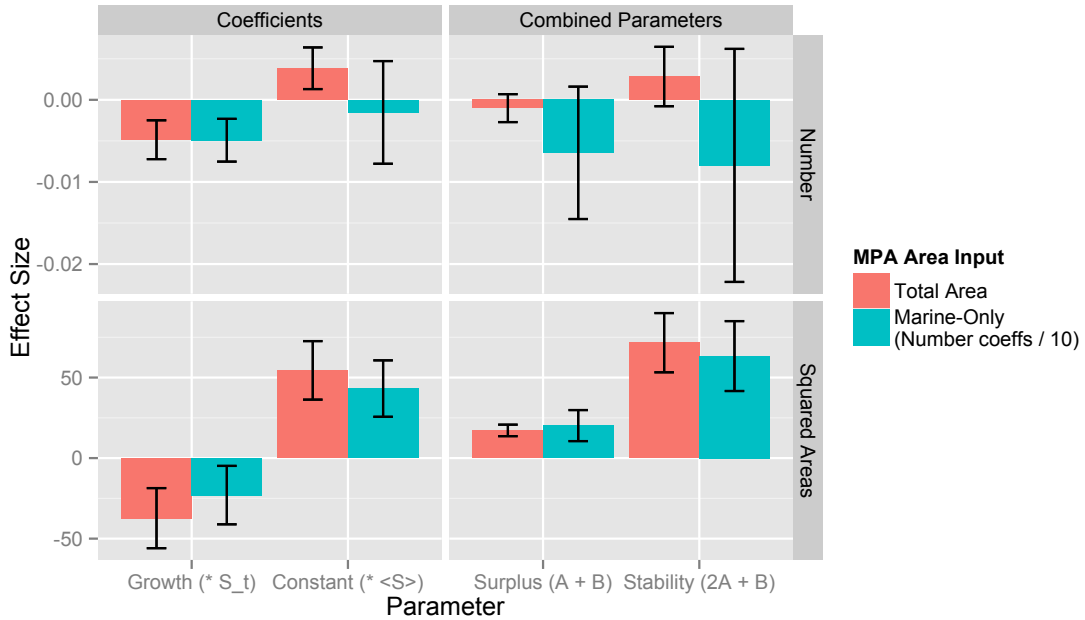
Figure 6.12: Coefficients and combined parameters from the model with both year fixed-effects and stock-specific trends. The combined parameters represent the surplus boosting and stabilizing effects of protection. While the effects of the number of MPAs are insignificantly estimated, the size of MPAs shows a positive value for both of these effects. Bars shown for marine-only number of MPAs are divided by 10 for clarity. Error bars show 95% confidence intervals.

For another perspective on these effects, we consider the surplus as a fraction of the mean stock size, $A + B\frac{S}{\bar{S}}$. Figure 6.13 shows this effect across stock sizes (as a fraction of the mean stock size) and the portion protected. For a single MPA, the benefit is largest for large protected areas and small stocks. Beyond a single MPA, the distribution of MPA sizes has a large effect. For equally sized MPAs, as the number of MPAs increases, the contours smooth, until they are only a function of stock size. For "extreme" sized MPAs, where all but one MPA covers zero area, the threshold for a positive benefit at the mean stock size slowly extends toward a larger area protected as the number of MPAs increase.[7] In both cases, the size of the effect for small protected area portions increases with the number of MPAs.

6.6.1 MSY changes

Another way to study this change is to consider the growth curve, normalized by the carrying capacity. Let $z = \frac{s}{K}$ and the mean stock be denoted $\bar{z}$. Then, dropping controls and the minimum viable population intercept ($\zeta_1, T_{i,t-1}S_{i,t-1}$, $\zeta_{2,i}T_{i,t-2}S_{i,t-1}$, $\theta_i t$, ϕ_t, and δ_i) and letting $r = .5$, we define $g(s) = \frac{R}{K}$ as,

$$g(z) = \frac{1}{2}z(1 - z) + \beta_0 MPNz + \beta_2 \frac{\sum MPA_j^2}{Area_i^2}z + \alpha_0 MPN\bar{z} + \alpha_2 \frac{\sum MPA_j^2}{Area_i^2}\bar{z}$$

Three example growth curves are shown in figure 6.14. Large amounts of protection ($> 2\%$) produce large increases in the growth rate across a large span of the growth curve. A greater number of small MPAs can produce as large of an increase in growth, when stock-sizes are low, but depresses growth for large stock sizes.[8]

A useful statistic for comparing these adjusted growth curves is the maximum sustainable yield (MSY). The stock at the MSY is,

$$z_{MSY} = \frac{S_{MSY}}{K} = \frac{1}{2} + \beta_0 MPN + \beta_2 \frac{\sum MPA_j^2}{Area_i^2}$$

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[7]In theory, the effect of MPAs with zero size should be zero. However, very spall protected areas can produce large effects, and are often placed in highly productive regions. We treat this effect as an empirical approximation, accounting for non-linearities that allow very small MPAs to produce non-zero effects.

[8]We have developed an online tool to graphically apply these estimated effects to any parameter values: `http://existencia.org/mpa/`
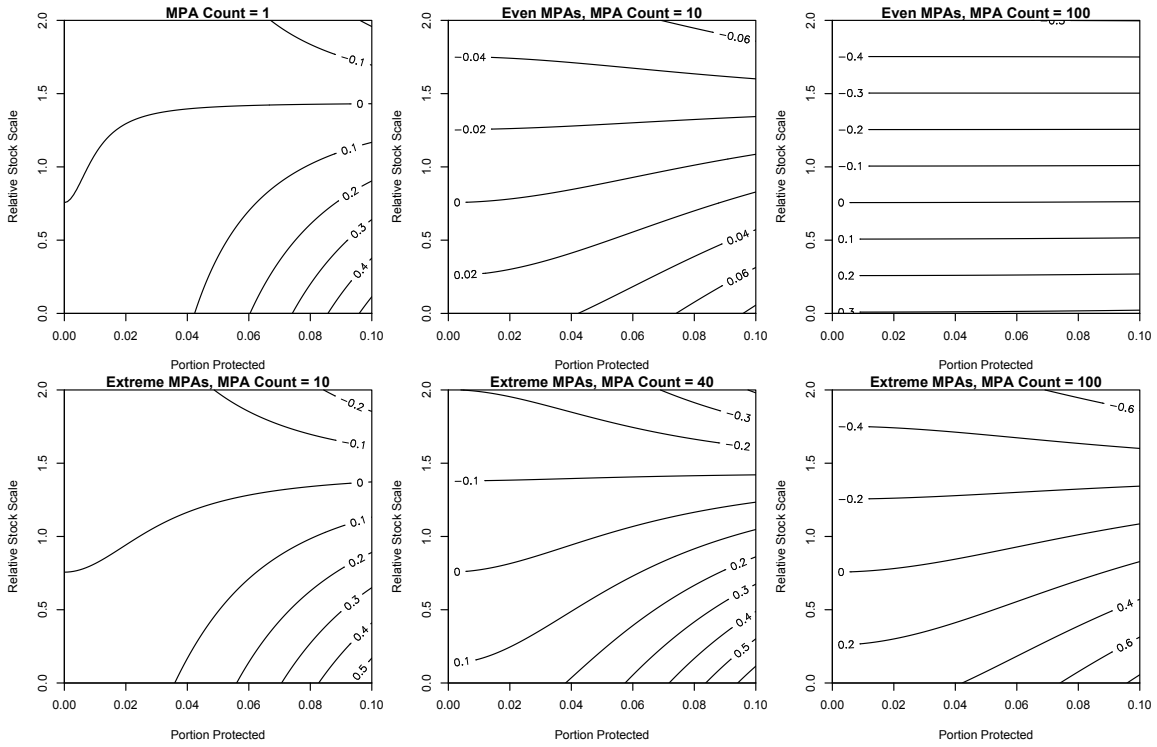
Figure 6.13: The surplus boost (additional stock surplus) as a fraction of the mean stock size. The x-axis represents total area protected; the y-axis is the stock size as a fraction of the mean stock size. The plots are divided into two groups. Along the top row, the MPAs are assumed to be identically sized, so that each MPA has a relative size of p/N. Along the bottom, every MPA except one has a size of 0, and one has a size of p.
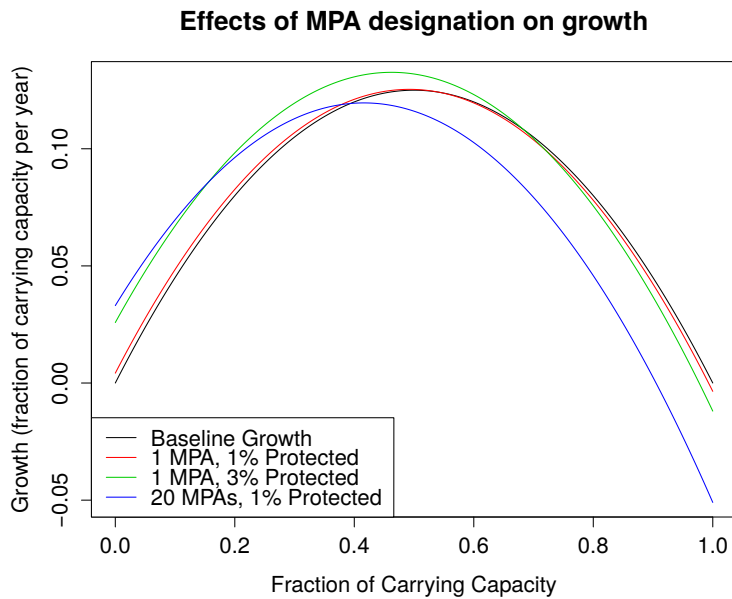
Figure 6.14: The baseline (zero protection) growth curve, normalized by the carrying capacity, for the baseline growth rate $r = 0.5$. A single MPA protecting 1% of the assessed area makes very little difference, while one protecting 3% of the area produces a large effect. Similarly, 20 MPAs results in large decreases in the growth rate for large stocks, but boosts growth rates for small stocks as much as protecting 3% of the region.

Substituting this expression into the growth expression above gives a relative measure of the MSY growth. This is shown across a range of values for the "Trends and Year" model in figure 6.15. The ratio of the MSY growth under existing protection, compared to MSY growth with zero protected areas, is also shown. For low total amounts of protected area, the driving factor is the number of MPAs, with MSYs increasing when more than 60 MPAs were designated. MSYs also increase for a single MPA when the size is greater than 0.7%, and for all MPA counts when $\frac{\sum MPA_j^2}{Area_i^2} > 0.0006$, reachable when at least 2.4% of an area is protected.
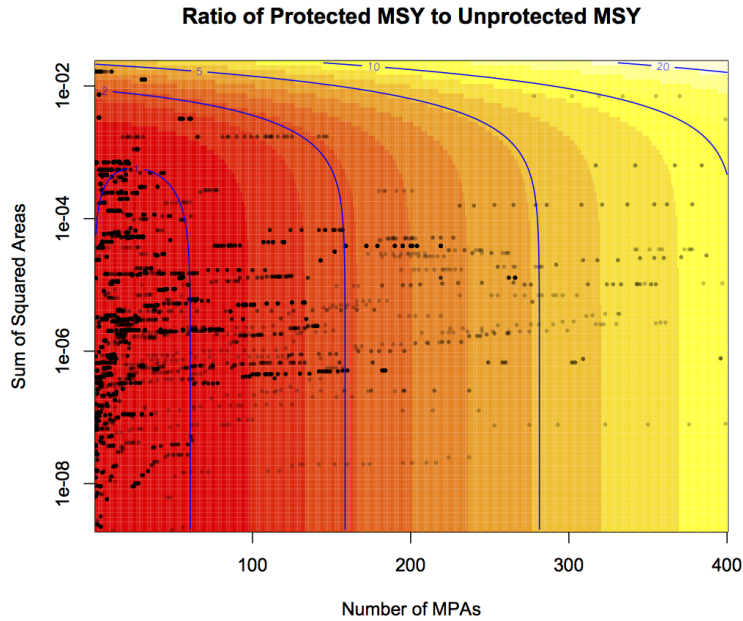
Figure 6.15: Ratios of MSY growth (as a fraction of the carrying capacity, K) accounting for protection, divided by the MSY that would be achieved if there were no protection, under the estimated model "Trends and Year" model. Dots show observations (which also extend outside of graph range), using partial transparency to show the number of years occupied by each assessment. The majority of the data are represent fewer than 50 MPAs and 1e-3 for the sum of squared area, however the entire range of impacts is represented in the data (with less density for large numbers of MPAs representing both high and low squared areas).

6.6.2 Localization of effects

The effects from MPAs are naturally local, with the influence generally expected to extend less than 10 km to 100 km from a reserve boundary (Palumbi, 2004). It is commonly claimed that protected areas only benefit fish that spend time within them. We explore to what extent MPAs produce localized effects.

First, as a counterpoint we aggregate the effects of MPAs at a country-wide level, using country-wide stock surplus, as described in appendix F.3. Pooled to this level, the effects of MPAs appear to be negative: the growth of MPAs has coincided with a decrease in the growth capacity of stocks. This result is not surprising, given the long-term degradation of fishery ecosystems that have occurred over this period.

We can also better limit the MPAs within assessment regions. Assessment regions tend to represent the convex hull around fish populations, describing only a rough approximation to the spatial population distribution. The AquaMaps database (Kaschner *et al.*, 2007) provides a finer resolution approximation to the location of fish populations. We estimate the non-linear, stock-dependent model including both the MPAs interacted with the fish distributions and the "remaining" MPA areas and counts. The results are shown in figure 6.16. Additional details on this analysis is described in appendix F.7.

The estimates for the effect of the number of MPAs within the overlapping area are similar to the non-linear model estimated without AquaMaps intersections. The effect of those outside the regions cannot be estimated. The magnitudes for the effect for overlapping MPA sizes are a factor of 100 times greater, and the signs are reversed. The value of the growth coefficient $(A + B)$, however, is similar but 100 times greater. The key result from this analysis, however, is that the estimated effect for remaining MPA sizes is very closely estimated to be zero. MPAs that do not intersect with a species have no effect on them.

6.6.3 Stabilization effect

Relative to their effects at the mean stock size, MPAs further increase growth when the stock is low, and depress growth when it is large. This results in a stabilization of the stock. This result is the converse of evidence that exploitation increases variability in fish stocks (Anderson *et al.*, 2008).
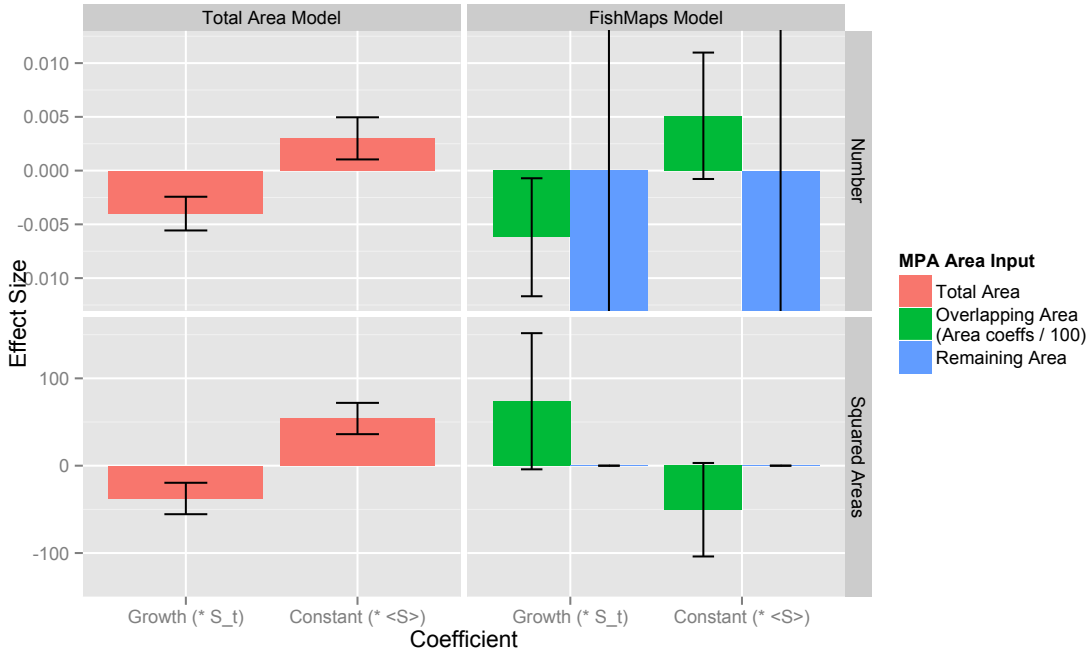
Figure 6.16: A comparison between the original model (left) and an interaction with the FishMaps distributions (right).

To identify this analytically, we complete a simple Gordon-Schaefer bioeconomic model by adding effort-based catch to the expression above (Gulland, 1983):

$$S_{t+1} = S_t + rS_t(1 - S_t/K) + BS_t + A\bar{S} - ES_t + \epsilon'_\Delta$$

$$z_{t+1} = z_t + rz_t(1 - z_t) + Bz_t + A\bar{z} - Ez_t + \epsilon_\Delta$$

where $B = \beta_0 MPN + \beta_2 \frac{\sum MPA_j^2}{Area_i^2}$ and $A = \alpha_0 MPN + \alpha_2 \frac{\sum MPA_j^2}{Area_i^2}$; E is a measure of both fishing effort and catchability, so that catch $C_t = ES_t$; and ϵ_Δ is a normally-distributed error term. The top line is in terms of stocks (S_t), while the bottom uses normalized units, $z_t = S_t/K$. The effort expression allows the bioeconomic system to have a stable point at some point (which will, by virtue of its stability, be $\bar{z}$).

We choose $\bar{z}$ to be the maximum economic yield (MEY) under a given discount rate ρ[9]:

$$\bar{z} = \frac{r - \rho + B}{2\rho}$$

The level of effort, E, to maintain the stock at $\bar{z}$ is,

$$E = r(1 - \bar{z}) + A + B$$

We define ϵ^2 as the total variance across several periods,

$$\mathbb{E}(z_t - \bar{z})^2 = \epsilon^2$$

Finally, to study the amount of variability as produced by this process, we assume stationarity of the variance:

$$\mathbb{E}(z_{t+1}(z_t + \epsilon) - \bar{z})^2 = \epsilon^2$$

From this, we can derive an expression for the long-run variance as a function of the other parameters. It is,

$$\epsilon^2 = \frac{4\epsilon_\Delta^2}{4 - (2 - r + \rho - 2A - B)^2}$$

These expressions are only defined where $0 < r - \rho + 2A + B < 2$, and ϵ^2 decreases as $2A + B$ increases. Therefore, $2A + B$ is a summary statistic the determines the effect that MPAs have on stability.

6.7 Conclusions

We show that, on average, MPAs have clear benefits for fisheries. They drive increases in stock surpluses through growth rates. The effects are local, but captured by assessment regions. Large MPAs are have the largest effects, and since they also have the smallest per-area costs, provide the best opportunity for profits to exceed costs. Finally, MPAs provide a stabilization effect which even small MPAs support. Our results correspond roughly to comparable case studies of the effects of marine protection on catch per unit effort in adjacent fisheries (see figure 6.18).

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[9]Maximum economic yield (MEY) is identified by treating the fishery as a production mechanism which is a function of stock size: $R = f(S)$, as with MSY. MEY is the level of yield at which the $f'(S) = \rho$, the economic discount rate. If ρ corresponds to the externally available interest rate, then this is the highest level of "investment" in S which provides a return at least as large as ρ.
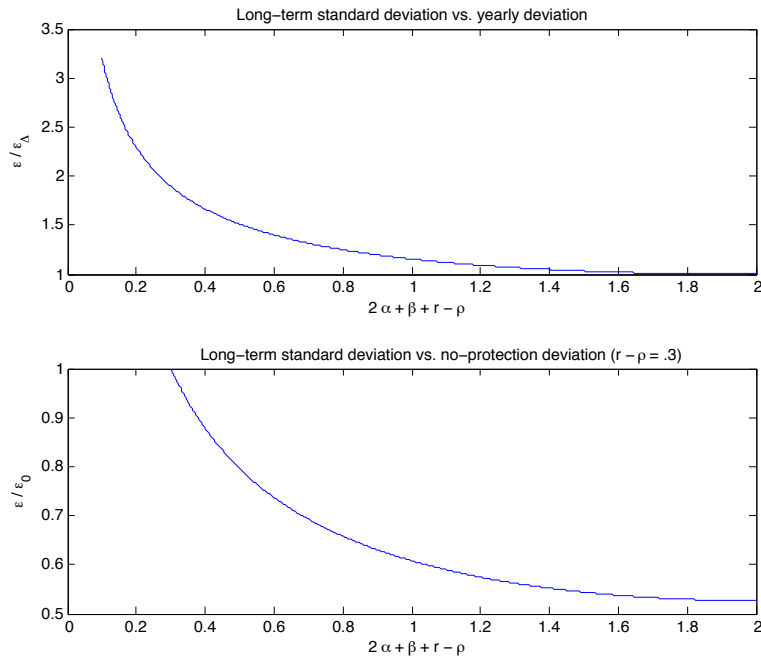
Figure 6.17: The addition of protected areas decreases the variability of stocks. The top panel shows the total variability of a bioeconomic model of a stock, shown as its total standard deviation, relative to its yearly variation. As the protected area terms, $2A + B$, increase, this variability decreases, toward having no added variability. The bottom panel sets $r - \rho$, the growth rate minus the discount rate, at .3 and displays the variability of the stock with protected areas, relative to the same stock without protections. In this case, the variability can shrink to half of its unprotected value.
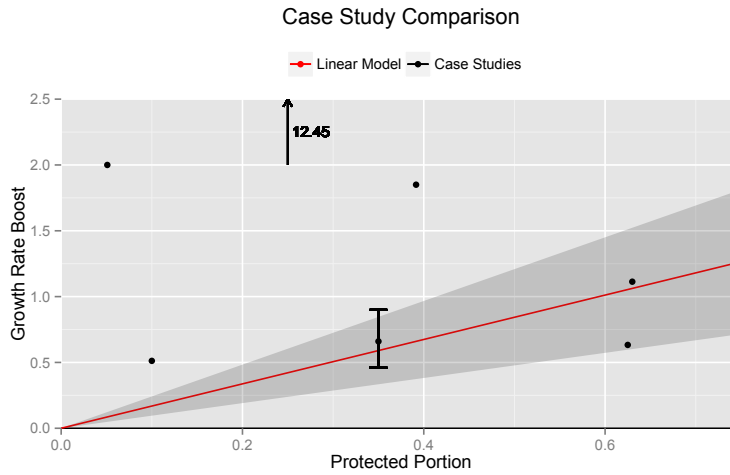
Figure 6.18: Comparison between the linear model and case studies described in appendix F.6. Dots and line ranges show the estimated changes in catch-per-unit-effort (CPUE) that result from designation of neighboring marine reserves. The linear model is shown with 95% confidence intervals. The spread between case study results is very large, but appears roughly comparable to the model results.

There are a number of caveats to the results presented here. We are estimating an average effect across widely varying, complex, social-ecological systems. We apply a reduced-form, ex post approach, which black-boxes a wide range of relevant dynamics and uncertainties. For example, while shifts in effort from marine protection are relevant to management, we look only at the ecological production outcomes. While a better understanding of the global averages can inform outcomes, individual estimates can vary greatly. The logistic model used here is also very simple. It does not include age cohorts or ecosystem interactions, and environmental impacts are limited to the linear effects of temperature. Analyses that incorporate more detailed contextual information, such as vessel-level effort data, gear types, and concurrent management practices, can provide better internal validity (e.g., Smith *et al.*, 2006).

It is likely that less than a third of MPAs are effectively managed and protected (Kelleher, 1996, Pauly *et al.*, 2002). This suggests that our results are conservative since well-protected MPAs are expected to have greater benefits. However, the RAM database is also biased toward developed countries, due to the dearth of scientific assessments in developing countries. The true global average benefit, and potential average benefit for MPAs may be

outside of the confidence intervals of our estimate.

This analysis provides a new entry point for exploring a wide variety of topics, including the institutional features that support effective MPAs (Charles and Wilson, 2009) and the role of fish mobility (Hilborn *et al.*, 2004) and growth rates (Roberts *et al.*, 2001) on MPA effectiveness. It also informs discussions of trade-offs between the costs and benefits of protection (e.g., Smith and Wilen, 2003). Economic models suggest that marine reserves increase fishing costs and decrease optimal catches (Hannesson, 1998). While our dataset does not include fishing costs, and our results show increases in optimal catch levels, these are estimated mostly for low levels of fishing.

Our results corroborate evidence from Guidetti and Claudet (2010) that catch-per-unit-effort (CPUE) within MPAs can be twice that of surrounding areas. This suggests that MPAs can be amongst the most effective forms of fishery management, and future work needs to explore the conditions that affect the distribution of benefits.

Our work supports the hypothesis that protecting large portions of the ocean can greatly benefit both fish and fishers. We find that the greatest benefit from MPAs comes from the largest MPAs, in expectation. It is not possible to place strong confidence in this result, however, since most of the regions we evaluate only have small protected portions, and evidence from the Great Barrier Reef reserve does not support this hypothesis (Fletcher *et al.*).

We find that for most countries, marine protection easily generates more than it costs. However, this cost comparison is most favorable where MPAs are large and have small per-area costs due to economies of scale.

Finally, this analysis does not address the effect of ITQs or other catch restrictions, since these are implicitly incorporated into our dependent variable. Our results suggest that MPAs provide an benefit that is independent of the effects of ITQs to manage optimal stock and catch levels. Because of this, MPAs and ITQs can work together to support more productive marine resources.

Chapter 7

A Distributed Fishery Model
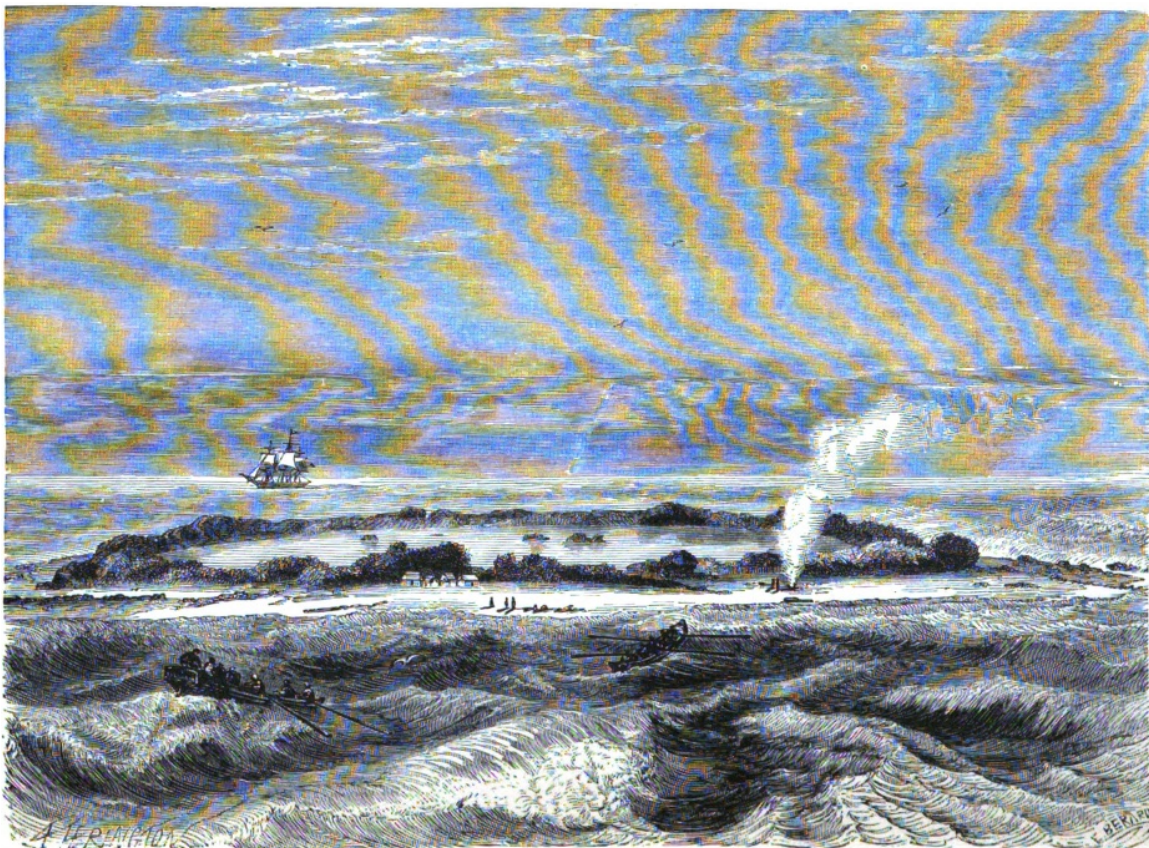
"FAIRY RING OF OCEAN."

Figure 7.1: Reproduced from Kirby (1884, p. 65).

THIS WORK WAS DEVELOPED BETWEEN 2013 AND 2015. THE ANALYSIS OF LAR-

VAL DRIFT IS AN EXTENSION OF WORK FROM A COLLABORATION WITH KIMBERLY LAI, DENYSE DOOKIE, AND NANDINI RAMESH.

The distributed commons model developed in chapter 4 is conceptual and analytic, and a qualitative connection with policy was explored. The previous chapters of this empirical section have provided a more quantitative perspective on spatial features in fisheries, which can inform and help calibrate a computational model of the distributed commons. The model presented below is simple but sophisticated enough to investigate complex interactions and questions of management.

This chapter is divided into two parts. The first part consists of three estimates of the rate of diffusion for marine resources. Diffusion is a key parameter for the distributed commons, and can be used as a summary statistic for the distance scale of spatial impacts in the distributed commons. The second part of the chapter describes a computational model of the distributed fishery commons, and then performs three experiments using it.

The three computational experiments are aimed at studying overfishing in the distributed commons and solutions to it. The first experiment considers the capacity of individual fisheries to overfish, and the role that their mobility plays. This investigates, in effect, how open-access plays out across space. The second experiment considers the capacity of the entire fishery to become over-exploited through the process of free entry. This is another feature of open-access, which can here be understood as access across the boundaries of the fishery system. The third experiment studies a possible resolution to the fishery commons problem through local management, and the issues of equity that result in that attempt.

The distributed commons model emphasizes the movement of both resources and resource users, in contrast to most spatial fishery models (e.g., Sanchirico and Wilen, 1999, Pezzey *et al.*, 2000, Sanchirico and Wilen, 2001b). Much of spatial fisheries research focuses on source-sink theory, using networks of patches rather than continuous space. Models that are fully spatial often assume a static distributions of fish resources. For example, Anderson and Seijo (2011) studies the interplay between exploitation patterns and economic incentives against spatially heterogeneous resources. In contrast to these approaches, the model presented here studies the endogenous emergence of resource patterns. The approach is

most similar to Smith *et al.* (2009), which use a diffusive growth model and relate these dynamics to economic patterns. Here, the fishing resources are initially uniform, but the role of diffusion and the spatial patterns of extraction shape the fishery. It is then through these emergent resource patterns that different users impact each other.

7.1   Estimating diffusion

The fish ecology model used below is a reaction-diffusion system,

$$\frac{\partial z}{\partial t} = \mathbf{A}\nabla^2 z + g(z)$$

where $z = S/K$ is the stock of a fishery resource as a fraction of its carrying capacity, as it varies in space and time. $\mathbf{A}$ is a vector of diffusion coefficients, and $g(z)$ is a logistic or another growth equation.

The key parameter of the distributed commons model is the rate of diffusion. Diffusion determines the extent of spatial and cross-boundary interaction. For very high rates of diffusion, a region can act like a well-mixed common pool resource. For very low rates, spatial effects are insignificant, since every location acts independently. It is between these two extremes that the distributed commons model is useful.

In both the model below and the estimates here, diffusion is treated as a discrete process. A one-dimensional diffusion process,

$$\frac{\partial z}{\partial t} = \alpha \frac{\partial^2 z}{\partial x^2}$$

can be expressed in discrete space and time as

$$\frac{z(t+\Delta t, x) - z(t,x)}{\Delta t} = \alpha \frac{z(t, x-\Delta x) - 2z(t,x) + z(t, x+\Delta x)}{\Delta x^2}$$

$$z(t+\Delta t, x) - z(t,x) = \frac{\alpha \Delta t}{\Delta x^2}\left(z(t, x-\Delta x) - 2z(t,x) + z(t, x+\Delta x)\right)$$

We define $\gamma = \alpha \frac{\Delta t}{\Delta x^2}$ as the discrete form of α, which increases with the length of time steps and as the modeled resolution of space increases.

7.1.1 Diffusion from the plankton analysis

The first estimate of diffusion is for plankton, much of which is spawn of larger species and which is a important component of ocean food webs. In chapter 5, the spatial effects studied are mediated by plankton, and the characteristic length scale is estimated in Kriging variograms. Combined with an estimate of how quickly plankton changes in time, the diffusion rate for plankton can be estimated.

Consider a diffusive model between two regions, A and B, with stock densities at time t of z_{At} and z_{Bt}, respectively. The recursive relationship between A and B as time evolves is,

$$z_{A,t+1} = \beta z_{At} + \gamma(z_{Bt} - z_{At}) + \epsilon_{At}$$

$$z_{B,t+1} = \beta z_{Bt} + \gamma(z_{At} - z_{Bt}) + \epsilon_{Bt}$$

This represents simple diffusion, with a decay term (β) which is necessary so that the entire system has a defined variance. The error terms, ϵ_{At} and ϵ_{Bt}, represent exogenous changes in the densities of plankton (for example, due to nutrient influxes or consumption by higher trophic levels), and are correlated in space.

We can calculate the variance and covariance of the stock densities in the two regions using these relationships. The covariance is,

$$C \equiv \mathbb{E} z_{At} z_{Bt} = \mathbb{E}\left(\beta z_{At} + \gamma(z_{Bt} - z_{At}) + \epsilon_{At}\right)\left(\beta z_{Bt} + \gamma(z_{At} - z_{Bt}) + \epsilon_{Bt}\right)$$

$$= (\beta^2 - 2\gamma\beta + 2\gamma^2)C + (2\gamma\beta - 2\gamma^2)V + S$$

Where $V = \mathbb{E} z_{At}^2 = \mathbb{E} z_{Bt}^2$, the variance of the values in region A and B over time; $C = \mathbb{E} z_{At} z_{Bt}$, the covariance of these values; and $S = \mathbb{E}\epsilon_{At}\epsilon_{Bt}$, the covariance in the spatial errors.

Similarly, the variance is,

$$V \equiv \mathbb{E} z_{At}^2 = \mathbb{E}\left(\beta z_{At} + \gamma(z_{Bt} - z_{At}) + \epsilon_{At}\right)^2$$

$$= (\beta^2 - 2\gamma\beta + 2\gamma^2)V + (2\gamma\beta - 2\gamma^2)C + E$$

Where $E = \mathbb{E}\epsilon_{At}^2 = \mathbb{E}\epsilon_{Bt}^2$, the variance of the errors in A and B.

These can be solved for C and V in terms of E and S. They are,

$$C = \frac{\zeta E + \psi S}{\psi^2 - \zeta^2}$$

$$V = \frac{\psi E + \zeta S}{\psi^2 - \zeta^2}$$

where $\zeta = 2(\gamma\beta - \gamma^2)$ and $\psi = 1 - \beta^2 + 2\gamma\beta - 2\gamma^2$.

The semivariance, $\hat{\gamma}$, measured in the variogram, is,

$$
\begin{aligned}
\hat{\gamma} &= \frac{1}{2}\mathbb{E}(z_{At} - z_{Bt})^2 \\
&= \frac{1}{2}(\mathbb{E}z_{At}^2 + \mathbb{E}z_{Bt}^2) - \mathbb{E}z_{At}z_{Bt} \\
&= V - C \\
&= \frac{(\psi - \zeta)E + (\zeta - \psi)S}{\psi^2 - \zeta^2}
\end{aligned}
$$

To estimate β and E, I use an autoregressive relationship among plankton concentrations measured 1 month apart in the same location:

$$z_{At} = \mu + \beta z_{A,t-1} + \eta_{At}$$

Note that the estimated $\eta_{At} = \epsilon_{At} + \gamma\epsilon_{B,t-1}$, so the mean squared-error of this regression, $F \equiv= \mathbb{E}\eta_{At}^2 = E + \gamma^2 E$ (assuming $\mathbb{E}\epsilon_{At}\epsilon_{B,t-1} = 0$). Similarly the covariance of the errors from this regression, $G \equiv \mathbb{E}\eta_{At}\eta_{Bt} = S + \gamma^2 S$ (assuming $\mathbb{E}\epsilon_t\epsilon_{t-1} = 0$). To calculate the covariance of the errors across space, we computer another variogram, the result of which, $\hat{\gamma}_\eta = F^2 - G^2$.

Some estimated values are shown in table G.1, and the estimates across the range of the variogram are shown in the comparison figure 7.7.

7.1.2 Diffusion from the protected area analysis

The second estimate of diffusion is for protected stocks neighboring exploited stocks. In chapter 6, the effects of marine protected areas are mediated by diffusion of fish out of the protected region. One interpretation of the additional surplus that is found in that chapter to scale with both the protection area portion and the size of the exploited stock is as

follows. Larger MPAs allow the stock density of the protected area to reach higher levels than it would if the fish were constantly diffusing out of the region. Furthermore, larger exploited stocks result in larger neighboring protected stocks. This then produces more fish which diffuse into the exploited area in the case where the stock is depleted below MSY, as shown in figure 7.2
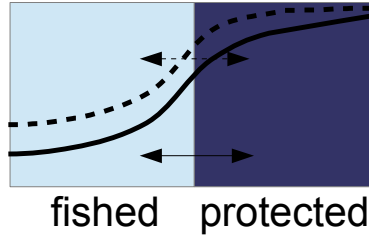
fished   protected

Figure 7.2: Conceptual relationship between protected and exploited stock levels under diffusion. The area below the solid curve is the fish density across space. An additional portion of density under the dashed line occurs when the fished area has a larger stock, resulting in also a larger stock in the protected area.

Consider a simple two pool model, with exploited and protected regions in equilibrium. Stocks in both regions increase under logistic growth. Stock in the exploited region is held in balance due to catch and diffusion with the neighboring protected region. Growth in the protected region is in balance only due to diffusion with the exploited region. In steady-state:

$$qES_1 = rS_1 \left(1 - \frac{S_1}{K_1} \right) - \gamma \left(K \frac{S_1}{K_1} - K \frac{S_2}{K_2} \right)$$

$$0 = rS_2 \left(1 - \frac{S_2}{K_2} \right) - \gamma \left(K \frac{S_2}{K_2} - K \frac{S_1}{K_1} \right)$$

where E is the catch effort per unit time, q is a catchability coefficient, S_1 is the size of the exploited stock, S_2 is the size of the protected stock, and γ is the diffusion per unit time, in terms of a fraction of the total carrying capacity ($K = K_1 + K_2$). This represents a chunk of water is exchanged between the two regions, with a fish density $\frac{S}{K}$, as above.

If $\gamma = 0$, this simplifies to the single pool result. We define the degenerate steady-state catch from this case as C_0,

$$C_0 = qES_1 = qEK_1 \left(1 - \frac{qE}{r} \right)$$

For $\gamma > 0$, an analytical solution exists, but the expression itself provides little insight.[1] Instead, it is evaluated across a range of parameter values in figure 7.3. The analytical coefficient that corresponds to β in the empirical logistic analysis in chapter 6 is approximately constant for small protected areas.
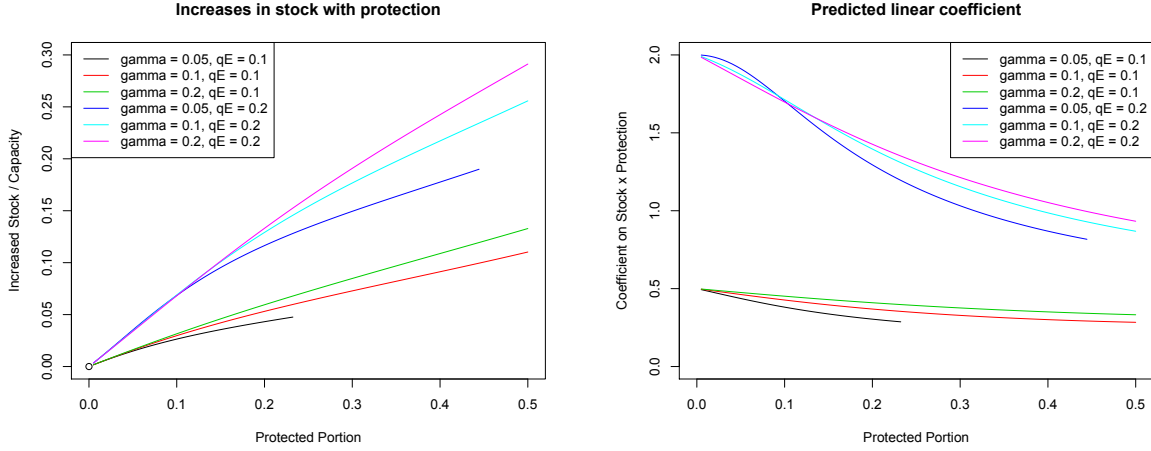
Figure 7.3: Results of diffusion, as a function of portion protected, for different values of qE and γ. The protected portion in the model is $\frac{K_2}{K_1+K_2}$. **Left:** Additional stock available to the fished region by virtue of protected area diffusion. This is $S_1(p) - S_1(0)$, where the different results from diffusion. **Right:** The value corresponding to β in the empirical analysis in chapter 6. This is $\frac{S_1(p)-S_1(0)}{pS_1(p)}$, the additional catch available, as a fraction of the MPA portion and stock.

---

[1] It is,

$$S_1 = X + \frac{-3\gamma pr + k^2(p^2 - p) + 3\gamma^2}{(9p^2r^2 - 9pr^2)X} - \frac{2k}{3r}$$

where $p = \frac{K_2}{K_1+K_2}$, $k = \frac{\gamma}{1-p} + qE - r$, and

$$X = \left(\gamma \frac{\sqrt{\begin{array}{c} \frac{1}{p}(4\gamma p^4 - 4\gamma p^3)r^3 + (-k^2 p^5 + 2k^2 p^4 + (-k^2 - 12\gamma^2)p^3 + 12\gamma^2 p^2)r^2 + \\ (2\gamma k^2 p^4 + (-4\gamma k^2 - 18\gamma^2 k)p^3 + (2\gamma k^2 + 18\gamma^2 k + 12\gamma^3)p^2 - 12\gamma^3 p)r + 4\gamma k^3 p^4 + \\ (-8\gamma k^3 - \gamma^2 k^2)p^3 + (4\gamma k^3 + 2\gamma^2 k^2 + 18\gamma^3 k)p^2 + (-\gamma^2 k^2 - 18\gamma^3 k + 23\gamma^4)p + 4\gamma^4 \end{array}}}{2 \; 3^{\frac{3}{2}} (p-1)^2 pr^3} + \frac{\gamma k(9pr - 9p^2 r) + k^3(2p^3 - 4p^2 + 2p) + \gamma^2 k(9p - 9) + 27\gamma^3}{54p^3 r^3 - 108p^2 r^3 + 54pr^3} \right)^{\frac{1}{3}}$$

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This model and the linear empirical model both give estimates of the additional surplus,

$$\beta p S_t = qE S_1(\gamma) - C_0$$

where $S_1(\gamma)$ results from the analytical diffusion expression.

For each assessed stock from chapter 6, I use the estimated value for the logistic r and K parameters, approximate qE as $\frac{C_t}{S_t}$, and then find the value of γ that explains the predicted increase in surplus under the globally estimated value for β. This γ represents diffusion over the entire year, and so is divided by 12 for comparison to the estimates from the previous and subsequent sections. The diffusion distance is taken as the square-root of the MPA area, representing the approximate distance that fish need to travel to be counted amongst the exploited stock. The comparison with the other estimates is shown in figure 7.7.

7.1.3 Diffusion from the larval floating

The third estimate of diffusion uses ocean currents in spawning regions. Fish spawn and floating eggs have little ability to move independently of ocean currents. The ocean currents that spawn are subject to for economically significant species is a concern for cross-boundary fisheries conflict. This section studies the regions and periods of spawning for 25 top fish groups.

Most marine species float for a period during their early development, as floating eggs and planktonic larva. The FishBase database contains the duration and characteristics of this period for some species (Froese and Pauly, 2014). Durations for species in the top economically important fish groups with floating data in FishBase are shown in table 7.1 and figure 7.4. The median floating duration from figure 7.4 is 45.5 days, and the mean is 61.5 days.

To estimate the total distance traveled during this dispersal period, we look at the spawning regions for each species. The maps in figure 7.5 highlight the EEZs of countries which are identified as having spawning activity in each month, for the 52 species for which Froese and Pauly (2014) contains spawning location data.

The distance traversed by floating spawn is determined by the velocity of the ocean current. We use the NCEP Global Ocean Data Assimilation System (GODAS) at a depth

Species	Larvae Duration	Egg Duration	Egg Floating	Float Bounds
Clupea harengus	160	NA	fixed	160
Decapterus pinnulatus	NA	0.38	buoyant	$[0.38, \infty)$
Decapterus polyaspis	NA	1.50	buoyant	$[1.5, \infty)$
Engraulis encrasicolus	37	2.00	buoyant	39
Engraulis japonicus	47	1.50	buoyant	48.5
Engraulis ringens	74	NA	buoyant	$[74, \infty)$
Gadus morhua	100	25.00	buoyant	125
Katsuwonus pelamis	20	1.10	buoyant	21.1
Mallotus villosus	150	NA	fixed	150
Nemipterus virgatus	NA	1.00	buoyant	$[1, \infty)$
Sardina pilchardus	40	NA	buoyant	$[40, \infty)$
Sardinella neohowii	NA	1.00	buoyant	$[1, \infty)$
Sardinops caeruleus	43	2.50	buoyant	45.5
Scomber japonicus	17	2.06	buoyant	19.06
Scomberomorus cavalla	12	NA	unknown	$[12, \infty)$
Scomberomorus maculatus	9	1.00	unknown	$[9, 10]$
Scomber scombrus	40	6.00	buoyant	46
Theragra chalcogramma	108	NA	buoyant	$[108, \infty)$
Thunnus albacares	25	1.40	buoyant	26.4
Trichiurus lepturus	NA	6.00	buoyant	$[6, \infty)$

Table 7.1: Available information in the FishBase database on larvae dynamics and fish egg development for top fish groups. The Float Bounds column represents a summary of the other columns, not a true representation of the bounds of possible range of floating durations, since the other columns only give approximate means.

Histogram of Floating Duration
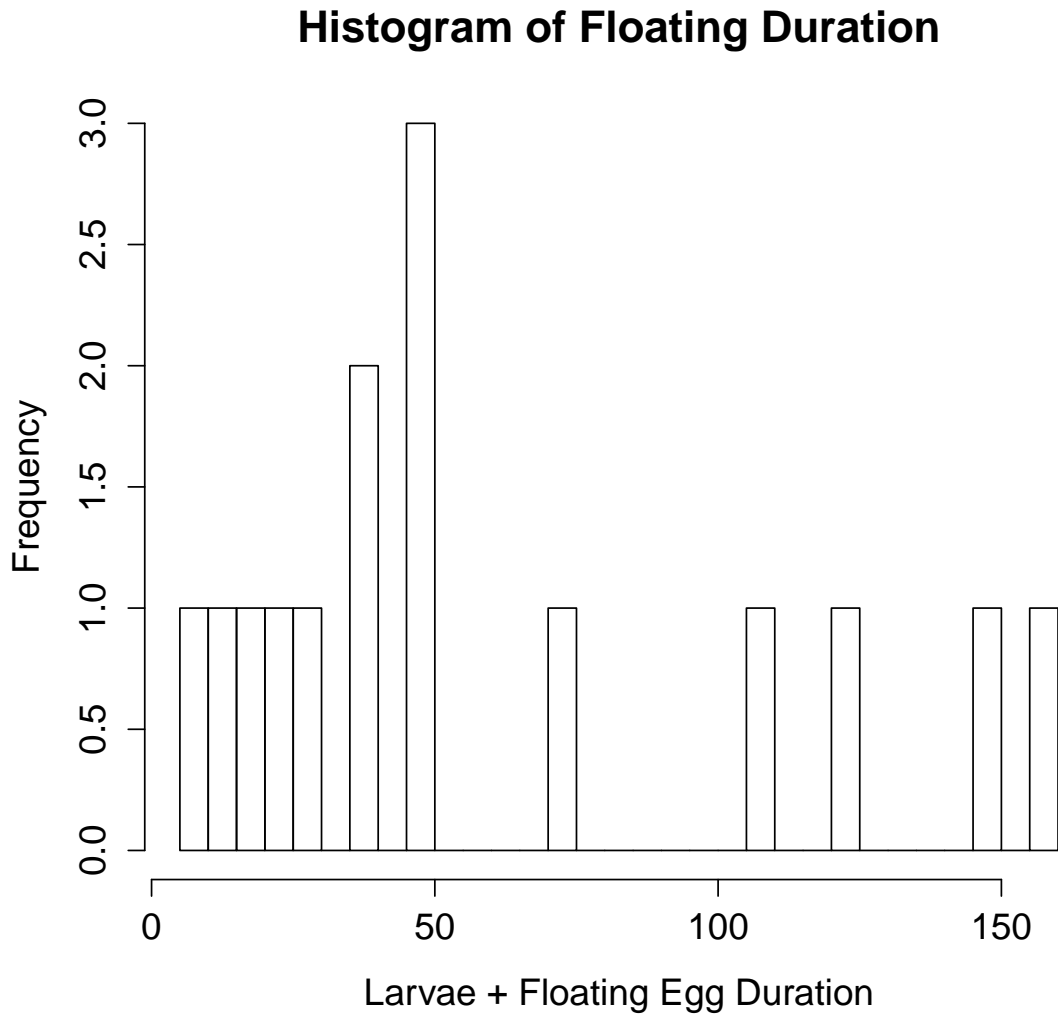
Figure 7.4: Histogram of the total floating time, using the larvae duration, plus egg duration in the cases where the egg is floating and the data is available. Species without larvae duration information are excluded, since this is typically the larger share of the floating duration.
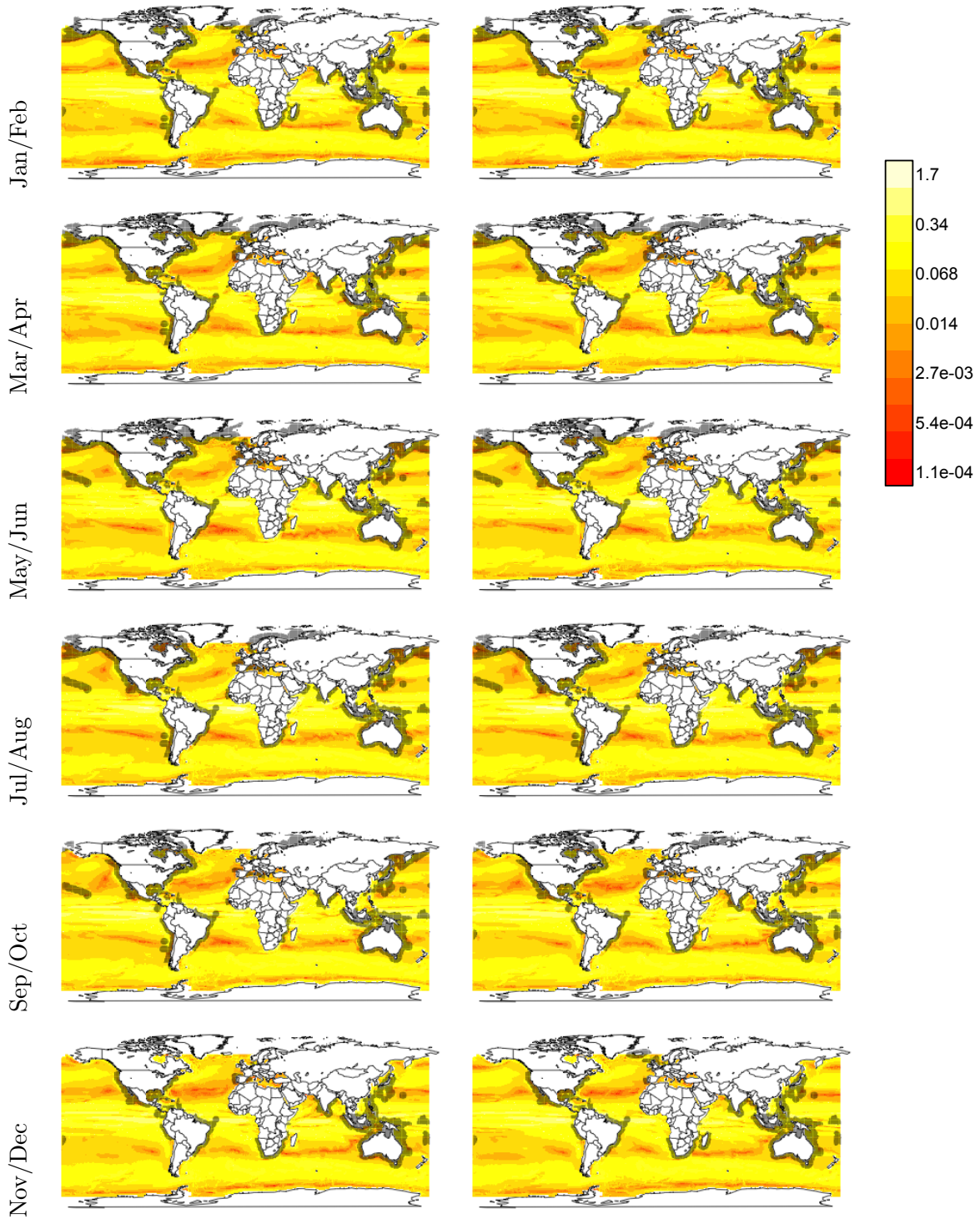
Figure 7.5: Masking maps highlighting areas of active spawning in each month of the year. Highlighted areas show all EEZs mentioned in the spawning tables available in Froese and Pauly (2014) for the studied species. Colors represent ocean current velocities, on a log scale.

of $5\,\mathrm{m}$.[2] The average current speed for spawning regions is calculated as follows.

For each of the species, the spawning data specifies which months spawning has been reported within specified regions. We associate these regions with country EEZs, of which 54 unique countries are included.

Of the species for which we have spawning data, 43 also have population distribution maps available in AquaMaps (Kaschner *et al.*, 2012). Within each of these EEZs, let the population distribution of fish for a given species be $D_i(x,y) = D(x,y) \cap EEZ_i$. Let the corresponding current speed across space in month m be $S_m(x,y)$. The average spawn speed is calculated as $\frac{\iint_{x,y} D_i(x,y) S_m(x,y)}{\iint_{x,y} D_i(x,y)}$.

The results are shown in figure 7.6. The median velocity is $0.092\,\mathrm{m/s}$ (95% CI 0.088 - 0.102).

Larval floating operates under a simpler model than diffusion, but it can easily be made to conform to it. Assuming that steady current of rate v flows to the right in a one-dimensional channel, the portion of the stock at $S(t,x)$ which exits a region of length Δx over a period Δt is $\frac{v\Delta t}{\Delta x}$, as long as $v\Delta t < \Delta x$. As a discrete dynamic (finite volume) equation,

$$z(t + \Delta t, x) - z(t, x) = \frac{v\Delta t}{\Delta x}(z(t, x - \Delta x) - z(t, x))$$

If the direction of flow is equally likely to be positive or negative, the expected stock change is,

$$\mathbb{E}z(t + \Delta t, x) - z(t, x) = \frac{v\Delta t}{\Delta x}\left(.5(z(t, x - \Delta x) - z(t, x)) + .5(z(t, x + \Delta x) - z(t, x))\right)$$

which is equivalent to the diffusion equation, with $\gamma = .5\frac{v\Delta t}{\Delta x}$. This transformation is appropriate if the ocean flow fluctuates, and is here used for comparison purposes. This estimate is also shown in figure 7.7.

7.1.4 Diffusion comparison

Figure 7.7 shows a comparison between the three estimated diffusion coefficients above. The range of values is large, from over 1 for currents in spawning grounds to just 0.001 for

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[2]GODAS data provided by the NOAA/OAR/ESRL PSD, Boulder, Colorado, USA, from their Web site at `http://www.esrl.noaa.gov/psd/`
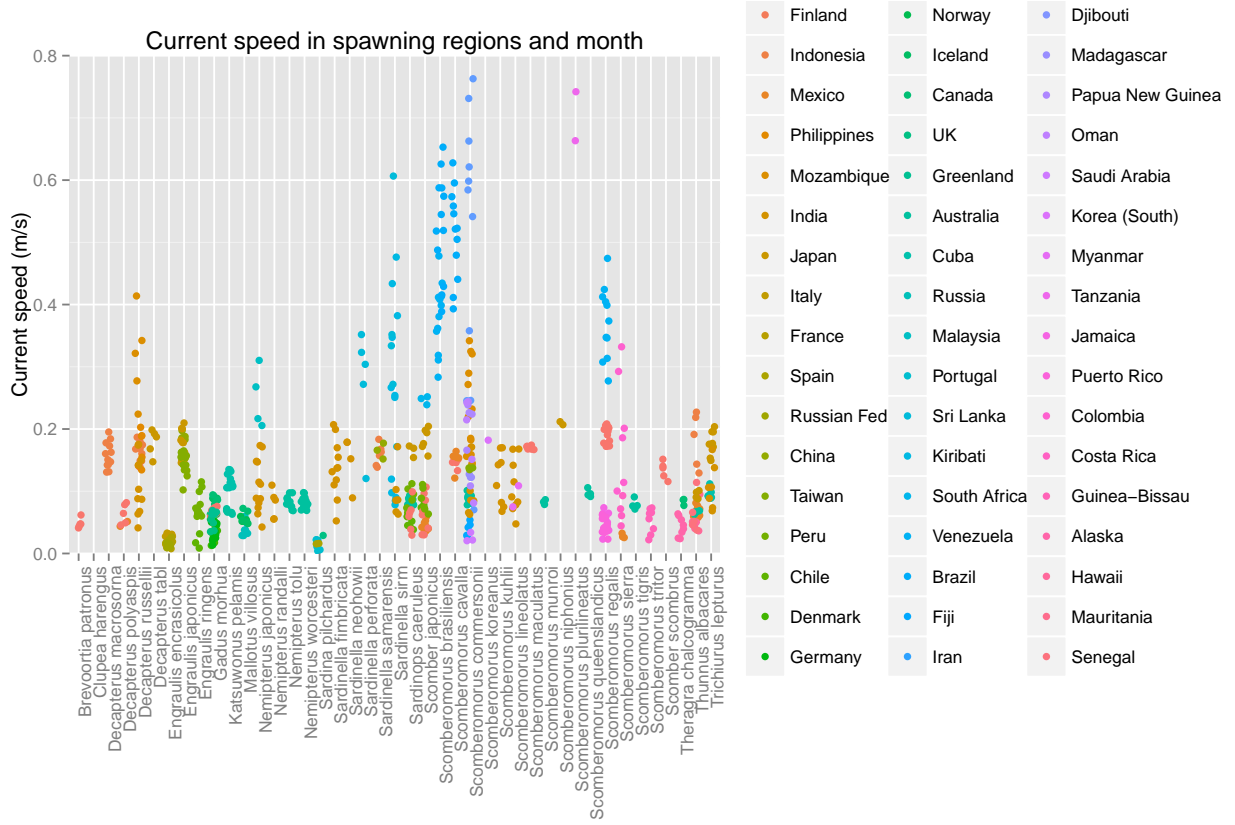
Figure 7.6: Ocean current speeds in spawning regions and months. Each point represents a region-month where the given species (displayed along the horizontal axis) is spawning.

some exploited stocks. This range reflects the capacity of species to affect their diffusion. Current speeds are nearly an upper bound on the rate of diffusion in the ocean, but many species will either actively swim to avoid this movement, or appear stationary at the level of a population confined to their natural range. Plankton were traditionally thought to simply drift, but the diffusion rates 100 times below current drift suggest much more active management.[3] This could be a combination of active swimming and depth control (Hardy, 1953) or operate more at the population level where individuals who drift outside of a region die off. Moreover, zooplankton appear to have greater ability to control their drift than phytoplankton, perhaps reflecting their greater mobility. Fish are the most capable of navigating currents and avoiding random diffusion, and correspondingly have the lowest diffusion rates.

7.2 An agent-based approach

Agent-based modeling is a useful tool for exploring the consequences of the distributed commons model. I construct a simple model of a distributed fishery with autonomous agents derived from a Gordon-Schaefer model. Agents represent individual fishers, while the fish resource is treated as a continuous field on a grid evolving under spatial reaction-diffusion.

The agent-based approach has particular strengths for informing the distributed commons. First, we are interested in the consequences of fisher decision-making, which can here be represented in the agents' logical rules. Second, inequality is a key concern in the commons (see chapter 2), and we can study the evolution of wealth and inequality between fishers. Third, the ABM framework is suited for studying emergence, where modifications to the underlying micro-scale assumptions and result in macro-scale emergent results.

The axioms of the model are as follows.

- The fishery is a one-dimensional array of cells, representing locations along a shore or river.

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[3] Ocean current speeds in the region studied for plankton are similar to those in spawning regions, at about 0.11 m/s (see appendix G.1).

Three estimates of marine diffusivity
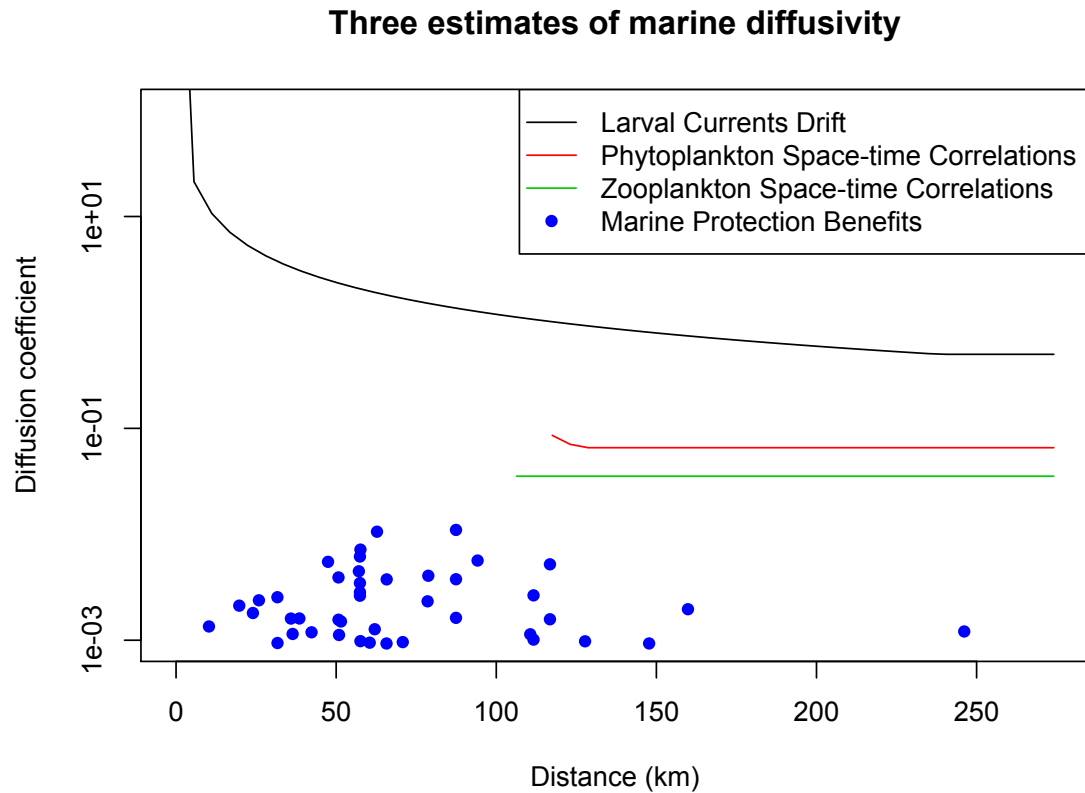
Figure 7.7: Comparison between three estimates of the marine diffusion coefficient, γ, over one month. The top line shows diffusion from the flow rate of ocean currents in fish spawning grounds. The red and green lines show diffusion for phytoplankton and zooplankton, respectively, based on the spatial and temporal covariance of plankton populations. The blue dots each correspond to an exploited fish stock for which the diffusion constant could be estimated, located at the square-root of the total marine protected area of the assessment region.

- Each cell in the array has its own stock of fish, which grows according to a logistic growth model: $(\Delta S_{it})_1 = rS(1 - \frac{S_{it}}{K})$.

- Every time step, a portion of the stock in each cell diffuses to neighboring cells: $(\Delta S_{it})_2 = \gamma S_{i-1,t} + \gamma S_{i+1,t} - 2\gamma S_{it}$

- Agents occupy a grid cell in each time step, and can only harvest fish in that grid cell in that time step.

- Agents join the fishery with some wealth and pay costs according to their fishing effort and earn profits by catching fish. They go bankrupt (and exit the system) if their wealth reaches 0.

- All agents receive the same price per ton of fish, pay the same operating costs for the same level of effort, and extract stock in their grid cell each time step according to $C_t = (1 - e^{-E\Delta t})S_t$. This is the integral of the continuous equation $C(t) = ES(t)$ with $\dot{S} = -C(t)$ over one time step.

One element that is absent from this model is the cost of travel far from port. The agents are assumed to be able to bank their catch immediately, and never need return to a particular port. In contrast, the different results in the static distribution spatial model used by Anderson and Seijo (2011) are driven by different assumptions about the cost of travel. In a model of fishing along a shore, port location and travel time can plausibly be ignored.

The operating costs, fish price, and diffusion rate are chosen so that a user can only make a profit by either moving around the fishery or taking advantage of diffusion. In particular, diffusion per time-step is 0.1 of the stock in each grid cell. The range of diffusion values found above is between .001 and 10 over distances between 25 km to 125 km (see figure 7.7). The value of $\gamma = 0.1$ is near the center of these estimates, and with $\Delta t = 1$ month and $\Delta x = 100$ km, we find $\alpha = 33$ km^2 / day. Now, we can understand what this value of α implies across a range of spatial scales.

The values of α, γ, Δt, and Δx are related. Taking $\alpha = 33$ km^2 / day and $\gamma = 0.1$ as in the model, we consider two values for Δt and the values of Δx that correspond to

them. If each time-step is a single day, suggesting that the ABM agents are allowed to sail at will, the spatial scale corresponding to an $\gamma = 0.1$ is 18 km per grid cell. The coastline of Maine, measured at a large scale is 367 km, or 20 grid cells (U.S. Department of Commerce and Administration, 1975). This is the number of grid cells used in two of the three experiments below. Alternatively, if each time-step is a year, suggesting that fishers are locked-in to regions through an institutional process, then the spatial scale for this value of γ is 350 km. When the coastline of Maine is calculated including tidal areas, it is 5597 km, or 16 grid cells at this scale (Office of Ocean and Coastal Resource Management, 2011). This suggests that space and time assumptions used here are appropriate for either open, deep-sea fishing, or trap-based, near-shore fishing, for species that diffuse at an intermediate pace similar to plankton.

I explore three experiments, by changing the rules by which agents move and harvest resources. They are,

1. **Comparing the maximum occupancy of stationary and mobile fishers**

2. **Exploring the consequences of local and global open-access rules**

3. **Locally restraining fishing to maintain maximum sustainable yield (MSY)**

7.3   Maximum occupancy

Mobile fishers are defined as be able to identify fish resources near them and move in the direction of greater fishing resources in each time step. Intuitively, a population of mobile fishers should be able to take better advantage of the resources available across a spatially distributed fishery than stationary fishers. However, I find that for evenly-spaced stationary users and under a range of biological and economic parameters, a greater number of stationary fishers can be maintained on a given fishery.

For this experiment, mobile and stationary fishers are constrained to use the same level of fishing effort. Both mobile and stationary fishers are initially placed at regular intervals spanning the fishery. For a fishery that is 10 grid cells long, only one fisher can be sustainably supported in both mobile and stationary cases. If more than one fisher uses the resource,

eventually one or both of the fishers will go bankrupt. Similarly, for 20 grid cells, three fishers can be sustainably supported in both cases. However, the asymptotically greater growth rate of sustainable fishers is clear at 40 grid cells. In this case, more than six mobile fishers deplete the resource, while a fishery with stationary fishers can support seven.

An example with 7 stationary fishers compared to 7 mobile fishers is shown in figure 7.8. As expected, stationary fishers deplete their local resources more quickly than mobile fishers. However, diffusion from undepleted areas continues to sustain them, like intersperced protected regions, with the total stock near the fishery's MSY. In contrast, seven mobile fishers are more effective at fishing down the resource. As a result, the total stock of the fishery dips below MSY, and one or more of the fishers eventually go bankrupt.

As with all results in this model, this effect allowing more stationary fishers to be sustainable than mobile fishers, is driven both by the constant effort assumption and the diffusion rate. For low diffusion rates, stationary fishers are not sustainable at all. For high diffusion rates, stationary fishers and mobile fishers experience identical outcomes. It is the intermediate region where the benefits of unharvested regions left by stationary fishers become significant. See figures 7.9 and 7.10.

7.4 Open-access rules

The model can simulate an open-access commons using a spatially-adapted Vernon Smith function (Smith, 1969). A new fisher is added to the fishery if the unoccupied grid cells contain sufficient fish to cover the fisher's startup costs. The profit after a time T can be approximated as,

$$\pi_T = p(1 - e^{-ET})S_0 - cT$$

where p is the price per ton of fish, S_0 is the available stock before joining, and c is the operating costs per time-step. The maximum value of this profit function is,

$$\max_T \pi_T = \frac{c}{E}\left(1 - \log\frac{c}{pES_0}\right)$$

At each time step, if this maximum value is greater than the initial wealth of the fisher, then a new mobile fishing agent is constructed within the fishery. This experiment is run
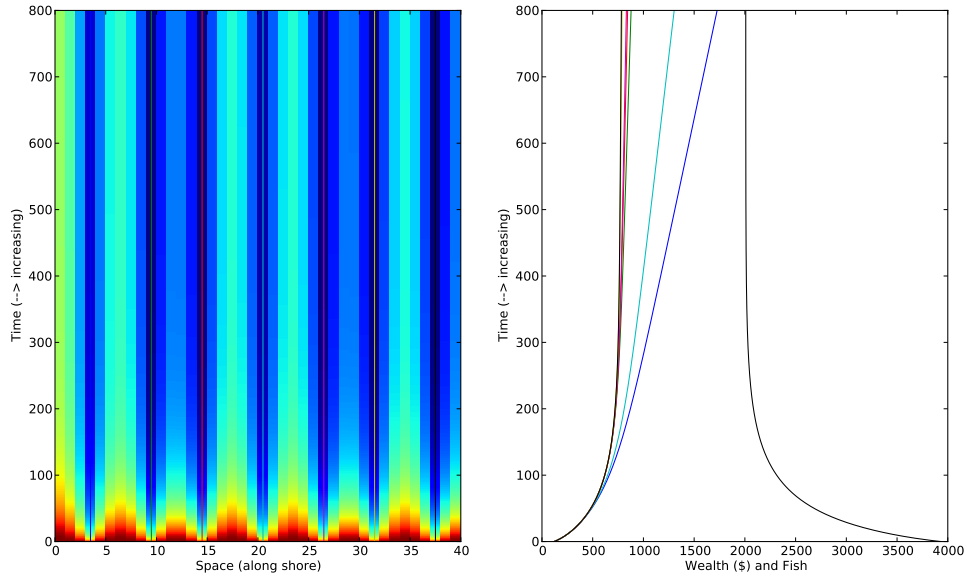
Seven stationary fishers
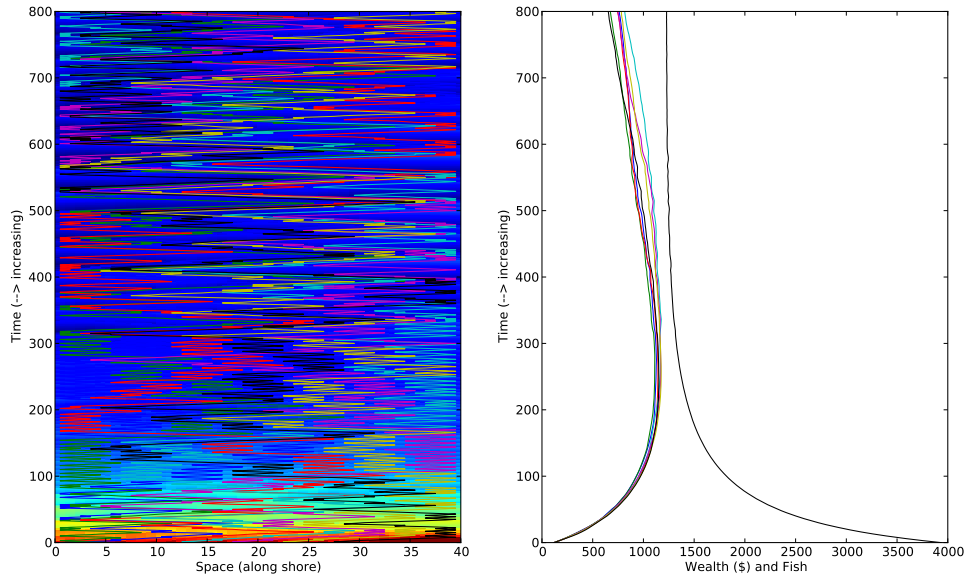
Seven mobile fishers

Figure 7.8: Each row is a time step, for 40 grid cells across 800 time steps. Background grid colors represent the density of fish biomass in each cell. Lines show the location of each fisher, and correspond to the colored lines in the right graphs showing fisher wealth. The black line on the right is the total fish stock across the fishery. **Top:** For stationary fishers, the grid cells occupied by the fishers are quickly depleted, but having 2-3 neighboring unfished grid cells provides sufficient diffusion to support the fishers. For this arrangement, the aggregate fish population asymptotes near MSY. **Bottom:** Mobile fishers move to grid cells with high fish biomass, which allows them to extract more biomass. The fishers' wealth lines peak at over $1000, but then decline as the fish stocks are no longer sufficient to support them.
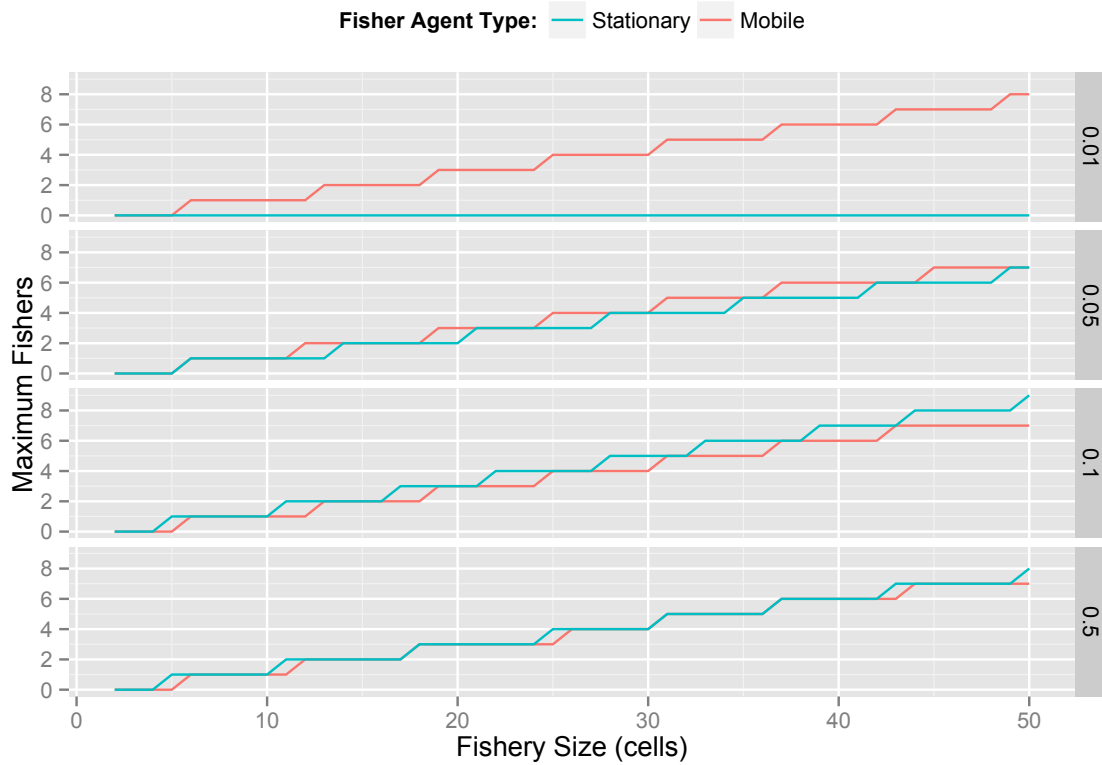
Figure 7.9: Maximum sustainable fisher curves, versus the size of the fishery. Each row is a different diffusion rate, labeled on the right. For very low diffusion rates, no stationary fishers are sustainable. As diffusion rates increase, the advantages of stationary fishing increase until more stationary fishers are sustainable than mobile ones at $\gamma = 0.01$. As diffusion rates further increase, the sustainable fisher counts for the two strategies converge.

Figure 7.10: Maximum sustainable fisher regimes, versus the size of the fishery and difussion rate. Colors denote whether more stationary fishers or mobile fishers can coexist on a fishery. For low diffusion rates, the mobile strategy dominates, in order to take full advantage of the fishery. At intermediate rates of diffusion, more stationary fishers can use the fishery, since they maintain unfished areas. At high rates of diffusion, the two strategies become indistinguishable.

both by treating all fish stocks as an aggregate for calculating startup profits, or by making the decision based on individual grid cell statuses (see figure 7.11).

In the simulation, both rules initially operate identically, with a new fisher added every period. The earliest fishers see the largest gains over the first 20 timesteps, with the last fishers to join having essentially flat wealth lines as their marginal costs match their marginal revenue.

Under the bulk open-access rule, after fishing agents begin to go bankrupt, new fishing agents are continuously added, since the whole fishery has sufficient resources, if the new agent could access them all. Under the local access rule, the fishery recovers before new fishing agents are added. However, in the local open-access rule, the open-access tragedy reoccurs after a period of recovery (between time steps 165 and 170). The third experiment solves this tragedy.

7.5 Maximum sustainable yield (MSY) management

I now allow fishing agents to "go into preservation mode", a state where they continue to occupy a grid cell but no longer fish and have reduced operating costs. For example, a fishery with seasonally enforced preservations can support greater numbers of fishers than one without. For this experiment, I require that fishers stop harvesting if their local stock levels fall significantly below MSY, defined as half of the carrying capacity (see figure 7.12). The result is a fishery that in aggregate maintains stocks near MSY, but at a social cost: while some fishing agents go into preservation due to their local conditions, others can continue to operate. Eventually, these can take advantage of the preserving fishers by harvesting their diffusing stocks, as happens to the cyan agent between periods 175 and 375. As a result, the stocks in the preserving agents' grid cells never reach a high enough level for them to return to fishing, and they eventually go bankrupt.

As a solution, I add a new rule, "tit-for-tat". Preserving fishers continue to monitor the region around them. If they see another fishing agent operating within their neighborhood, they return to fishing themselves, even if their local conditions have not attained MSY. Instead of resulting in greater collapse, this forces all of the fishing agents to synchronize
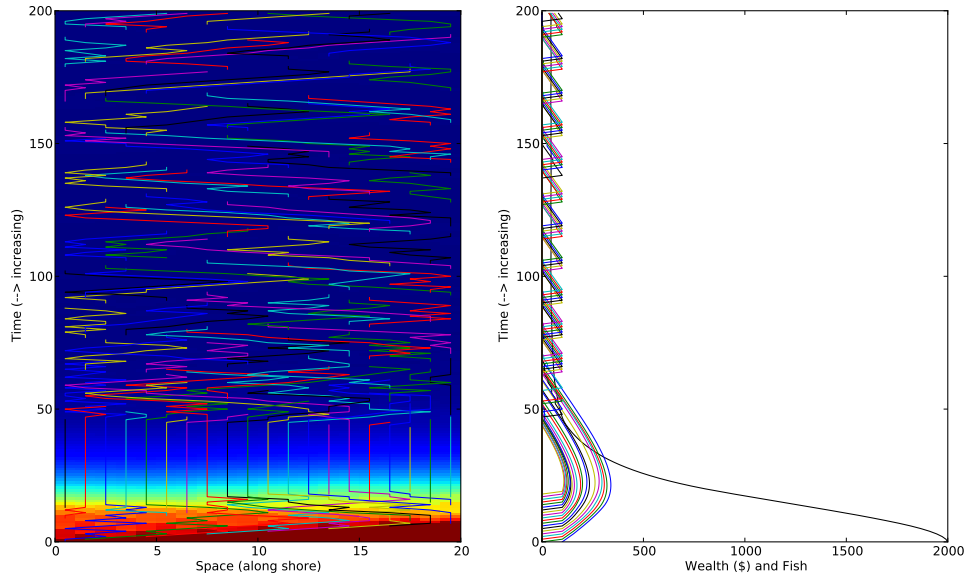
Bulk open-access rule
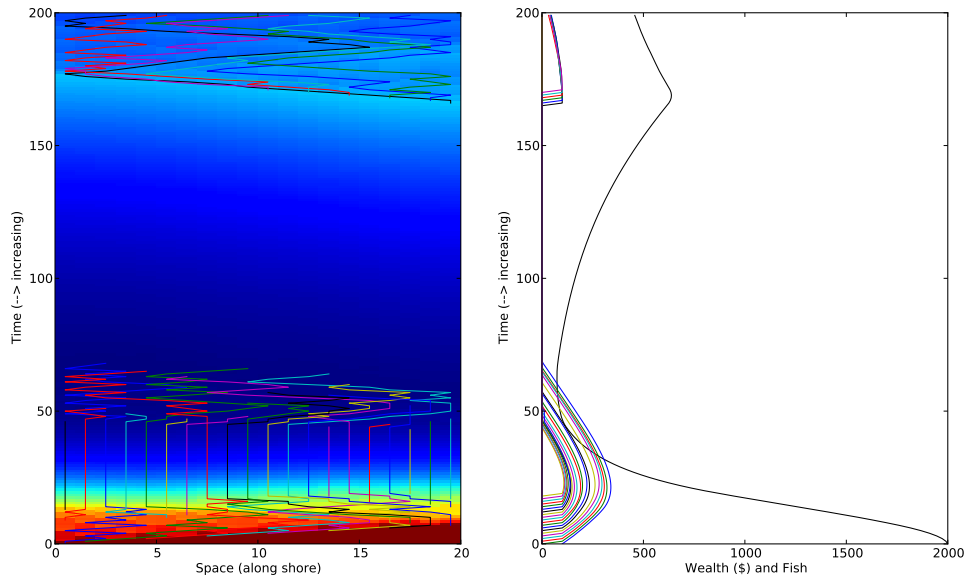
Local open-access rule

Figure 7.11: See figure 7.8 for an explanation of the axes. The first 45 time steps are identical between the two experiments: A new fisher is added each time step until all locations are occupied. Between time-steps 20 and 45, the agents fish down the stocks until agents begin to go bankrupt. **Top:** Under the bulk open-access rule, new fishing agents are continuously added whenever there is a free location, despite the negative slope in their wealth trajectories. **Bottom:** Under local open-access, the fishery recovers for about 100 time steps, until new fishers can join the fishery without having a negative wealth slope.

their preservations, ensuring both stable fish populations and equitable management (see the lower part of figure 7.12). This works because the rogue fishers who would otherwise exploit these preserving fishers are bound by the same MSY rule, they simply perceive different local conditions. Finally, a more aggressive form of the "tit-for-tat" behavior can be used to "starve" fishers who do not follow the MSY preservation rule, by fishing in close proximity to them.

The "tit-for-tat" rule also operates effectively under open-access rules. Wealth potential is calculated locally, considering only the surplus above the MSY stock (half of the carrying capacity). While initially too many fishers join the fishery, after a few have gone bankrupt, the remainder maintain the fishery near MSY.

7.6   Conclusion

This simple one-dimensional fishery model offers new insights into the dynamics and opportunities of the distributed commons. The range of fishing activity across a distributed commons can have a large impact on the sustainability of the resource, where movement restrictions (including protected regions) can turn an unsustainable fishery into a sustainable one. Spatial information also matters: there may be no remaining local availability of resources, even while the aggregate resource appears to still be able to support more extraction. And finally, management rules needs to be cognizant of the spatial aspects of the fishery as well, where requiring that every fisher not overly deplete a resource can save the resource but if poorly implemented fail to support the resource users.

The last example of local fishing management is also poignant with respect to the equality issues discussed in the chapter 2. When local fishers can reliably enforce their norms, as is famously the case in Maine lobster fisheries, the commons tragedy can be avoided. Where industrial fleets access the same resource under a laissez-faire regime that local fishing communities do not experience, these norms can quicken the fishing community's demise.

The one-dimensional fishery provides a simple case for exploring these and many future issues. In addition, it is possible to apply the same model to a two-dimensional fishing
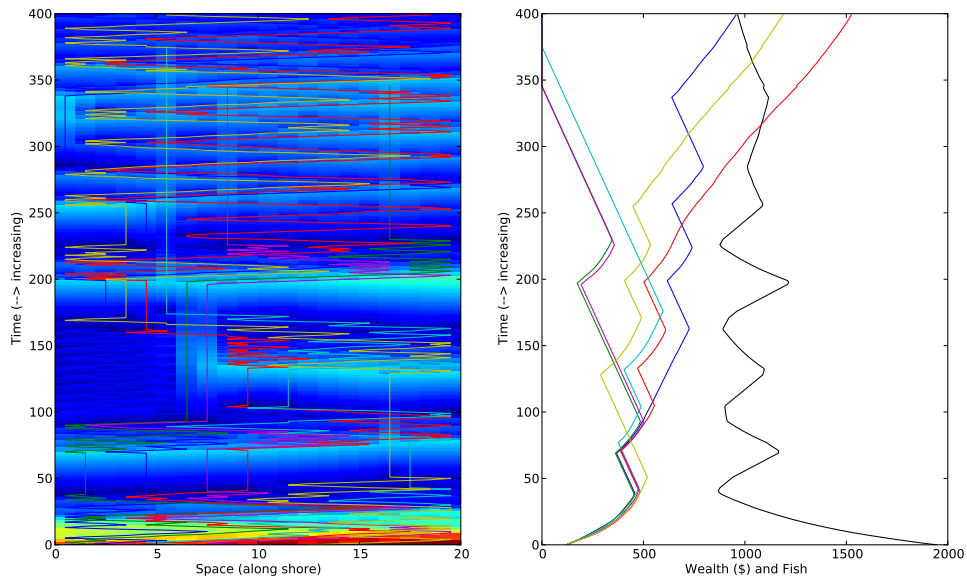
Local MSY rule
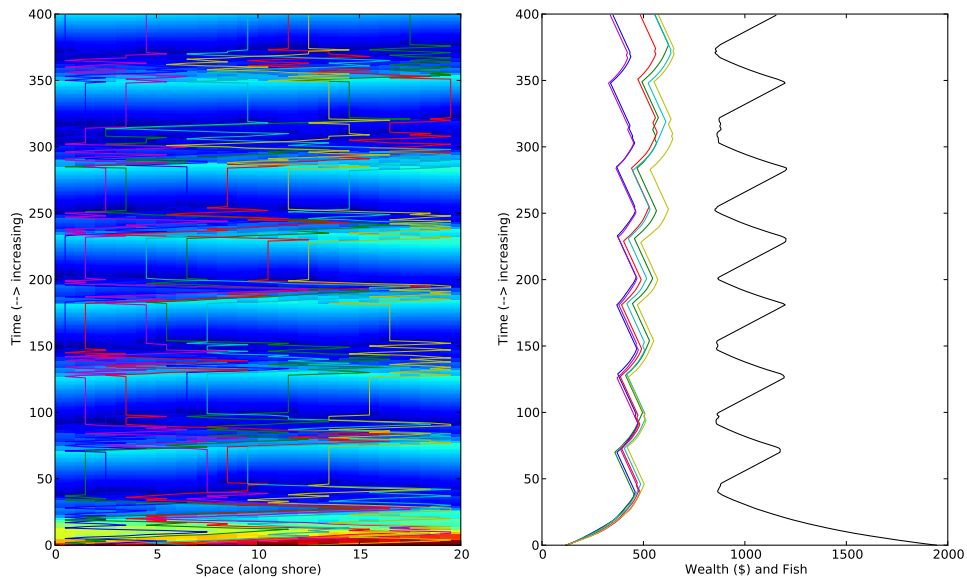
Local MSY/Tit-for-tat rule

Figure 7.12: See figure 7.8 for an explanation of the axes. **Top:** Local MSY management allows fishers to be taken advantage of and never return from "vacation". This happens to the green, purple, and cyan fishers between time-steps 230 and 400. **Bottom:** Local MSY/Tit-for-tat management forces fishing agents to synchronize their vacations. When only some fishers go on vacation, as happens between time steps 310 and 320, others return from vacation and fish down the fishery until all fishing agents must go on vacation.
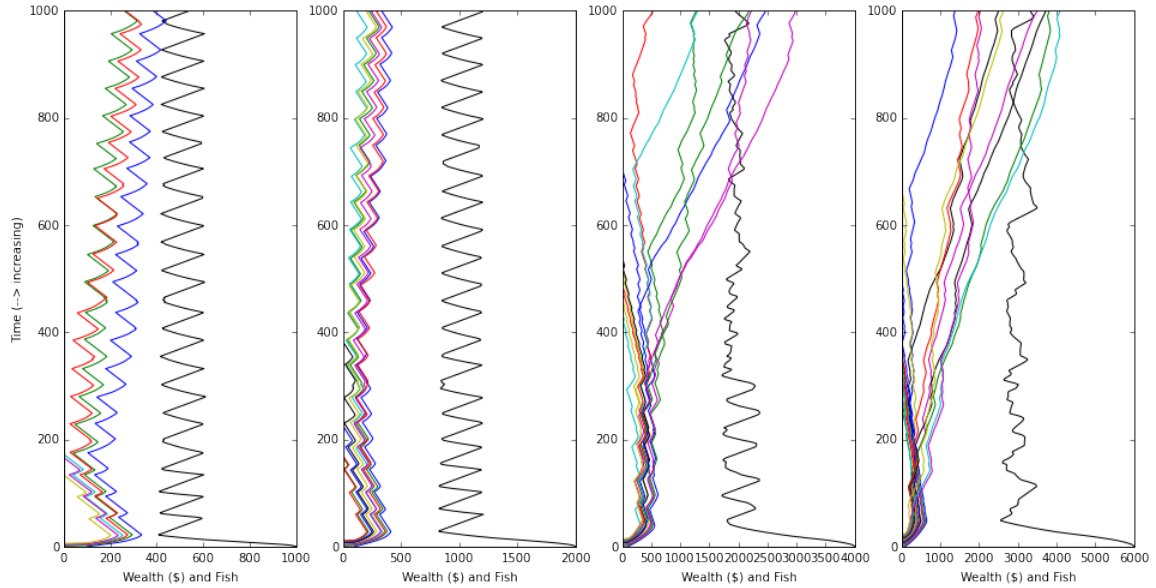
Figure 7.13: Open-access dynamics tit-for-tat MSY fishers, for four sizes of fishery. Fisheries larger than 20 cells rarely sync completely. However, the population of each fishery stabilizes after an initial rush of fishers.

context, as shown in figure 7.14. In this case, we see similar fractal patterns to those studied in the literature on spatiotemporal plankton-fish dynamics (e.g., Malchow *et al.*, 2002, Ishikawa and Takai, 2012). While the dynamics are more difficult to study, many of the results still hold. In particular, the sum of the fishing resource and fishing activity along each of the rows approximates the behavior of the one-dimensional fishery.

Computational modeling allows management practices, such as ITQs, MPAs, and gear restrictions to "play out" within an approximation to the spatial medium that they are ultimately aimed at. While many assumptions within the model are very simplistic, they improve upon simple aggregate models and provide new insights. Most importantly, while fishers can affect each other through their local impact, they can also produce robust management through local norms.
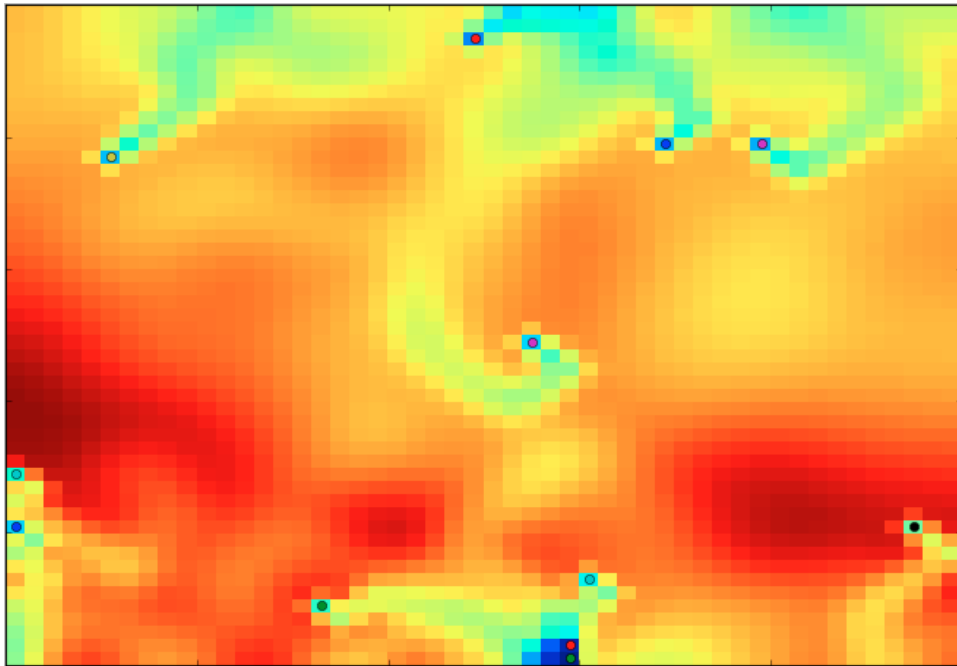
Figure 7.14: A snapshot of a fishery with 12 fishers. Blue areas have been fished-out, while red areas remain populated with fish.

Part III

Conclusions

Chapter 8

Conclusions

The tragedy of the commons remains as vexing and intractable as when Hardin first described it, nearly 50 years ago. However, in that time, our understanding of the functioning of actual common resources and the surrounding political economies has grown exponentially. The classic tragedy is a vast simplification, but one that continues to inform management and research.

During the same period, techniques in spatial analysis, global data collection, and computer simulations have revolutionized how we study and understand many social-ecological systems. These new techniques and data can help us understand how local features and interactions produce emergent consequences. Hardin described the collective consequence of agents acting independently, but it is exactly the interdependence of agents that produces aggregate collapse.

This thesis has aspired to shed light on the spatial dynamics of commons by focusing on the fishery. Improvements in data, analysis, and theory have all combined to provide a more complete and complex understanding of fisheries as social-ecological systems with emergent dynamics. Engaging fisheries on this level can provide new opportunities for management, conservation, and prosperity.

The investigations of this thesis are by no means comprehensive. Data limitations have necessitated the use of very simple models of ecological dynamics which miss age cohort effects, species interactions, and biochemical mechanisms explored elsewhere. A great many features of fishing communities are absent as well, such as the effect of markets and supply

chains, different types of gear, and the politics surrounding management decisions. Finally, some of the most active topics in fisheries research, such as the effects of climate change and ocean acidification and planning considerations for marine reserves are under-represented.

The studies presented here provide insights into specific mechanisms. The World Bank was not the only institution shaping ideas around fishery commons and management, but it provides a useful case study in chapter 2. Self-organized criticality, as considered in chapter 3, is one of several new models of emergence, but its implications help to inform cross-scale action and catastrophic events more generally. Additionally, these two chapters are not the only theories upon which the distributed commons model of chapter 4 is founded, but they highlight its central issues.

In the second part, the two empirical studies again provide a foundation for the integrative model of the last chapter. Chapter 5 is one possible approach for elucidating the spatial features of fisheries given existing data, but it highlights the role of spatial covariance relevant to the other studies here. The benefits marine protected areas provide to exploited fisheries, studied in chapter 6, are only one of their important effects. However, they emphasize the consequence of diffusion of fish out of MPAs, which supports the model in chapter 7.

The computational model of distributed commons presented in the last chapter is grounded in these empirical studies, but remains a "toy" model. Its simplicity, makes it possible to explore possible mechanisms behind observed behavior and the consequences of different assumptions. In this way, it resembles Hardin's original model. It is too far removed from reality to describe any particular species or fishery or to justify specific policy suggestions.

However, this model and the other studies in this thesis suggest new avenues for policy study. Fisheries are an ideal example of the need and potential for multi-level governance. For decades, governments have struggled with overfishing and degradation of marine and inland waters. These failures of management are partly driven by spatial and scale effects. Small fishing communities, fish stocks and food chains, factory ships, and policy-makers all act on different scales. The various components of fisheries policy, such as gear and catch restrictions, protected areas, and monitoring, also act on idiosyncratic scales.

Despite decades of experience, fish stocks continue to collapse. This is the consequence of a complex tragedy of the commons, playing out across scales and across boundaries. Fisheries are constantly confronted with multi-level issues: multiple stressors, from acidification to invasive species; environmental and human variability; cross-scale issues driven by the scale of fish ranges, environmental forcing, and foreign fleets; and failures of traditional management. Governance of fisheries that is focused on a single scale cannot effectively manage resources that have their dynamics driven by other scales. Fishery contexts have many of the characteristics that make commons management difficult: ownership rights are weak, dynamics are unpredictable, stocks are mobile and widely dispersed, and outside pressures are strong. Many effective traditional management practices fail when confronted with modern demands. These are many of the same problems confronted in other areas of sustainable development, for example around climate change, water use, and biodiversity loss.

Multi-level governance exposes possibilities for management that do not exist at any single scale. The general approach to fisheries multi-level management is called "co-management": regional and national government acts on a large scale with policies explicitly designed to support local fishing communities acting on small scales. The functions of large-scale government include monitoring of fish stocks, setting targets and allotments, identifying ecosystems for protection, enforcing boundaries, capacity building and legitimization, and facilitating communication. Each of these addresses a dynamic that cannot be captured at the local scale due to cross-boundary flows. The local management level focuses on ensuring fair fishing practices, coordinate amongst stakeholders, identify community needs, and monitor fisherman compliance and boundaries.

The experiences of fisheries multi-level management dove-tail with the consequences of spatial fisheries explored in this dissertation. Together, their implications are applicable to many other commons situations. In particular, five lessons come out of this work.

First, regime shifts, and tipping points and resilience, are concrete, measurable phenomena. We see them regularly in fisheries, and they manifest in cross-scale ways. Regime shifts can be very difficult to reverse, but sometimes they repair themselves far better than we could expect when released from external pressures. Proactive management can ensure

long-term sustainability and buttress resilience, and evidence that empirically distinguishes effective and ineffective management continues to mount.

Second, uncertainty and unpredictability are pervasive in many ecosystems, and the multi-scale perspective will not diminish that problem. Predictability and slow changes are typically unobtainable, driven by inter-dependencies, feedbacks, and history. Although these undermine predictive power, it is exactly these systemic features that we can learn about. Models that capture these can help us understand the range of behaviors of a system and how to respond to them. We need robust institutions that can coexist with chaos and catastrophe and learn from systemic failure.

Third, in multi-level situations, spatial organization matters. It matters because commons tragedies play out in spatial ways, fish populations are spatial entities, and spatial management approaches work. Models that ignore spatial structures, spatial heterogeneity, spatially-mediated resilience, neighborhood effects and teleconnected regions typically miss important dynamics. The conditions for sustainability and stress differ across space and cause key dynamics play out differently in different areas. Policies that do not support spatial choices or recognize the importance of spatial arrangements can miss important opportunities. The ways that local areas interact with each other is important for governance, because no problem is just local.

Fourth, boundaries within multi-level environments are not predetermined and where they are drawn can make a huge difference in sustainability. The divisions that seem natural at one scale can be integral components of another scale, which highlights the opportunities to make important choices. Boundaries can carve out healthy, protected areas, which, through cross-boundary effects, can support sustainability throughout a region. Boundaries create institutions, and they can be formed to delineate groups with common interests and areas with coherent dynamics. Boundaries allow groups the space to self-organize local institutions. The construction of institutional boundaries and other government policies has been at the heart of much harm in fishery commons, by undermining traditional regimes. The process of boundary construction needs to be married to a deep political process that engages both stakeholders and scientists.

Fifth, cross-boundary effects are the foundation of multi-level commons. Whether they

are in the form of the benefits beyond boundaries of protected areas, or cross-boundary pollution, or the impact of exploitation of resources on scales greater than that of a given community, when a commons is situated within a large area, the larger scale dictates the constraints of the local commons.

A wide range of empirical questions remains to be addressed, but I would like to identify two challenges ahead that are central. First, we need to better understand how multi-level and multi-scale perspectives can be incorporated into our quantitative models, to form new techniques of asking multi-level research questions, and to bring those results into the policy realm. Techniques for tying complex models to empirical research are currently being developed which can revolutionize this relationship.

Second, inequality is a core factor and a key challenge for multi-level governance. The history of local fisheries management is dominated by transfers of power and shifts in the control of exploitation, either through centralization, or to the much larger players that come with greater market forces. Early critiques of the privatization of the commons focused on the inequality that it created. The measurement of success in fisheries–whether by maximum yield or maximum economic benefit–is politically charged. Inequality also plays out across space and scales, and often the distribution of benefits is more important than the mean.

The study of fisheries can do much to help us understand the nature of sustainable development more broadly. If oceans are the blood that breaths life into the natural and human world, than fisheries are its pulsing heart. The ongoing fisheries crisis is not only a microcosm of the growing challenge to find new sustainable foundations for society; they form one of its most important pillars, where humanity, ecosystems, space, and institutions come together.

As E. O. Wilson formulated in what he calls "Wilsons Law": "If you save the living environment, the biodiversity that we have left, you will also automatically save the physical environment too. But if you save only the physical environment, you will ultimately lose both." We need to find a place in our world for healthy relationships with our ecosystems and the people who rely on them.

THE END

Part IV

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Part V

Appendices

Appendix A

Estimating wild catches

Many fisheries are unassessed, and the status of unassessed fisheries may be significantly different from assessed ones (Costello *et al.*, 2012). In addition, the dynamics of these fisheries may be different, undermining models, such as the one by Costello *et al.*, fit to assessed fisheries.

I use a catch-only model (COM), estimated using Bayesian methods. The model predicts the most likely trajectory of fish stocks to explain observed catches, as well as fishing effort and the catchability of the fish. A key input is the year of the peak catch. This catch is used to place bounds on the yearly stock growth, and determines the year in which fishing effort first reached its maximum level. The model estimates for each stock a logistic growth model, a normal error around stock growth, fishing efficiency as a distribution of catch per unit stock, and the fraction of fishing efficiency at the beginning of the time series.

I provide a prior of stock which is uniform up to the carrying capacity until the time of the peak catch, and then biased to be up to 50% of the carrying capacity (since this would be MSY). The key dynamic equations for the model are:

$$S_t \sim N\left(S_{t-1} + r\ S_{t-1}\left(1 - \frac{S_{t-1}}{K}\right) - C_{i,t-1}, \sigma^2\right)$$

$$\frac{C_t}{S_t} \sim \begin{cases} Beta\left(E\alpha e^{tlog(E)/T_{peak}}, \beta\right) & \text{if } t < T_{peak} \\ Beta(\alpha, \beta) & \text{if } t \geq T_{peak} \end{cases}$$

A range of estimates are shown in figure A.1.

Figure A.1: Stocks are trajectories shown with 50% and 80% confidence intervals. The dots at the bottom of the graph represent catches, and the line at the top is the mean carrying capacity.

Appendix B

Fish distribution maps

The AquaMaps database include generated population distribution maps, which are further subject to expert review (Kaschner *et al.*, 2012). The maps are based on observed relationships between species occurrence and environmental factors including bottom depth, temperatures, salinity, primary production, sea ice concentration, and distance to land.
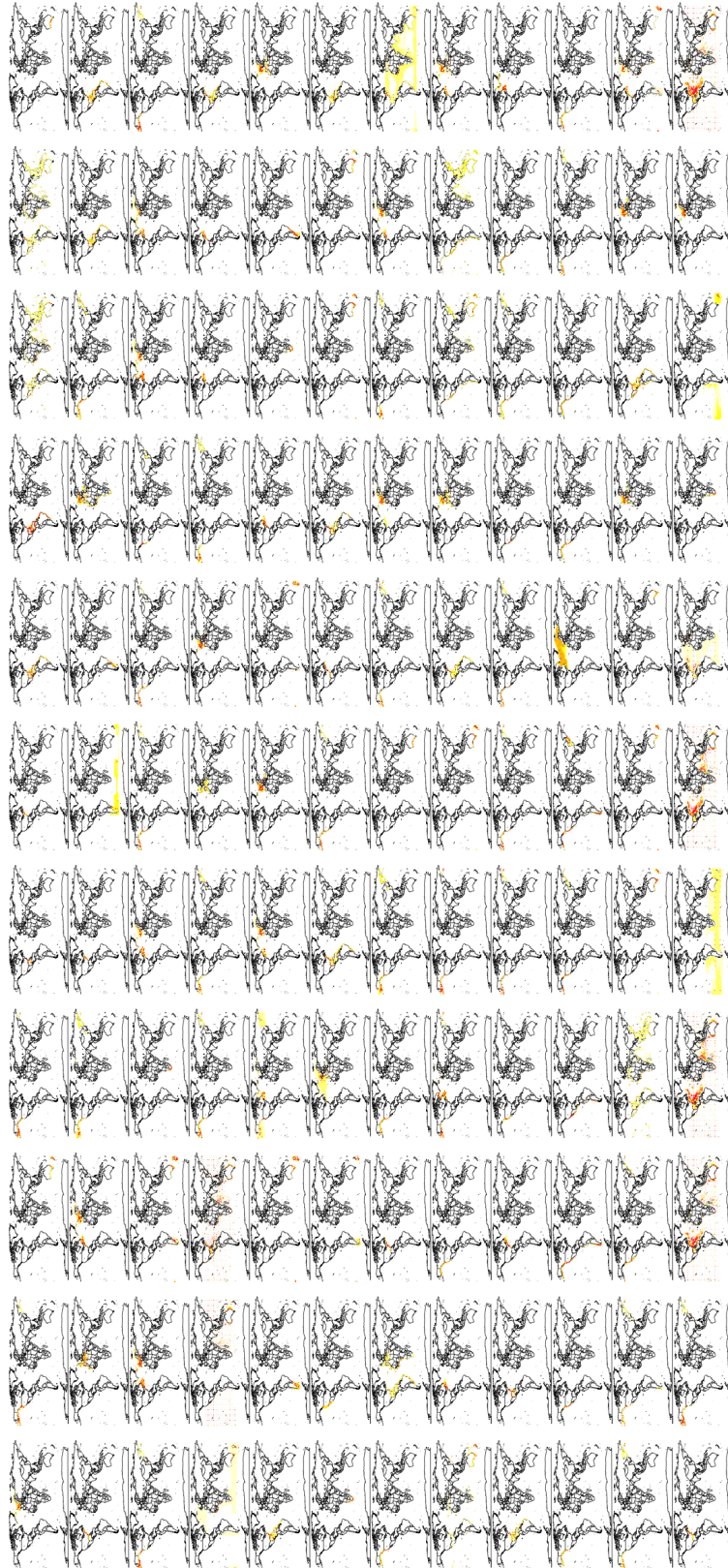
Figure B.1: 121 species with population distributions in the AquaMaps database.

Appendix C

An SOC model of ecosystem dynamics

The following model describes a plausible mechanism for SOC dynamics between ecosystem patches, similar to the neighboring regions studied in chapter 4. We begin with a simple model of ecosystem growth and harvesting:

$$Y(t) = rS(t)^{\alpha}\epsilon(t)$$

$$\frac{dS}{dt} = Y(t) - cS(t)$$

where $Y(t)$ is the recruitment minus natural mortality represented in the α exponent (where $\alpha < 1$), as a stochastic function of stock. Stock grows with the recruitment, and decreases with catch $(cS(t))$ resulting from a constant level of effort. Parameter definitions and values are shown in table C.1.

A run of the aggregate model for these parameters is shown in figure C.1. This model provides a baseline for the distributed model, which has the potential to exhibit SOC. The purpose of the distributed model is to incorporate discrete, distributed dynamics over a graph of ecosystem niches while holding true to the aggregate relationships.

The distributed model is developed on a circular graph of ecosystem niches. Initially, each niche is connected to two neighboring niches, a feature that will influence dynamics later. Each niche has an individual fish stock. All dynamics occur in this model at the

Both models: Growth concavity	α	0.3	Lower values produce more concavity in recruitment as a function of stock, reflecting natural limits.
Aggregate model: Initial stock	$S(0)$	0.1	
Aggregate model: Growth rate	r	0.4	
Aggregate model: Shock	$\epsilon(t)$	uniform(0.9, 1.1)	The multiplicative shock should be centered at 1.
Aggregate model: catch effort	c	0.1	This portion of the stock is harvested every period.
Distributed model: Niche count	N	100	
Distributed model: Baseline growth	r_0	0.1	Multiplied by edge count + 1 to become equivalent to the r value in the aggregate model.
Distributed model: Edge growth	$P(e)$	0.1	Chance that each niche forms a new connection to another niche each timestep.

Table C.1: Parameters for the aggregate and distributed (SOC) models.

niche-level, so I will not label each parameter with a subscript.

The growth dynamics are a distributed version of aggregate growth. Stock growth is divided into two terms: $g[t] = Y[t]$ is the growth term, and $d[t] = cS[t]$ is potential harvesting. Harvesting only occurs when the stock reaches a point near its local carrying capacity, making it attractive to fishers. As long as $g[t] \geq d[t]$, the entire extent of the growth applies– that is, $S[t+1] = S[t] + g[t]$. Eventually, $g[t] < d[t]$, at which point growth stalls in the aggregate model. This is reflected in the distributed model by literally stalling growth at this point: $S[t+1] = S[t]$, and adding a probability of harvesting every time step. The probability of a harvesting, $P(c)$, is such that the expected stock follows the aggregate curve:

$$S'[t+1] = S[t] + g[t] - d[t] = (1 - P(c))S[t] \implies P(c) = (d[t] - g[t])/S[t]$$

Each time step, a small fraction of niches get random connections to other niches. The number of edges a niche has enters into the equations to increase total ecosystem productivity, as $r = r_0(E+1)$, where r is the growth rate for a given niche, r_0 is a parameter across all niches, and E is the number of edges for a given niche (initially 2). An example niche network is shown in figure C.2.

Additional connection edges serve two additional purposes beyond their direct contribution to growth. They decrease the characteristic path length of the entire network, inching
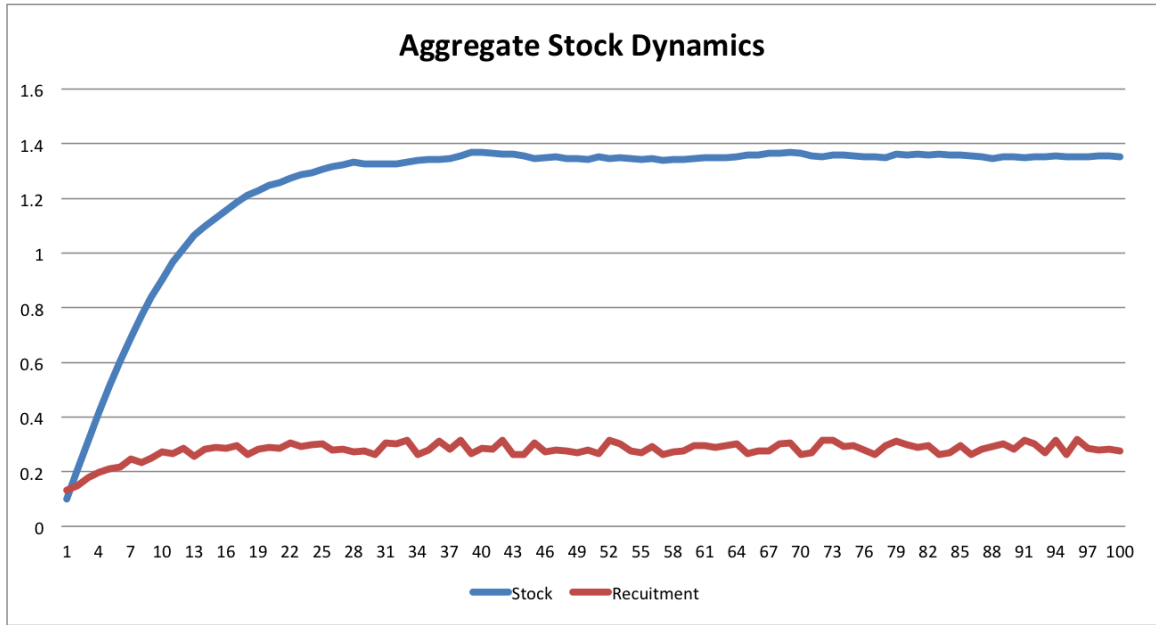
Figure C.1: Sample run of the basic ecosystem model.

it closer to the small-world network typical of real ecosystems, in a pattern similar to the one used by Watts and Strogatz (1998). Further, by increasing connectivity, they provide more pathways along which harvests can result in a propagation of collapses.

When a niche is either harvested, its stock is set to 0 and its connections to other niches are removed. This lowers the productive potential of those neighboring niches, which can destabilize them resulting in an avalanche. The resulting boom-and-bust dynamic, characterized by low stocks punctuated with great periods of ecosystem productivity, results naturally in the modified model (see figure C.3).

High ecosystem productivity appears once the niche network is highly integrated. However, there is a critical point of integration which causes global instability, and system-wide cascades can occur. This suggests that there are three dynamics within an SOC-Ecosystem system, as driven by $r(E)$ and c (see figure C.4).

Disconnected Regime: The critical SOC integration is below the point where a stock can be highly productive. High productivity regimes cannot occur.

Boom and Bust: The critical SOC integration is above the high-productivity integration, so the normal state of this ecosystem will oscillate between accelerating growth and
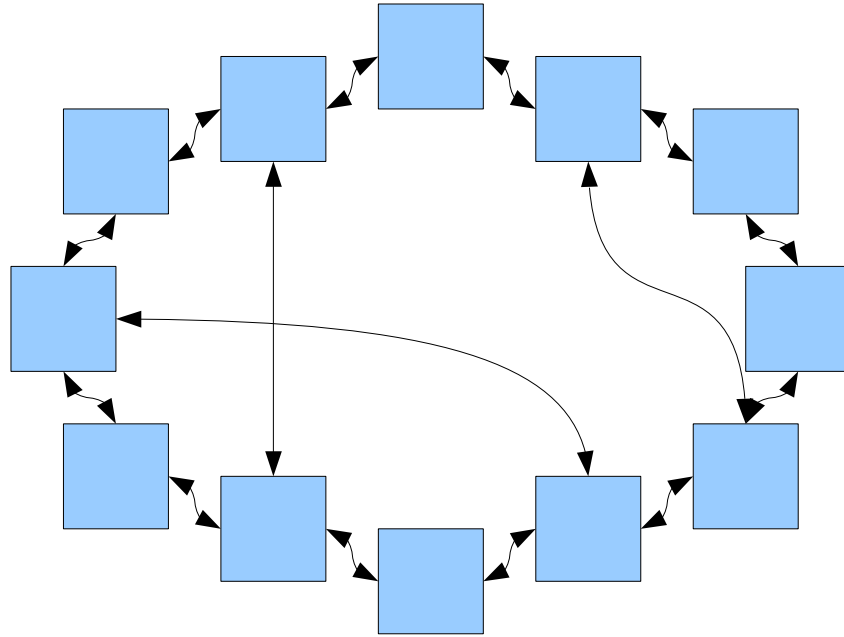
Figure C.2: Connected network of ecosystem niches, showing some cross-connections from long-distance migration.

harsh collapses. This is shown in figure C.3.

Resilient Regime: The critical SOC integration is above the stagnation integration. Large collapses will be rarer than the power law expectation, and the ecosystem quickly returns to a highly productive regime.

Ironically, it might be exactly in the "Boom and Bust" region where fisheries try to manage their stocks. The disconnected region is unproductive, so management will decrease fishing effort in an attempt to allow the ecosystem to recover. The resilient regime is considered to be under-harvested, since many regimes remain to support others.
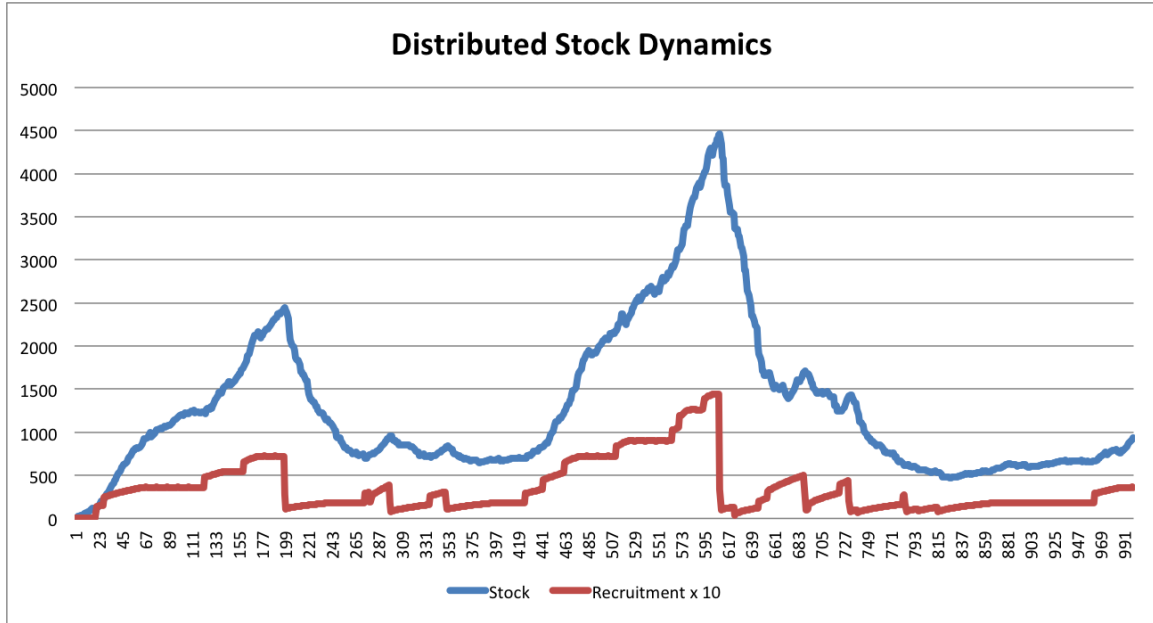
Figure C.3: Example run of the distributed model. Boom and bust dynamics immediately result from the existence of connections between niches.
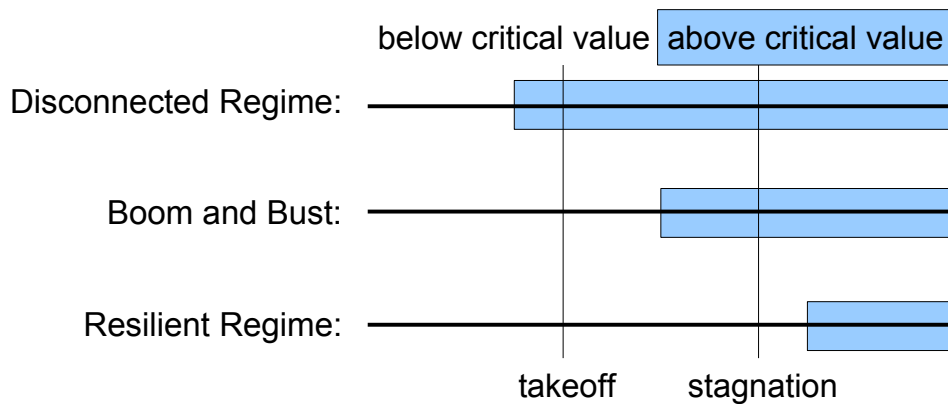
Figure C.4: Three possible bioeconomic dynamics in the SOC-Ecosystem system.

Appendix D

Diffusion and logistic growth scenarios

The relative strengths of the growth and diffusion in a fishery with both determine the amount of interaction between users. This is studied with a system that evolves under both in discrete time:

$$S_{i,t+1} = S_{it} + rS_{it}\left(1 - \frac{S_{it}}{K}\right) + \gamma S_{i-1,t} + \gamma S_{i+1,t} - 2S_{it}$$

The steady-state catch is determined by the ratio of r to γ, where a high ratio results in each grid cell acting like a distinct fishery with little cross-boundary flows (shown in figure D.1), and a low ratio produces high cross-boundary flows and strong interactions (shown in figure D.2). The values from figure D.2 are applied in chapter 7.

Figure D.1: One and two agents extracting from a logistic-diffusion system with low diffusion ($\gamma = 0.01$ and $r = 0.025$). Low diffusion causes a single user to only partially draw down the resource, and two users do not interact much. **Top, Left:** Steady state stock for a single user, at maximum sustainable yield. **Top, Right:** Sustainable yield across levels of stock for a single user. **Bottom, Left:** Steady state stock for two users, at maximum sustainable yield. **Top, Right:** Sustainable yield across levels of stock for two users.

Figure D.2: One and two agents extracting from a logistic-diffusion system with high diffusion ($\gamma = 0.1$ and $r = 0.025$). High diffusion causes a single user to entirely draw down the resource, and causes two users to interact strongly. **Top, Left:** Steady state stock for a single user, at maximum sustainable yield. **Top, Right:** Sustainable yield across levels of stock for a single user. **Bottom, Left:** Steady state stock for two users, at maximum sustainable yield. **Top, Right:** Sustainable yield across levels of stock for two users.

Appendix E

Inferring spatial fisheries supplement

E.1 Spatial fisheries supplemental material

E.1.1 Data selection

We considered a variety of anchoveta landing series for Peru to analyze in conjunction with the plankton data. FAO provides yearly production data since 1950, while IRIs EFFP contains both monthly catches from 1953 to 1987 and weekly catches from 1993 to 1999. All sources provide the catch in tonnes. The graph below displays these three sources, scaled to daily values.

Between these, the weekly data set seems superior for three reasons. (1) It provides the most data points within the time period for which plankton data is available (330 as compared to 34 (yearly) and 322 (monthly)). (2) It corresponds very well to the yearly totals, while the aggregate monthly data diverges considerably after 1976 (see figure below). (3) The weekly data is available separately at three ports, Pisco, Chimbote, and Paita. These ports account for 65% of the Peru anchoveta catch during this time period.

The monthly data is available disaggregated into length classes (monthly tonnes by length, 4 - 20 cm). Incorporating this length data as a proxy for age distributions could be used to improve our life-cycle results.
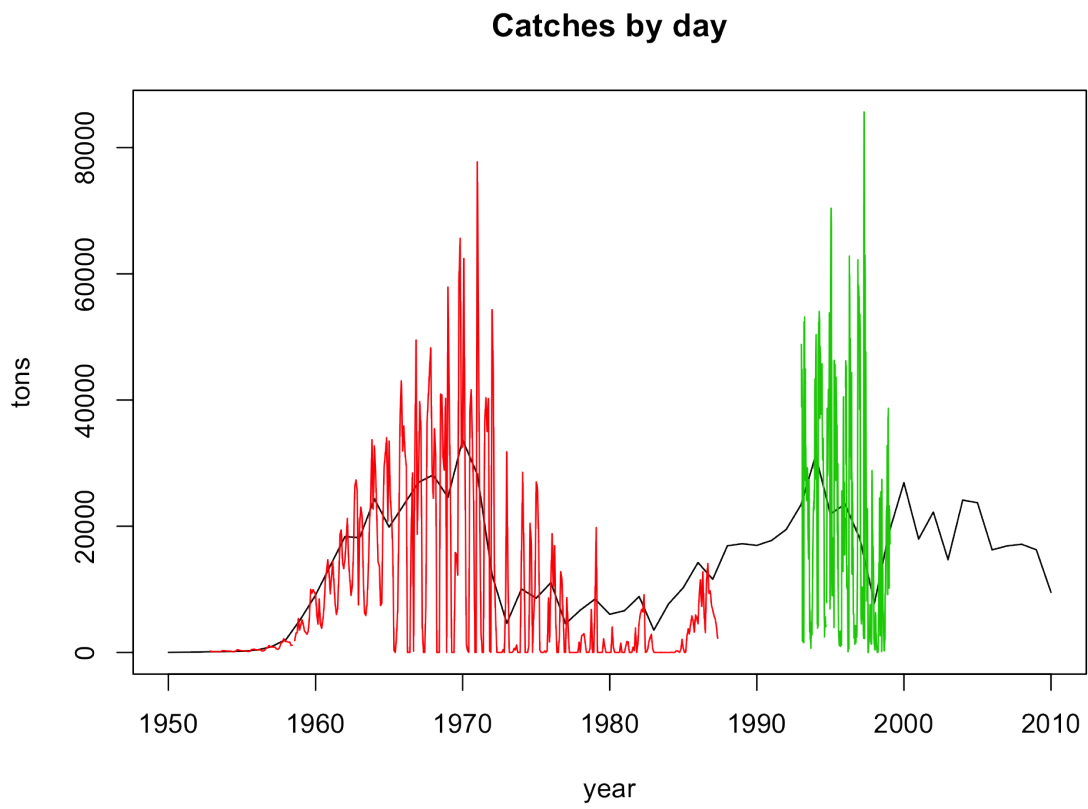
Figure E.1: Yearly (black), monthly (red), and weekly (green) catches of anchoveta.
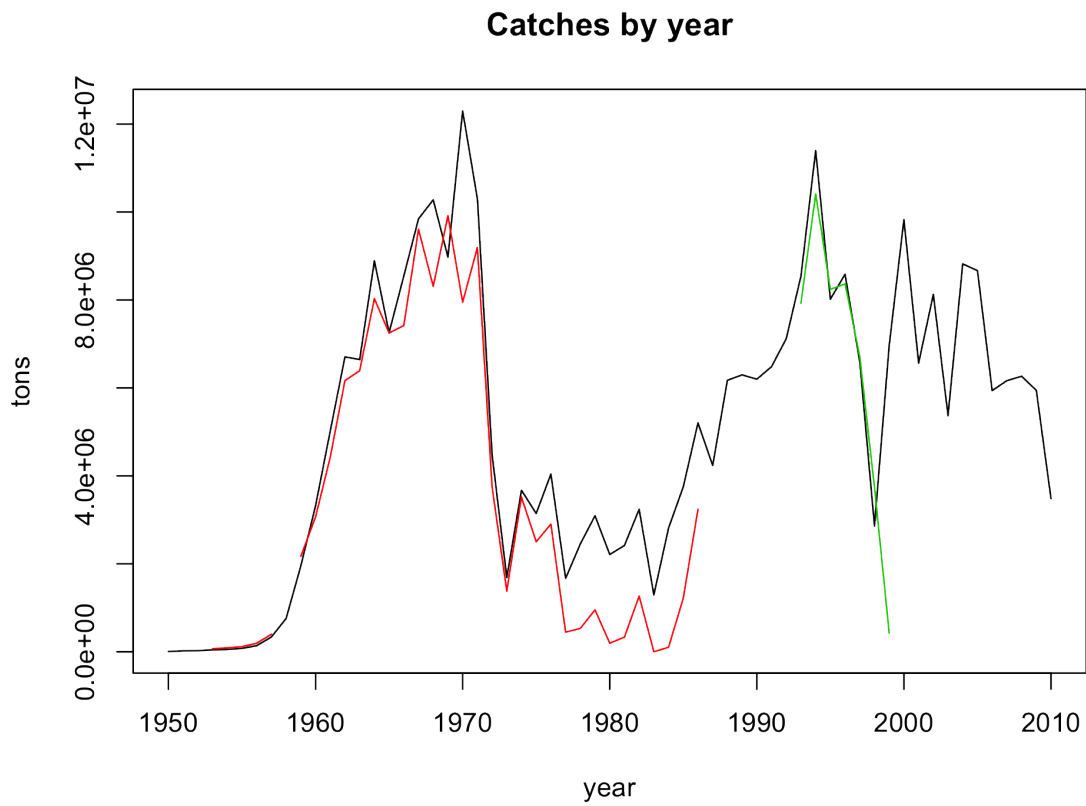
Figure E.2: Yearly (black), and monthly (red) and weekly (green) catches of anchoveta aggregated to the yearly level.

E.1.2   Isotrophy

The data does not show evidence of scaling issues between latitude and longitude. This is diagnosed by comparing the distribution of squared derivatives in latitude and longitude, as shown below. This is the distribution of the difference in sequential values of plankton, across all time, divided by the longitude or latitude distance between them, analyzed in squares.
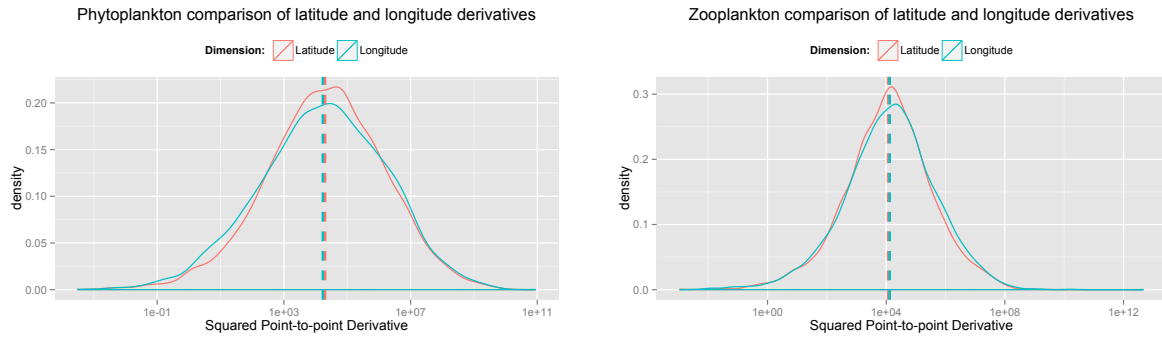
Figure E.3: Distributions of the point-to-point derivatives of phytoplankton and zooplankton along the longitude and latitude axes, with means shown.

For phytoplankton, the squared derivatives in latitude has a slightly higher mean than those in longitude (a ratio of 1.18), however for zooplankton, the squared differences in latitude have a small mean (a ratio of 0.88).

This is a clear difference in plankton changes along the coast, compared to the radial direction out from the coast, which is not accounted for in our analysis.

E.1.3   Autocorrelation

Temporal autocorrelation in the plankton data reduces the statistical significance of the plankton-growth correlations we study. Figure E.4 shows the autocorrelation for average plankton observations, across temporal lags. Figure E.5 shows the autocorrelation by grid-cell with a one month delay.

For correlations with average observation values, we correct for autocorrelation by multiplying standard errors by $\frac{1-\rho}{1+\rho}$, where ρ is the phytoplankton and zooplankton autocorrelations, respectively. For spatial estimates, we correct within each grid-cell individually, by

Autocorrelation of monthly average phytoplankton

Autocorrelation of monthly average zooplankton
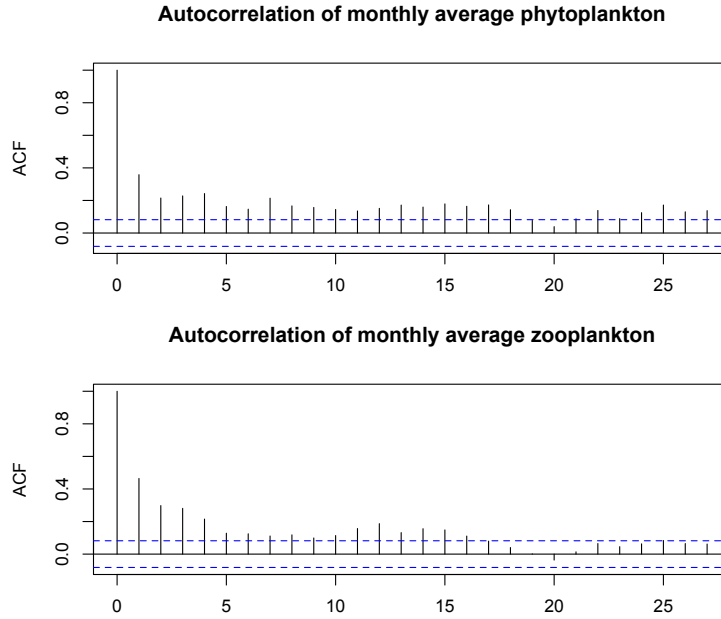
Figure E.4: Autocrrelation in average plankton observations. The x-axis denotes lags in the autocorrelation estimate. Single-month lags produce a 0.36 autocorrelation for phytoplankton and a 0.46 autocorrelation for zooplankton.

first multiplying standard errors as above separately for phytoplankton and zooplankton, setting insignificant values within each grid to zero, and than averaging them.

E.1.4 Expectation maximization procedure

As usual, the EM algorithm alternates between estimating a probability distribution and identifying the maximum likelihood parameters under it. However, the complicated functional forms involved make identifying the maximum likelihood parameters difficult. Instead, a randomized algorithm is used.

The estimation maximization procedure used the following steps:

1. Assume initial values for S_0, α, β, and $\{\gamma_i\}$. The procedure is very sensitive to this initial set of values. The values used are listed in the table below.

2. Model the stock growth under these parameters, producing a sequence $\{S_t\}$.
   Estimation
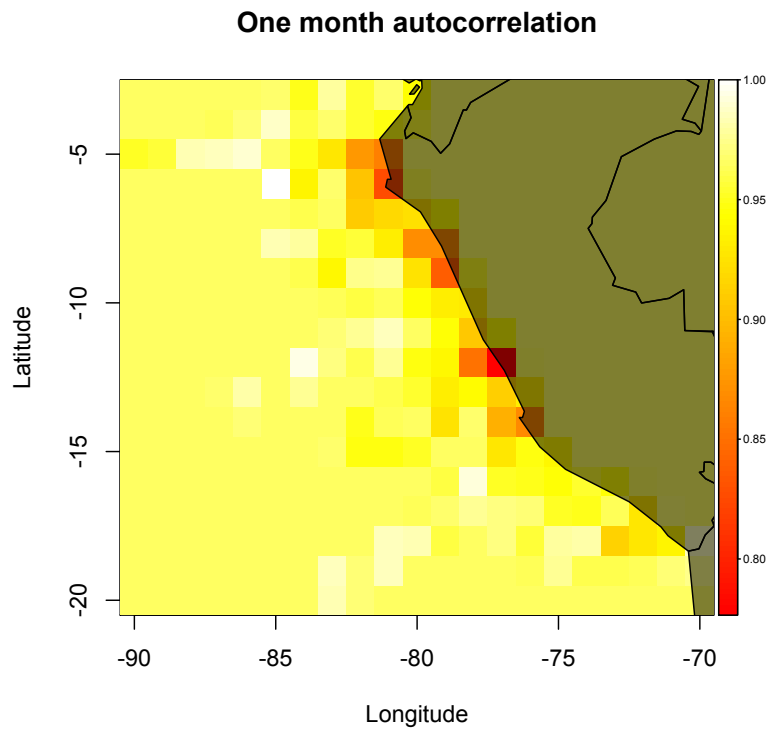
Figure E.5: Autocrrelation by grid-cell for plankton. Autocorrelations are calculated using a one month delay separately for phytoplankton and zooplankton, and then averaged for each grid cell.

3. Estimate the probability distribution for catch given effective stock, $f\left(\frac{C_t}{S_t \sum_i \gamma_i P_{i,t}}\right)$, as a kernel-estimated empirical distribution.

4. Determine the summed log-likelihood of the sequence of observed catch and estimated stocks, under the estimated distribution.

   <u>M</u>aximization

5. Construct a randomly adjusted new set of parameters, slightly changed from the most optimal set identified.

6. Model the growth under these parameters.

7. Calculate the summed log-likelihood under the existing empirical distribution.

8. If the summed log-likelhood for the new parameters is greater than the previous most-optimal parameter set, use these new parameters as the new most-optimal set and return to step 3.

9. Otherwise, return to step 5.

Below are the initial parameters used, a rationale for selecting them, and the final parameters determined by the EM algorithm. Rather than reporting α and β, we show the underlying logistic parameters, R (the growth rate) and K (the carrying capacity).

	Initial	Rationale	Final
S_0	16 790 000	28 x maximum catch	22 790 000
$R = \alpha - 1$	.01	arbitrary	0.007745
$K = -(\alpha - 1)/\beta$	1 679 000 000	100 x S0	10 400 000 000
γ_1 (Pisco)	0.4168	$\max(P_{1,t})$/sum maxes	0.2127
γ_2 (Chimbote)	0.1045	$\max(P_{2,t})$/sum maxes	0.1395
γ_3 (Paita)	0.3013	$\max(P_{3,t})$/sum maxes	0.2692
γ_4 (remaining)	0.1775	$\max(P_{4,t})$/sum maxes	0.3785

The estimated distribution for catch/effective stock is below.

E.1.5 Some missing patterns

We failed to find some patterns that might be expected.
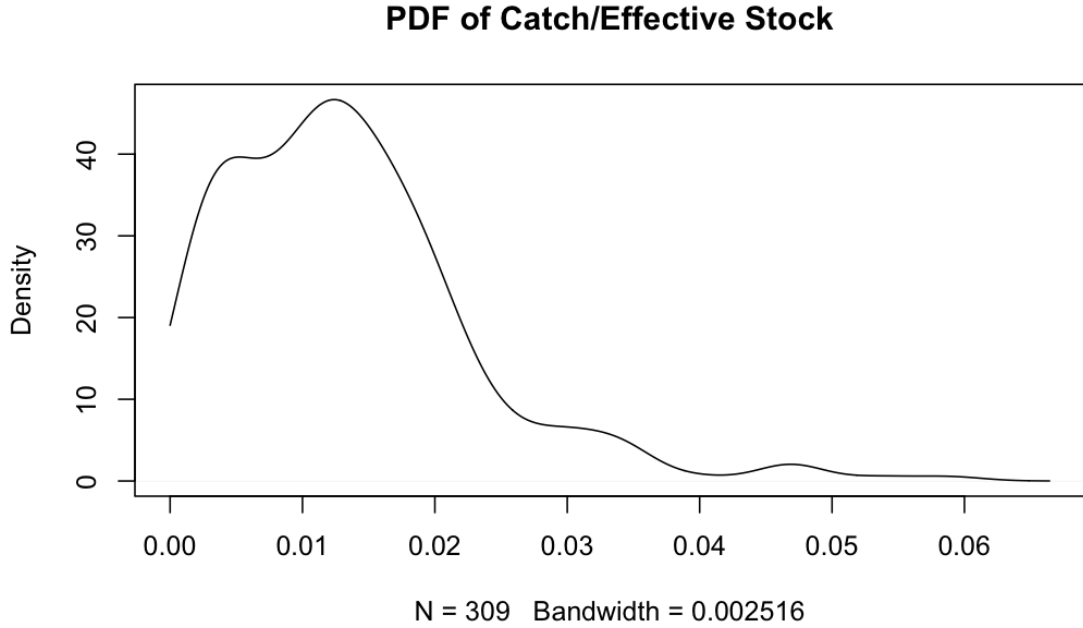
Figure E.6: Distribution of the ratio of catch to effective stock.

- The correlation analysis of plankton with catch shows no pattern and no extended regions of statistically significant correlation. This suggests that fishers do not use plankton as an effective indicator for regions to fish. Based on the results above, the low correlation of plankton with current fish stocks suggests that this is reasonable.

- Catches are not well-predicted by NINO3 or any combinations of it delayed. SSA applied to yearly catches identified no cycles of the relevant length:
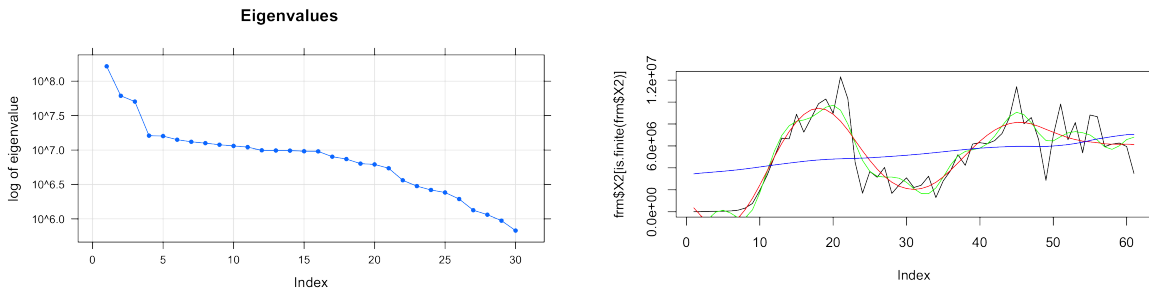
Figure E.7: Singular spectrum analysis of anchoveta catches.

An AIC model selection from of two years of delayed monthly NINO3 values, linearly predicting monthly catches, identified eight delays which combined produce a R2 of only .056. The graph below shows the model parameter estimates for all delays, and identifies (dots) the delays selected by the AIC criteria.
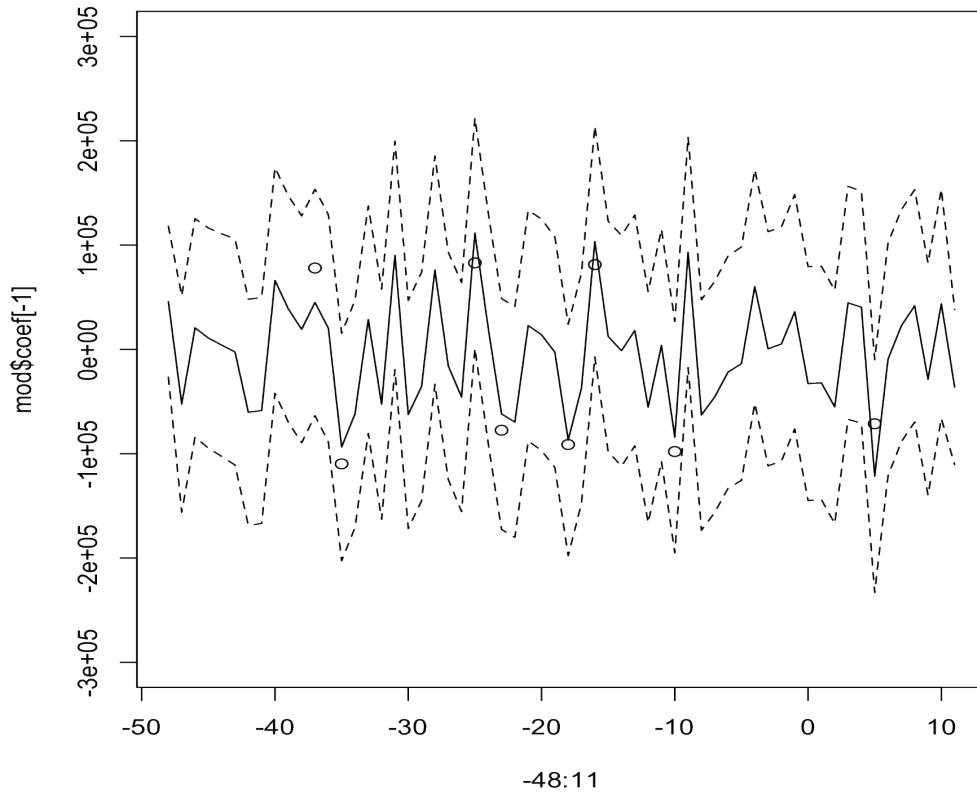
Figure E.8: Correlations between catches and NINO 3, showing no relationship.

Appendix F

Global benefits of marine protection supplement

F.1   Syntax for geocoding

Stock assessments in the RAM database are associated with named areas. We estimated coordinates and bounding boxes for each of these areas, using country EEZs and fishing area shapefiles when appropriate. In addition, we developed a simple language to encode the GIS shapes of the areas, along with an interpreter to translate these codes into polygons. The syntax supports using political entities, shapefile regions, circles and rectangles, clipped versions of these, and combinations of these.

The various cases handled by the syntax are shown below:

Region Type	Syntax and Example
Region Type	**Syntax and Example**
FAO regions	Space-delimited sequence of FAO fishing subareas or major areas followed by ∗
Example:	Labrador - NE Newfoundland (Canada-DFO-23K)
	`2G 2H 2J 3K`
Example:	Indian Ocean (multinational-IOTC-IO)
	`51* 57*`
FAO Regions Restricted	Major area number followed by `:` for subareas or `::` for divisions
Example:	Eastern Baltic (multinational-ICES-25-32)
	`27: 25 26 27 28.1 28.2 29 30 31 32`
Example:	Baltic Sea (multinational-ICES-IIId)

Region Type	Syntax and Example
	`27::27.III.d`
Shapefile regions	Shapefile or region designator, followed by [sub]region names
Example:	New Zealand Areas LIN 6b (New Zealand-MFish-LIN6b)
	`NZ: 607 608 609 613 614 615 621 622`
Circular Disc	Latitude, Longitude, Radius (in km)
Example:	Queen Charlotte Islands (Canada-DFO-QCI)
	`52.683043, -131.791992, 100`
Bounding Boxes	$\pm$Latitude$_1$ $\pm$Longitude$_1$ $\pm$Latitude$_2$ $\pm$Longitude$_2$
Example:	Cascade Plateau (Australia-AFMA-CASCADE)
	`-43.83 +150.38 -44.02 +150.54`
Political Entities	Quoted name of entity, or country followed by `#`
Example:	Macquarie Island (Australia-AFMA-MI)
	`"Australia (Macquarie Island)"`
Clipped Regions	Region designations, followed `\~` for NE, `~/` for NW, `/~` for SE, `~\` for SW of the following latitude, longitude point
Example:	California (USA-NMFS-CAL)
	`77 \~ 32 -177.714844`
Combinations	Space-separated sequences of above commands
Example:	South Africa South coast (South Africa-DETMCM-SASC)
	`47: 1.6 2.1 2.2 51: 8 ~\ -25 +36`

The following table shows all RAM database regions as coded, as well as their calculated size and centroid.

Region Name	Encoding	Size (km^2)	Latitude	Longitude
Argentina-CFP-ARG-N	41: 2.3	979766	-36.33	-56.93
Argentina-CFP-ARG-S	41: 3.1 3.2	2924714	-50.00	-59.28
Australia-AFMA-CASCADE	-43.83 +150.38 -44.02 +150.54	271	-43.92	150.46
Australia-AFMA-ESE	-37.71859, 150.556641, 600	894716	-37.72	150.56
Australia-AFMA-MI	-54.72305, 158.86208, 370	247031	-54.72	158.86
Australia-AFMA-GAB	-35.746512, 131.308594, 1000	2557954	-35.75	131.31
Australia-AFMA-GAB-SESSF	-35.746512, 131.308594, 1000	2557954	-35.75	131.31
Australia-AFMA-NAUST	57: 5.1 71:_71.5 71:_71.8	8355535	-17.24	133.18
Australia-AFMA-SESSF	6:57.6 81:_81.1	3041179	-35.27	157.28
Australia-AFMA-TAS	-42.032974, 146.601563, 500	581887	-42.03	146.60
Australia-AFMA-SE	Australia# /~ -24.5 +129 Australia:Lord_Howe_Island#	5430238	-34.03	147.02
Australia-AFMA-WSE	-36.879621, 138.691406, 700	1232722	-36.88	138.69
Canada-DFO-23K	2G 2H 2J 3K	1228864	54.40	-54.66
Canada-DFO-2J3KL	2J 3K 3L	1004991	51.43	-51.89
Canada-DFO-2J3KLNOPs	2J 3K 3L 3N 3O 3Ps	1556983	48.97	-51.74

Canada-DFO-3Pn4RS	3Pn 4R 4S	505847	50.38	-65.12
Canada-DFO-3Pn4RSTVn	3Pn 4R 4S 4T 4Vn	913775	49.10	-65.88
Canada-DFO-3Ps	3Ps	125784	46.50	-55.89
Canada-DFO-4R	4R	70600	49.81	-58.02
Canada-DFO-4RST	4R 4S 4T	858583	49.22	-66.22
Canada-DFO-4T	4T	364842	47.51	-67.57
Canada-DFO-4TVn	4T 4Vn	399488	47.41	-66.87
Canada-DFO-4Vn	4Vn	35668	46.39	-59.58
Canada-DFO-4VsW	4Vs 4W	570694	41.86	-58.22
Canada-DFO-4VWX	4Vn 4Vs 4W 4X	816503	42.41	-60.11
Canada-DFO-4VWX5	4Vn 4Vs 4W 4X 5Y 5Zc 5Zu 5Zw	1076868	42.27	-62.39
Canada-DFO-4VWX5Zc	4Vn 4Vs 4W 4X 5Y 5Zc	944464	42.56	-61.27
Canada-DFO-4X	4X	218106	43.14	-65.16
Canada-DFO-4X5Y	4X 5Y	329151	43.36	-66.54
Canada-DFO-4X5YZ	4X 5Y 5Zc 5Zu 5Zw	511800	42.41	-67.30
Canada-DFO-5Zejm	40.5, -68, 200	92253	40.50	-68.00
Canada-DFO-5Zjm	40.5, -68, 200	92253	40.50	-68.00
Canada-DFO-ATL	2G 2H 2J 3K 3L 3M 3N 3O 3Ps 4Vn 4Vs 4W 4X	3171572	48.34	-53.94
Canada-DFO-CC	52.375599, -129.023437, 200	74251	52.38	-129.02
Canada-DFO-HS	53.120405, -130.825195, 150	39969	53.12	-130.83
Canada-DFO-PCOAST	Canada# ~\ 57 -120	1114534	52.18	-126.97
Canada-DFO-PRD	53.120405, -130.825195, 250	115649	53.12	-130.83
Canada-DFO-QCI	52.683043, -131.791992, 100	17952	52.68	-131.79
Canada-DFO-SOG	49.353756, -123.859863, 100	19260	49.35	-123.86
Canada-DFO-WCVANI	49.639177, -126.694336, 200	78771	49.64	-126.69
Iran-Iran-CS	+47.115 +46.625977 +36.244273 +55.480957	885906	41.68	51.05
multinational-CCAMLR-RS	-75.050354, -174.726562, 600	10060140	-75.05	-174.01
multinational-CCSBT-SO	88* 48* 58*	46033059	-70.49	-0.85
multinational-GFCMED-BLACKW	42.747012, 37.880859, 400	366951	42.75	37.88
multinational-IATTC-EPAC	61 71 81	35180027646	3.56	71.62
multinational-IATTC-NEPAC	61	8589330178	42.74	120.79
multinational-ICCAT-NATL	21* 27* 31* 34*	56168545	49.41	-22.72
multinational-ICCAT-WATL	21* 31* 41*	4730204	5.19	-50.66
multinational-ICCAT-MED	37: 1.1 1.2 1.3 2.1 2.2 3.1 3.2	5423167	37.04	15.86
multinational-ICCAT-EATL	27* 34* 47*	38136898	33.96	-1.26
multinational-ICCAT-SATL	41* 47*	23416509	-30.52	-24.68
multinational-ICES-22-24	22 23 24	98290	54.75	12.06
multinational-ICES-22-24-IIIa	22 23 24 IIIa	239628	57.04	10.99
multinational-ICES-22-32	27: 22 23 24 25 26 27 28.1 28.2 29 30 31 32	1104398	61.16	21.50
multinational-ICES-25-32	27: 25 26 27 28.1 28.2 29 30 31 32	1002955	61.68	22.26
multinational-ICES-28	28.1 28.2	84998	57.46	21.71
multinational-ICES-29	29	73469	59.63	20.42
multinational-ICES-30	30	234862	62.17	23.00
multinational-ICES-31	27: 31	295786	65.91	23.16
multinational-ICES-32	32	115565	59.92	26.76
multinational-ICES-I	Ia Ib	2566804	77.14	46.78
multinational-ICES-IIa	IIa1 IIa2	1824009	67.90	6.29
multinational-ICES-IIa-IIIabd-IV-Vb-VI-VII-VIIIabcde-XII-XIV-Ixa	IIa1 IIa2 IIIa 23 27::27.III.d IVa IVb IVc Vb1a Vb1b Vb2 VIa VIb1 VIb2 VIIa VIIb VIIc1 VIIc2 VIId VIIe VIIf VIIg VIIh VIIj1 VIIj2 VIIk1 VIIk2 VIIIa VIIIb VIIIc VIIId1 VIIId2 VIIIe1 VIIIe2 XIIa1 XIIa2 XIIa3 XIIa4 XIIb XIIc XIVa XIVb1 XIVb2	11045306	65.32	-12.56
multinational-ICES-IIb	IIb1 IIb2	1180643	81.75	9.50
multinational-ICES-I-II	Ia Ib IIa1 IIa2 IIb1 IIb2	5629995	76.92	26.36
multinational-ICES-IIIa	IIIa	141326	58.49	10.31
multinational-ICES-IIIa-IV	IIIa IVa IVb IVc	1066012	56.84	3.91
multinational-ICES-IIIa-IV-VI	IIIa IVa IVb IVc VIa VIb1 VIb2	1580319	56.98	-1.01
multinational-ICES-IIIa-IV-VIId	IIIa IVa IVb IVc VIId	1137714	56.47	3.67

multinational-ICES-IIIa-IV-VI-VII-VIIIabd	IIIa IVa IVb IVc VIa VIb1 VIb2 VIIa VIIb VIIc1 VIIc2 VIId VIIe VIIf VIIg VIIh VIIj1 VIIj2 VIIk1 VIIk2 VIIIa VIIIb VIIId1 VIIId2	2842784	54.14	-3.86
multinational-ICES-IIIb(23)	23	5693	55.73	12.57
multinational-ICES-IIIc(22)	22	47945	54.84	10.62
multinational-ICES-IIId	27::27.III.d	1050714	61.43	21.96
multinational-ICES-II-IIIa-IV-VI-VII-VIIIabc	IIa1 IIa2 IIb1 IIb2 IIIa IVa IVb IVc VIa VIb1 VIb2 VIIa VIIb VIIc1 VIIc2 VIId VIIe VIIf VIIg VIIh VIIj1 VIIj2 VIIk1 VIIk2 VIIIa VIIIb VIIIc	5810170	70.74	5.13
multinational-ICES-I-II-III-IV-V-VI-VII-VIII-IX-XII-XIV	Ia Ib IIa1 IIa2 IIb1 IIb2 IIIa 23 22 27::27.III.d IVa IVb IVc Va1 Va2 Vb1a Vb1b Vb2 VIa VIb1 VIb2 VIIa VIIb VIIc1 VIIc2 VIId VIIe VIIf VIIg VIIh VIIj1 VIIj2 VIIk1 VIIk2 VIIIa VIIIb VIIIc VIIId1 VIIId2 VIIIe1 VIIIe2 IXa IXb1 IXb2 XIIa1 XIIa2 XIIa3 XIIa4 XIIb XIIc XIVa XIVb1 XIVb2	16299447	70.38	5.30
multinational-ICES-IV	IVa IVb IVc	921642	56.58	2.88
multinational-ICES-IVa	IVa	347372	59.77	1.92
multinational-ICES-IVb	IVb	376667	55.40	3.40
multinational-ICES-IVc	IVc	197903	52.19	3.84
multinational-ICES-IXa	IXa	350956	39.41	-8.35
multinational-ICES-IXb	IXb1 IXb2	467680	39.50	-14.50
multinational-ICES-Va	Va1 Va2	485043	65.11	-19.26
multinational-ICES-Vb	Vb1a Vb1b Vb2	193507	61.52	-9.58
multinational-ICES-Vb1	Vb1a Vb1b	173007	61.61	-9.66
multinational-ICES-Vb2	Vb2	20853	60.77	-8.84
multinational-ICES-VIa	VIa	285871	57.29	-8.18
multinational-ICES-VIa-VIIb-VIIc	VIa VIIb VIIc1 VIIc2	427820	56.12	-9.74
multinational-ICES-VIb	VIb1 VIb2	221241	57.25	-15.00
multinational-ICES-VIIa	VIIa	134226	53.46	-4.98
multinational-ICES-VIIb	VIIb	52721	53.54	-10.20
multinational-ICES-VIIb-k	VIIb VIIc1 VIIc2 VIId VIIe VIIf VIIg VIIh VIIj1 VIIj2 VIIk1 VIIk2	773580	50.81	-9.72
multinational-ICES-VIIc	VIIc1 VIIc2	88531	53.50	-15.00
multinational-ICES-VIId	VIId	74116	50.34	-0.25
multinational-ICES-VIIe	VIIe	94165	49.58	-3.36
multinational-ICES-VIIe-k	VIIe VIIf VIIg VIIh VIIj1 VIIj2 VIIk1 VIIk2	557231	50.13	-10.03
multinational-ICES-VIIf	VIIf	34479	51.04	-4.52
multinational-ICES-VIIf-g	VIIf VIIg	85513	51.10	-6.16
multinational-ICES-VIIg	VIIg	50945	51.14	-7.26
multinational-ICES-VIIh	VIIh	57089	48.89	-7.14
multinational-ICES-VIII	VIIIa VIIIb VIIIc VIIId1 VIIId2 VIIIe1 VIIIe2	793200	45.40	-9.07
multinational-ICES-VIIIa	VIIIa	88259	47.15	-3.53
multinational-ICES-VIIIb	VIIIb	65318	44.72	-1.54
multinational-ICES-VIIIc	VIIIc	150420	43.57	-6.21
multinational-ICES-VIIIc-IXa	VIIIc IXa	501146	40.69	-7.69
multinational-ICES-VIIId	VIIId1 VIIId2	181917	45.99	-7.71
multinational-ICES-VIIIe	VIIIe1 VIIIe2	305209	45.50	-14.50
multinational-ICES-VIIj	VIIj1 VIIj2	106942	50.25	-10.50
multinational-ICES-VIIk	VIIk1 VIIk2	213068	50.25	-15.00
multinational-ICES-X	Xa1 Xa2 Xb	2746301	42.00	-30.00
multinational-ICES-XII	XIIa1 XIIa2 XIIa3 XIIa4 XIIb XIIc	2085132	54.29	-29.05
multinational-ICES-XIVa	XIVa	1403557	79.37	-25.20
multinational-ICES-XIVb	XIVb1 XIVb2	982556	64.38	-35.53
multinational-IOTC-IO	51* 57*	27809007	-15.25	80.65
multinational-IPHC-NPAC	+66 +120 +20 +180 +66 -180 +20 -105	3664874159	43.00	-12.50
multinational-NAFO-01ABCDEF	0A 0B 1A 1B 1C 1D 1E 1F	2732505	69.30	-57.46
multinational-NAFO-1	1A 1B 1C 1D 1E 1F	1884144	69.27	-52.54
multinational-NAFO-23K	2G 2H 2J 3K	1228864	54.40	-54.66

multinational-NAFO-23KLMNO	2G 2H 2J 3K 3L 3M 3N 3O	2278723	50.32	-51.88
multinational-NAFO-3L	3L	208594	47.63	-50.35
multinational-NAFO-3LN	3L 3N	492042	44.81	-49.45
multinational-NAFO-3LNO	3L 3N 3O	633125	44.56	-50.13
multinational-NAFO-3M	3M	409464	44.12	-44.25
multinational-NAFO-3NO	3N 3O	426476	42.92	-50.01
multinational-NAFO-3O	3O	140834	43.66	-52.55
multinational-SPC-WPO	67 77 87*	12970145	2.70	-124.65
multinational-SPRFMO-CH	87* /˜ -18.33 -90	8150045	-38.87	-79.31
multinational-TRAC-5Z	5Zu 5Zc 5Zw	185058	40.67	-68.69
multinational-UNKNOWN-NWPAC	+66 -180 +20 -105	29420870	43.00	-142.50
multinational-WCPFC-SPAC	81 87* /˜ +25 -120	7308057451	-35.02	-70.33
New Zealand-MFish-8	NZ: 037 039 040 041 801	67280	-39.17	173.30
New Zealand-MFish-CR	NZ: 020 021 022 023 401 402 403 404 405 406 407 408 409 410 411 412 049 050 051 052	879324881	-44.09	-26.98
New Zealand-MFish-CRA1	NZ: 046 102 \˜ -36 -165 047 048 103 104 105 001 002 003 106	399295	-33.63	172.65
New Zealand-MFish-CRA2	NZ: 004 005 006 007 008 009 009H 010 011 107 201	26699684693	-36.10	95.66
New Zealand-MFish-CRA3	NZ: 012 013 202 203 204 205 402 403	2303671631	-40.32	6.76
New Zealand-MFish-CRA4	NZ: 014 015 016 017 039 019 401	74563	-41.66	176.21
New Zealand-MFish-CRA5	NZ: 018 020 021 022 023 407 301	142860	-44.38	174.66
New Zealand-MFish-CRA7	NZ: 024 026 302 303	84814	-46.55	171.90
New Zealand-MFish-CRA8	NZ: 027 028 029 030 031 032 033 501 502 503 504 601 602 603 610 616 617 618 619 623 624 625 705 706	941730	-49.64	166.95
New Zealand-MFish-CIR	NZ: 618 619 620 624 625	227989	-53.16	171.32
New Zealand-MFish-ENZ	-38.203655, 176.044922, 600	45518619	-38.14	151.71
New Zealand-MFish-LIN3-4	-42.391009 +171.386719 -45.39845 +180 -42.391009 -180 -45.39845 -172.441406	275920239	-43.89	11.21
New Zealand-MFish-LIN5-6	NZ: 501 502 503 504 032 031 030 029 028 027 025 601 602 603 604 605 606 607 608 609 610 611 612 613 614 615 616 617 618 619 620 621 622 623 624 625	28354047681	-49.90	113.86
New Zealand-MFish-LIN6b	NZ: 607 608 609 613 614 615 621 622	815277624	-49.07	-5.69
New Zealand-MFish-LIN72	NZ: 701 702 703 704 705 706 801 041 040 039 038 037 036 035 034 033 201 202 203 204 205 206 011 012 013 014 015 016 017	45764801564	-40.11	71.68
New Zealand-MFish-LIN7WC-WCSI	NZ: 704 705 706 035 034 033	203793	-42.51	167.45
New Zealand-MFish-NZMEC	NZ: 014 015 016 017 018 019 203 204 205 206	44232957484	-40.89	-14.94
New Zealand-MFish-PAU5A	NZ: 030 031 032	54984	-45.68	166.37
New Zealand-MFish-PAU5B	NZ: 025 027 029 030	59787	-47.11	167.39
New Zealand-MFish-PAU5D	NZ: 024 026	31421	-45.99	171.20
New Zealand-MFish-PAU7	NZ: 036 /˜ -40.5 -171.5 038 017 018	24269	-41.62	173.67
New Zealand-MFish-NZ	7	4094877028	-35.70	32.11
New Zealand-MFish-SA	New_Zealand# ˜\ -46 +200	1665563263	-49.70	116.74
New Zealand-MFish-TRE7	NZ: 701 702 703 704 705 706 036 035 034 033	309307	-41.56	168.36
New Zealand-MFish-WECR	NZ: 020 021 022 023 401 402 403 407 408 409	5463743046	-44.09	150.18
New Zealand-MFish-WNZ	-44.087585, 169.453125, 600	812793	-44.09	169.45
Peru-IMARPE-NC	Peru# ˜/ -13 -76	2592119	-6.11	-79.14
Russia-RFFA-NSO	59.534318, 149.414063, 700	783976	59.53	149.41
Russia-RFFA-WBS	58.813742, 174.726563, 800	1541792552	58.81	146.09
South Africa-DETMCM-1-2	-28.5 +16 -31.25 +18	58895	-29.88	17.00
South Africa-DETMCM-3-4	-31.25 +16 -32.5 +18.5	32801	-31.88	17.25
South Africa-DETMCM-5-6	-32.5 +16 -33.25 +18.5	19468	-32.88	17.25
South Africa-DETMCM-7	-33.25 +16 -33.6 +18.5	9029	-33.42	17.25
South Africa-DETMCM-8	-34.05 +16 -36 +19.5	69157	-35.02	17.75
South Africa-DETMCM-PEI	South_Africa:Marion_Island#	120905	-46.88	37.20
South Africa-DETMCM-SA	10	3259438	-29.62	25.21
South Africa-DETMCM-SASC	47: 1.6 2.1 2.2 51: 8 ˜\ -25 +36	2389559	-34.33	27.53
USA-NMFS-5Y	5Y	111362	43.78	-69.21

USA-NMFS-5YCHATT	5Y 6C	302113	39.12	-72.47
USA-NMFS-5YZ	5Y 5Zc 5Zu 5Zw	296009	41.87	-68.89
USA-NMFS-5YZSNE	5Y 5Zc 5Zu 5Zw	296009	41.87	-68.89
USA-NMFS-5Z	5Zc 5Zu 5Zw	185058	40.67	-68.69
USA-NMFS-5ZSNE	5Zc 5Zu 5Zw	185058	40.67	-68.69
USA-NMFS-AI	+52 -175 +55 -160	331067	53.50	-167.50
USA-NMFS-AIES	+52 -167.5 +55 -160	165872	53.50	-163.75
USA-NMFS-AIWS	+52 -175 +55 -167.5	165845	53.50	-171.25
USA-NMFS-ATL	5Y 5Zc 5Zu 5Zw 6A 6B 6C 6D 6E 6F 6G 6H 31:_31.1 31:_31.5	4001904	34.74	-65.83
USA-NMFS-ATLC	5Y 5Zw 6A 6B 6C 31:_31.1	1237104	35.70	-75.55
USA-NMFS-BB	52.05249, -173.496094, 200	74842	52.05	-173.50
USA-NMFS-BS	57.657158, -178.242187, 1000	805634758	57.65	-45.42
USA-NMFS-BSAI	57.657158, -178.242187, 1200	1153939761	57.64	-36.03
USA-NMFS-CAL	77 \˜ 32 -130	1426282	35.11	-118.35
USA-NMFS-CBS	57.657158, -178.242187, 500	7143841	57.66	-85.62
USA-NMFS-CCOD5Y	5Y	111362	43.78	-69.21
USA-NMFS-CWPAC	77 \˜ 36 -130	428462	38.25	-125.03
USA-NMFS-EBSAIGA	67 ˜/ +50 -130	4196827	57.87	-152.56
USA-NMFS-EBSAI	67 ˜/ +50 -160	1539692	57.92	-167.54
USA-NMFS-EBS	67 ˜/ +54.5 -160	1032738	60.17	-167.55
USA-NMFS-EBSGA	67 ˜/ +54.5 -130	2710608	60.12	-152.58
USA-NMFS-EGM	31::_31.2.2 31::_31.2.3 31::_31.2.4 31::_31.2.5 31::_31.2.6	1009201	27.54	-87.61
USA-NMFS-GA	57.984808, -146.689453, 700	819151	57.98	-146.69
USA-NMFS-GM	31::_31.2.1 31::_31.2.2 31::_31.2.3 31::_31.2.4 31::_31.2.5 31::_31.2.6 31::_31.3.1 31::_31.3.2 31::_31.3.3	2496985	24.95	-91.62
USA-NMFS-GOMNGB	5Y	111362	43.78	-69.21
USA-NMFS-MATLC	6A 6B 6C	496532	38.02	-74.36
USA-NMFS-NATL	5Y 5Zu 5Zw	282174	41.88	-69.03
USA-NMFS-NCAL	+42 -125 +37.07271 -121.311035	173215	39.54	-123.16
USA-NMFS-NPAC	67	10811935	52.91	-147.62
USA-NMFS-NPCOAST	67 /˜ +55 -130	1206543	47.75	-125.00
USA-NMFS-NWATL	1F 2G 2H 2J 3K 3L 3M 3N 3O 3Pn 3Ps 4R 4S 4T 4Vn 4Vs 4X 4W 5Y 5Zc 5Zu 5Zw 6A 6B 6C 6D 6E 6F 6G 6H	6290311	46.62	-57.41
USA-NMFS-NWATLC	2G 2H 2J 3K 3L 3Pn 3Ps 4R 4S 4T 4Vn 4X 4W 5Y 5Zu 5Zw 6A 6B 6C	3815571	48.47	-61.32
USA-NMFS-NS	63.849, -164.269, 150	29408	63.85	-164.27
USA-NMFS-ORECOAST	+46.195042 -125.408936 42 -123.401184	74853	44.10	-124.41
USA-NMFS-PCOAST	67 /˜ +55 -130 77 \˜ +30 -130	3246712	39.54	-120.48
USA-NMFS-PI	56.824933, -169.980469, 300	153069	56.82	-169.98
USA-NMFS-SATL	31:_31.1	564206	31.09	-78.64
USA-NMFS-SATLC	31:_31.1 ˜/ +32 -75 31:_31.1 ˜\ +32 -79	392932	31.55	-79.34
USA-NMFS-SATLCGM	31:_31.1 ˜/ +32 -75 31:_31.1 ˜\ +32 -79 31::_31.2.1 31::_31.2.2 31::_31.2.3 31::_31.2.4 31::_31.2.5 31::_31.2.6 31::_31.3.1 31::_31.3.2 31::_31.3.3	2894673	25.90	-89.85
USA-NMFS-SCAL	+35 -125 +32.5 -117	205759	33.75	-121.00
USA-NMFS-SGBMATL	5Zu 6D	317858	38.39	-67.71
USA-NMFS-SMI	60.408611, -172.72, 400	247659	60.41	-172.72
USA-NMFS-SNE	+43 -74 +41.5 -68	82498	42.25	-71.00
USA-NMFS-SNEMATL	5Zc 5Zw 5Zu 6A 6B 6C 6D 6E 6F 6G 6H	1700861	37.68	-62.50
USA-NMFS-SNEMATLB	5Zc 5Zw 5Zu 6A 6B 6C	685280	38.76	-72.78
USA-NMFS-SPCOAST	+35 -125 +32.5 -117	205759	33.75	-121.00
USA-NMFS-WATL	4Vs 4X 4W 5Y 5Zc 5Zw 5Zu 6A 6B 6C 6D 6E 6F 6G 31:_31.1 31:_31.5 31:_31.4	8685164	29.67	-67.18
USA-NMFS-WGM	31::_31.2.1 31::_31.2.2 31::_31.2.5 ˜/ 0 -90 31::_31.3.2	1147829	24.97	-93.16

USA-US State-ATKINS	33.5984422, -92.0468047, 0.2	0	33.60	-92.05
USA-US State-HUNT	44.860052, -84.153192, 0.5	1	44.86	-84.15
USA-US State-KAB	48.472921, -92.991028, 10	196	48.47	-92.99
USA-US State-MIN	47.36935, -122.697458, 1.2	3	47.37	-122.70
USA-US State-NIMROD	34.949554, -93.218307, 4	39	34.95	-93.22
USA-US State-OKA	32.514394, -88.80249, 5	62	32.51	-88.80
USA-US State-PWS	60.615, -147.168, 100	14542	60.62	-147.17
USA-US State-RI	41.664705, -71.499023, 50	5518	41.66	-71.50
USA-US State-ROSS	32.479646, -90.009613, 14	488	32.48	-90.01
USA-US State-SHAL	47.6378593, -124.2526924, 8	127	47.64	-124.25
USA-US State-SITKA	57.053, -135.33, 100	16107	57.05	-135.33
USA-US State-SKY	47.755021, -121.46347, 2	8	47.76	-121.46
USA-US State-SNAH	47.761389, -124.133333, 5	50	47.76	-124.13
USA-US State-SNOW	48.345411, -124.552517, 1	2	48.35	-124.55
USA-US State-SPRI	42.674863, -121.891766, 1	2	42.67	-121.89
USA-US State-TAYU	47.45, -124, 8	128	47.45	-124.00
USA-US State-WILD	47.601996, -124.285583, 6	72	47.60	-124.29

F.2 Treatment and control matching

It is difficult to create matching pairs of "treated" stocks that have had increases in their protection a "control" stocks that have not. Four factors have considerable importance for matching stocks for comparison: their stock sizes and their surpluses during a "baseline" period, their locations, and the size of their assessment regions. The baseline is determined by the treated stock, and includes all of the years in which the protected area portion did not increase significantly (as described in section 6.2).

We construct four measures of how well each of these features matches between any two stocks, and treat the final match as a product of the four test measures. For stock size and year-to-year surpluses, we construct distributions for each assessment in the "baseline" years before the increase in protected region. We use the Mann-Whitney Wilcoxon tests to compare these distributions, and take the logarithm to ensure that the results of this test do not always dominate the other factors. The distributions of stock and year-to-year surpluses reflect the bioeconomic dynamic of each stock: some stocks are very stable due to both their biology and management, while other stocks are highly variable.

The four features measures are shown below, along with the 50-percentile, 75-percentile, and maximum feature comparison value between any pair of treatment and control regions.

Feature	Test Measures
Stock Distribution	$(-\log WilcoxonTestP - value(BaselineStocksA, BaselineStocksB))^{-1}$
Surplus Distribution	$(-\log WilcoxonTestP - value(Base.SurplusesA, Base.SurplusesB))^{-1}$
Location	$10/\left(10 + \sqrt{(LatitudeA - LatitudeB)^2 + (LongitudeA - LongitudeB)^2}\right)$
Region Size	$Min(RegionSizeA, RegionSizeB)/Max(RegionSizeA, RegionSizeB)$

Feature	50%	75%	100%
Stock Distribution	0.02694446	0.06184597	272.07847106
Surplus Distribution	0.03735511	0.14421754	139.66341320
Location	0.04216678	0.09633013	0.74351009
Region Size	0.0002629865	0.1197303684	0.9955717441

To match treated and control regions, we find the optimal match under which if a treatment region A is not matched to a control region B, it must be because B is more closely matched to a different control region (the Stable Marriage solution). The matched regions are shown in figure F.1 for two cases: when only the stock distribution is used, and

when all four features are used.
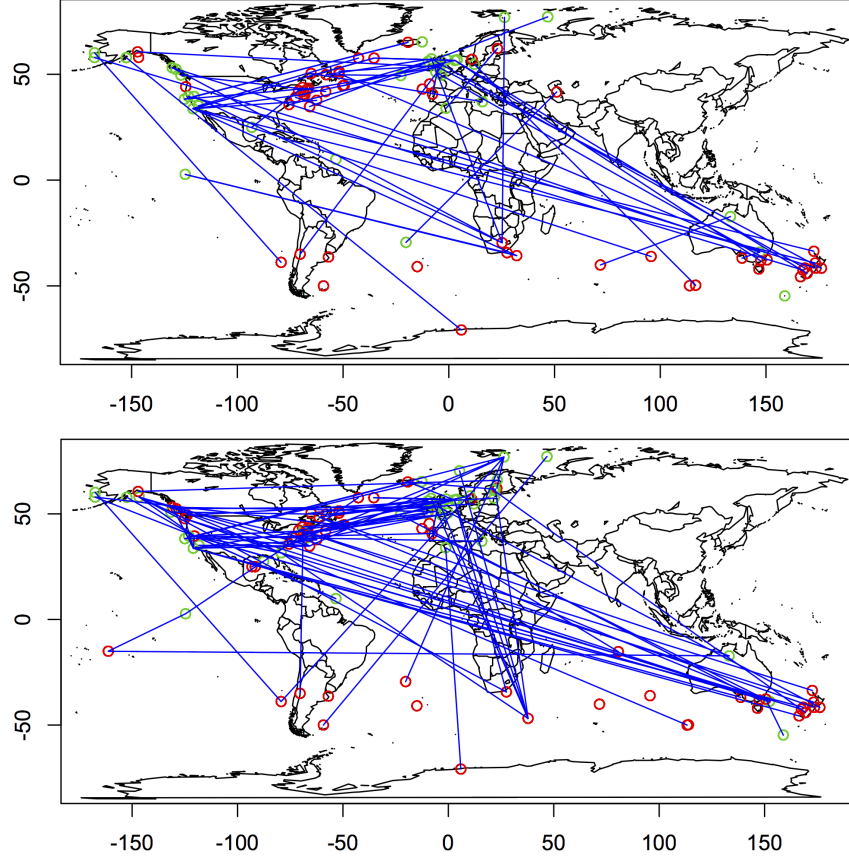
Figure F.1: Treatment (green) and control (red) assessment regions, and their pairing (blue). **Left** shows the matches when only the stock distributions are used, **right** shows the matches when all four features are used.

The results from this analysis are unstable, so we provide a collection of test combinations below. The control and treatment change measured relative to each assessments own baseline.

	N	Control Base.	Treated Base.	Control Chg.	Treated Chg.	P Value
stock	65	72119.66	92111.7	1.016591	1.023715	0.9948
stock, dynamics	65	97817.81	90624.1	-36.749505	1.147481	0.3305
stock, dynamics, dist.	63	98041.74	90837.06	1.367975	1.143872	0.7749
stock, dynamics, size	57	95000.75	89967.56	0.01151843	1.02446723	0.627
all	55	106892.4	67605.45	1.9238180	0.9740837	0.2367

In most cases, the treated stock change exceeds the control stock change, but under none is it significant.

F.2.1   Difference-in-difference checks

The difference-in-difference results are not very robust to differences in the way that differences are normalized, and this is reflected in the robust standard errors clustered at the assessment level: t-values for all three parameters are between 1 and 1.4. Residuals are not normally distributed (Shapiro-Wilks test), however the linearity assumption supported (Harvey-Collier test).

F.3 Catch regressions

Regressions between country catches and country MPAs show positive marginal effects (see table F.3), while regressions between country stock growth and country MPAs show negative effects (see table F.4). Combined with the observation that regressions between stock growth and local MPAs show positive effects, this suggests the following consequences:

- The effects of MPAs are local, so country averages are misleading.

- MPA designation corresponds to increases in exploitation. Stocks have decreased while MPAs have increased.

$$C_{it} = \delta + \beta MPA_{it}$$

$$C_{it} = \delta_{0i} + \delta_{1i}t + \delta_{2i}t^2 + \delta_{3i}t^3 + \delta_{4i}t^4 + \beta MPA_{it}$$

	Totals Only	All Regions	Country P(4)
(Intercept)	204119.15***	163321.54***	
	(7498.53)	(10156.04)	
Total	2.72***	14.36***	1.03***
	(0.21)	(2.02)	(0.32)
Marine		0.26	
		(1.15)	
No Take		−3.52	
		(2.40)	
R^2	0.02	0.03	0.93
Adj. R^2	0.02	0.02	0.92
Num. obs.	9405	2166	9405

***$p < 0.01$, **$p < 0.05$, *$p < 0.1$

Table F.3: Regressions between total MPA area and catch. The first column uses the equation at top. The second includes into this equation total yearly sizes of marine-only and no-take potions of MPAs. The third column uses a country-specific fourth-order polynomial, to capture the rise and fall characteristic of many country's catch series.

	Total	Marine	No Take	Combined
(Intercept)	14789.9231***	13944.1925***	13820.7312***	14965.0093***
	(2042.3211)	(1946.0383)	(1934.4529)	(2102.3685)
Total	−0.0316***			−0.0440
	(0.0117)			(0.0419)
Marine		−0.0311**		0.0300
		(0.0130)		(0.2410)
No Take			−0.0968**	−0.0495
			(0.0417)	(0.6772)
R^2	0.0021	0.0016	0.0015	0.0021
Adj. R^2	0.0018	0.0013	0.0013	0.0012
Num. obs.	3500	3500	3500	3500

$^{***}p < 0.01,\ ^{**}p < 0.05,\ ^{*}p < 0.1$

Table F.4: Simple regressions between country-wide stock surplus and protected areas show negative relationships. This is because the effects of MPAs are largely local, and over this period stocks have diminished despite the increase in MPAs.

F.4 Logistic model checks

AIC criteria produce similar results for the five models shown in table 6.2, with only a significant improvement with the inclusion of temperature, as shown below. Nonlinear models have lower AIC values, except for the model without temperature.

Logistic:	No Temp.	Timeless	Stock Trends	Year FE	Trends-Year
AIC	275121	274522	274182	274546	274225
Nonlinear:	No Temp.	Timeless	Stock Trends	Year FE	Trends-Year
AIC	302265	274415	274034	274446	274069

F.5   IUCN results

IUCN categorization can be used as a indicator of management regime. Of the 4259 MPAs in our data set, 3545 have IUCN classification, as shown in table F.5.

Accounting for the total area designated across all IUCN classified MPAs, the coefficient β is 1.2017 ± 0.3593***, under the time FE model. Accounting only for the marine area, where available, the coefficient is 1.3207 ± 1.4715. The significance decreases considerably due to the half of the MPA observations which are dropped due to missing marine-portion data. A 10% increase in protected area increases the median growth rate by 35%. The results are comparable to the total sample results.

The estimates of effects by IUCN category vary greatly, but estimates for classifications with more data tend to approach the general model. The results suggest that Habitat/Species Management Area are amongst the most effective IUCN classifications. The significance of this result may simply reflect the larger set of available data, representing almost twice the MPAs of any other IUCN category. However, it may also derive from the concerted management practices characterizing IUCN category IV regions. Protected Landscape/Seascape MPAs, which seek to secure the long-term capacity for human interaction, do worst, shown in the negative point estimate and wide variation. The low performance of Strict Nature Reserves is at odds with regression results that include all no-take reserve areas, which are positive, but both are highly insignificant due to the lack of data.

IUCN		N	β Estimate (trends)
Ia	Strict Nature Reserve	430	-1.9900 ± 2.3739
Ib	Wilderness Area	59	(insufficient data)
II	National Park	569	0.3430 ± 0.8792
III	Natural Monument or Feature	130	(insufficient data)
IV	Habitat/Species Management Area	1084	2.5883 ± 0.3576***
V	Protected Landscape/Seascape	611	-5.8342 ± 13.0779
VI	Sustainable use protected area	662	3.0647 ± 3.5272

Table F.5: MPAs with IUCN classification in the MPA Global database. The β Estimate column shows the estimated coefficient β in the stock trends model for total MPA area.

F.6   Case study comparison

We identified eight case-studies which provide information on the effect of marine protection on exploited fisheries. In each case, we identify the portion of a contiguous region which is protected, and the change in catch per unit effort (CPUE) before and after the creation of a marine protected area. Table F.6 describes these studies.

There is no necessary connection between the additional surplus measure used in this paper and CPUE. However, a general correspondence is expected, whereby the same level of effort should catch additional fish to the same extent that additional fish are available for catching.

Region	Area	Portion	CPUE Before	CPUE After	Increase	Pre-Years	Post-Years	Reference
Mombassa Marine Park, Kenya	$5\,km^2$	63 %	20.3 fisher CPUE	42.9 fisher CPUE	111.33 %	1991	1993-1994	McClanahan and Kaunda-Arara (1996)
Georges Bank		25 %	$35.41 \times .372$	$196.03 \times .878$	1244.59%	1994	1998	Murawski et al. (2000)
St. Lucia, Carribean	$0.9\,km^2$	35%			46 % to 90 %	1995-1996	2000-2001	Roberts et al. (2001)*
Nabq		35%			66%			Gell and Roberts (2003)
Apo Island, Philippines	$0.45\,km^2$	10 %	$0.84\,kg/p/hr$	$1.27\,kg/p/hr$	51.19 %	1981-1986	1998-2001	Russ et al. (2004)
Mediterranean	$4\,km^2$	5.09 %	0.01	0.03	200 %	1997	2003-2005	Follesa et al. (2008)
Mombassa Marine Park, Kenya	$6\,km^2$	62.5 %	229 \$/fisher/day	374 \$/fisher/day	63.3 %	1996-2000	2002-2007	McClanahan (2010)
Bolaerne, Norway	$0.7\,km^2$	39.16 %	0.52	1.48	185 %	2006	2010	Moland et al. (2013)**

Table F.6: Eight case-studies of the effects of marine protection on fished regions. **Area** specifies the total area of the marine reserve(s); **Portion** is the percent of a coherent fished area that is protected; **Increase** is the percentage increase in the CPUE in the period after the protection, relative to before the protection; and **Pre-Years** and **Post-Years** are the years used for baseline evaluation and for effect evaluation, respectively.

* Total area of protection estimated from maps.

** Protected region taken as the convex hull of treated and control regions and area estimated from maps; CPUE comparison calculated by the authors.

F.7   Fish distribution maps

	Timeless	Stock Trends	Year FE	Trends and Year
# MPA $S_t D$	-0.0036	-0.0060^*	-0.0038	-0.0062^*
	(0.0029)	(0.0028)	(0.0028)	(0.0028)
Sum MPA^2 / Area$^2 S_t D$	4065.8408	7184.9990	4453.7294	7377.5907
	(3407.5509)	(3818.4495)	(3405.7433)	(3976.5737)
# MPA $\bar{S}D$	0.0016	0.0049	0.0019	0.0051
	(0.0019)	(0.0029)	(0.0019)	(0.0030)
Sum MPA^2 / Area$^2 \bar{S}D$	-2856.3715	-4919.2919	-3110.0414	-5041.1299
	(2406.0669)	(2621.0389)	(2405.5161)	(2730.4827)
# MPA $S_t(1-D)$	-1868.0145	-5797.3068	-2036.3396	-5836.5614
	(4372.4575)	(5925.9287)	(4290.8233)	(5898.3006)
Sum MPA^2 / Area$^2 S_t(1-D)$	0.0000^{***}	0.0000^{***}	0.0000^{***}	0.0000^{***}
	(0.0000)	(0.0000)	(0.0000)	(0.0000)
# MPA $\bar{S}(1-D)$	1520.3748	-1019.7951	1842.3019	-1371.4873
	(3168.6134)	(8334.9102)	(3133.6152)	(8473.0339)
Sum MPA^2 / Area$^2 \bar{S}(1-D)$	0.0000^{***}	0.0000^{***}	0.0000^{***}	0.0000^{***}
	(0.0000)	(0.0000)	(0.0000)	(0.0000)
R^2	0.5161	0.5536	0.5198	0.5571
Adj. R^2	0.4370	0.4631	0.4372	0.4633
Num. obs.	8791	8791	8791	8791

$^{***}p < 0.001$, $^{**}p < 0.01$, $^*p < 0.05$

Table F.7: Coefficient estimates for non-linear, stock-dependent model, with both overlapping (D) areas and remaining ($1-D$) areas.

F.8 Gompertz model

The Gompertz model has also been shown to provide a plausible description of stock dynamics in the absence of age classes and other ecosystem features.

The Gompertz growth model is,

$$S_{it} - S_{i,t-1} + C_{i,t-1} = rS_{i,t-1} \log\left(K/S_{i,t-1}\right)$$

We estimate this using the following reduced form,

$$S_{it} - S_{i,t-1} + C_{i,t-1} = \left(\gamma_i + \beta\frac{MPA_{it}}{Area_i} + \zeta_{1,i}T_{i,t-1} + \zeta_{2,i}T_{i,t-2}\right) S_{i,t-1} + \kappa_i S_{i,t-1} \log S_{i,t-1} + \delta_i + \epsilon_{it}$$

The results are similar to the logistic model results, and are shown in table F.8.

	Timeless	Stock Trends	Year FE	Trends-Year
Data: MPA variables represent Total MPA areas				
β	1.0537	1.5525**	1.1955*	1.5205**
	(0.6593)	(0.5464)	(0.6055)	(0.5490)
R^2	0.5839	0.6291	0.5878	0.6314
Adj. R^2	0.5155	0.5534	0.5170	0.5534
Data: MPA variables represent Marine-only MPA areas				
β	2.0210	1.5245	2.3513*	1.4315
	(1.4454)	(1.8279)	(1.3156)	(1.8166)
R^2	0.5839	0.6279	0.5878	0.6303
Adj. R^2	0.5154	0.5520	0.5169	0.5520
Num. obs.	10093	10093	10093	10093

***$p < 0.001$, **$p < 0.01$, *$p < 0.05$

Table F.8: Estimated marginal change in growth rate for a Gompertz model; the columns are analogous to the Logistic regression model table.

F.9   Growth model

We also consider a model which describes growth and the effect of MPAs in relative terms.

$$log\left(\frac{S_{it} + C_{i,t-1}}{S_{i,t-1}}\right) = \beta \ log\left(\frac{MPA_{it}}{Area_i}\right) + \delta_i + \epsilon_{it}$$

As before, β is the coefficient of interest, and δ_i is an assessment FE. The left-hand side is the log of the fractional growth of the stock. The independent variable is the protected area, divided by the total assessment area. This formulation better handles the long tails in the data, but has less theoretical justification for adding stock-dependent controls.

In the first model, we include only assessment FE. In the remaining models, we modify the model to include the same controls as the logistic model:

$$log\left(\frac{S_{it} + C_{i,t-1}}{S_{i,t-1}}\right) = \beta \ log\left(\frac{MPA_{it}}{Area_i}\right) + \zeta_{1,i}T_{i,t-1} + \zeta_{2,i}T_{i,t-2} + \kappa_i S_{i,t-1} + \delta_i + \epsilon_{it}$$

The coefficient for the time FE model of total MPA area suggests that a doubling of the protected area results in a .25% increase in growth rate. Given the low median portion protected, this is roughly similar to the results above.

	No Controls	Timeless	Stock Trends	Year FE	Trends-Year
Data: MPA variables represent Total MPA areas					
β	0.0058	-0.0095^*	0.0054	0.0037	0.0036
	(0.0034)	(0.0041)	(0.0044)	(0.0042)	(0.0043)
R^2	0.6947	0.8251	0.8608	0.8293	0.8620
Adj. R^2	0.6850	0.8005	0.8354	0.8039	0.8357
Num. obs.	9208	9208	9208	9208	9208
Data: MPA variables represent Marine-only MPA areas					
β	0.0042	-0.0084^*	0.0047	0.0047	0.0029
	(0.0026)	(0.0034)	(0.0048)	(0.0042)	(0.0053)
R^2	0.4751	0.6737	0.7201	0.6808	0.7233
Adj. R^2	0.4575	0.6252	0.6661	0.6303	0.6672
Num. obs.	7633	7633	7633	7633	7633

$^{***}p < 0.001$, $^{**}p < 0.01$, $^*p < 0.05$

Table F.9: Estimated marginal change in growth rate for a fractional growth model; the columns are analogous to the Logistic regression model table.

	Trends and Year
# MPAs	$-2.950\,942 \times 10^{-3}$ $(1.012\,573 \times 10^{-3})^{**}$
Total MPA Area / Assessment Area	$1.752\,632$ $(4.066\,812 \times 10^{-1})^{***}$
Total MPA Area^2 / Assessment Area	$-1.004\,739 \times 10^{-6}$ $(5.507\,698 \times 10^{-7})$
R^2	0.6449
Adj. R^2	0.5696
Num. obs.	10093

$^{***}p < 0.001$, $^{**}p < 0.01$, $^{*}p < 0.05$

Table F.10: Estimate for the quadratic effect of total MPA area.

F.10 Nonlinear benefits

We estimate the following model to study how the marginal effect changes with increasing total protected area.

$$R_{it} = \left(\gamma_i + \beta_0 MPN_{it} + \beta_1 \frac{\sum_j MPA_{ijt}}{Area_i} + \frac{\left(\sum_j MPA_{ijt} \right)^2}{Area_i} + \zeta_{1,i} T_{i,t-1} + \zeta_{2,i} T_{i,t-2} \right) S_{i,t-1} +$$

$$\kappa_i S_{i,t-1}^2 + \theta_i t + \phi_t + \delta_i + \epsilon_{it}$$

In the expression above, the quadratic term is divided by the assessment area, $Area_i$, rather than the square of that area, since this was found to provide a better fit. The interpretation is that the effect decreases with increasing total square kilometers of MPA, rather than increasing portion of an assessment area protected.

The coefficient on the quadratic term is significant at a 10% level. Across the observations, the median protected area is $3600 km^2$, decreasing the marginal benefit of a portion of area protected from a growth rate increase of 1.753 to a growth rate increase of 1.749. Of the observations, 1.7 % have decreasing marginal benefits, from having protected regions greater than $872\,000\, km^2$. However, no observations exceed the level at which total benefits would be negative (the largest MPA area at $1\,560\,000\, km^2$ is estimated by this model as having a positive benefit from MPAs of a 0.19 increased growth rate per portion protected.

F.10.1 Colinearity of variables

See figure F.2. Because of the close colinearity between $\frac{\sum_j MPA_{ijt}}{Area_i}$ and $\frac{\sum_j MPA_{ijt}^2}{Area_i^2}$ we only include the quadratic term in the preferred model.
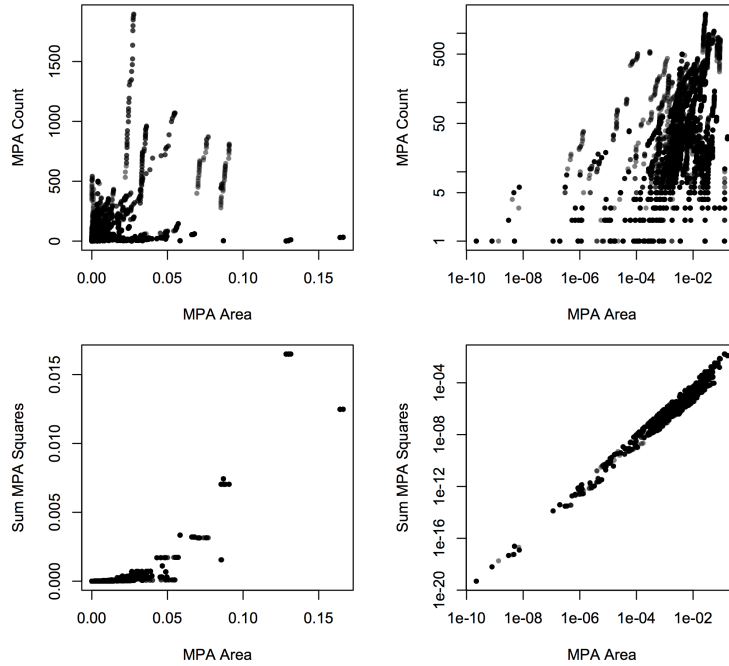
Figure F.2: Relationship between total MPA area, as a fraction of assessed area, and MPA count and sum of squared MPA sizes. Each year is a distinct observation, so regions that have a long-running campaign to designate small MPAs show up as vertical streaks in the top-left corner. As shown in the bottom-right corner, the MPA area and sum of squares of MPA areas are nearly colinear. Points show up as grey if they only represent a single year.

F.10.2 Comparison to MPA effect

A direct comparison between the linear MPA effect and the quadratic effect is difficult, due to the wide range in the numbers of MPAs that represent a given protected area portion. As an example, we take a region which achieves 10% protection with 16 MPAs, as compared in figure F.3.

The example uses a linear increase in MPA count, as a function of the portion of area protected. The quadratic variable, the sum of squared areas divided by the assessment area, is also a linear function portion of area protected, fit in log-log space. The stock size, as

a fraction of mean stock size, is 0.77, the median value of the values across assessments in their most recent recorded year.
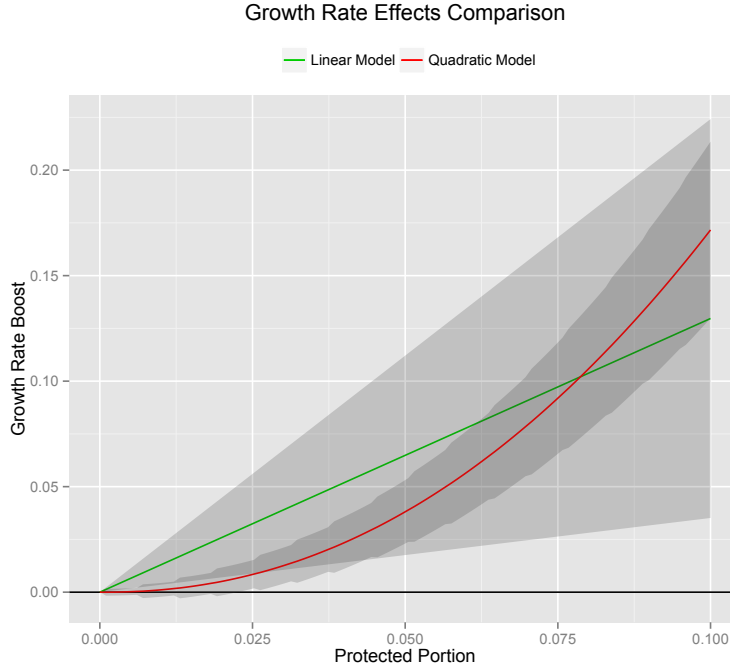
Figure F.3: The linear model (green) and quadratic model (red) for a region that grows from 0 to 16 MPAs as the area protected increases from 0 to 10%.

In the result, the quadratic model measures a smaller effect than the linear model, eventually surpassing around 8% protection. For examples with smaller average MPA sizes, or with larger relative stock sizes, the quadratic model would predict a lower effect than the linear model.

F.10.3   MPA size distribution comparison

The relationship between the total protected area and the sum of squared areas can be very different for different regions, depending on the distribution of MPA sizes. This distribution has two extremes, when every MPA is equally sized and where one MPA is large and the remaining are infinitesimally small. If we define the total portion of protected area as $\rho = \frac{\sum MPA_j}{Area}$, then $\frac{\sum MPA_j^2}{Area^2} = \frac{\rho^2}{N}$ for equal sizing and $\frac{\sum MPA_j^2}{Area^2} = \rho^2$ for extreme sizing.

A summary metric that allows us to explore the range between these extremes is τ, the

ratio between the largest MPA and the second-largest, where MPA sizes are assumed to follow a geometric progression.

Figure F.4 shows the growth rate boost for τ at 0 (extreme sizes), 0.5, and 1 (even sizes). Figure F.5 shows observed values of τ in our dataset.
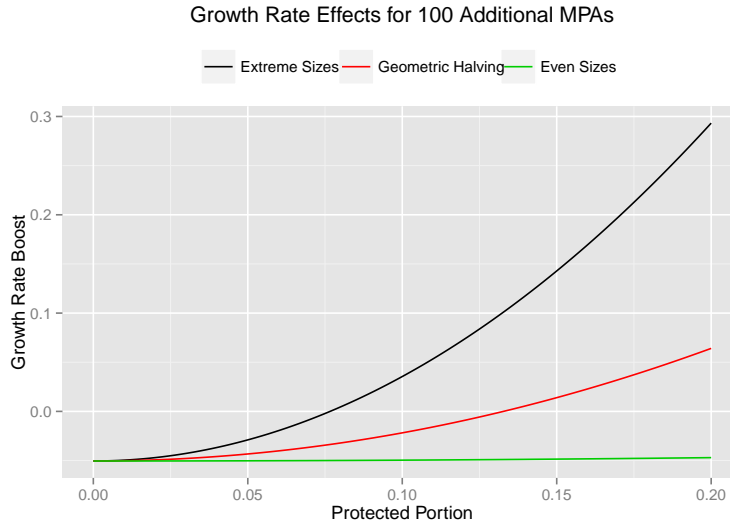
Figure F.4: The effect of size distributions on growth rate effects. The largest effect occurs where one MPA is large and the remaining are infinitesimally small, while very little effect is observed for evenly sized MPAs, since 100 evenly sized MPAs each cover only a small area.

Histogram of MPA Size Distributions
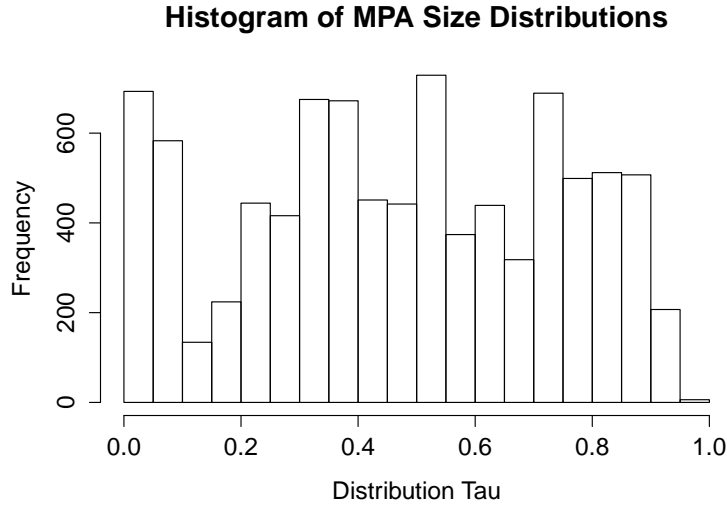
Figure F.5: Observed MPA size distributions, as a histogram of approximate values for τ. The distribution is fairly even, suggesting a wide range of MPA distributions.

F.11   Estimated economic surplus

Figure F.6 shows the estimated cost and benefit of marine protected areas by country, where the two tend two correlate.

Below are the estimated break-even points for economic gain, comparing the additional surplus due to protecting a given area against the cost of maintaining the area's protection.

Country	EEZ (km^2)	Landed Value ($)	Breakeven	Current	Cost ($)	Benefit ($)	Status[1]	Status[2]
Russia	8095881	4539086	0.24962	0.04146	16427766	317282	Loss	Loss
Australia	6362934	543693004	0.00059	0.08219	159877643	75337829	Gain	Loss
Indonesia	6079377	954862067	0.00029	0.01371	13043852	22070873	Gain	Gain
Canada	6006154	2737412010	0.00008	0.04152	178484288	191626682	Gain	Gain
France (French Polyn ...	4767242	86125455	0.00552	0.00003	471751	4429	Loss	Loss
Japan	4469020	1815410	0.67649	0.00259	56975600	7923	Loss	Loss
Alaska (USA)	3770021	1397534333	0.00016	0.05939	25771694	139942313	Gain	Gain
Kiribati	3437345	236619169	0.00144	0.11954	444053	47688191	Gain	Gain
New Zealand	3423231	643402312	0.00041	0.00154	8696561	1673562	Gain	Loss
Mexico	3269386	1007118228	0.00023	0.02436	20225148	41365638	Gain	Gain
Brazil	3179693	979543861	0.00024	0.01293	30578527	21351922	Gain	Loss
Micronesia	2992597	158507960	0.00229	0.00001	5645	2449	Loss	Loss
Hawaii	2474884	11965998	0.05526	0.28187	29533738	5686722	Gain	Loss
United States	2449144	1579438872	0.00012	0.03767	129811833	100306358	Gain	Loss
Papua New Guinea	2396214	1287686364	0.00016	0.00147	4671824	3189041	Gain	Loss
Denmark (Greenland)	2353703	909030202	0.00024	0.40985	2495229	628141001	Gain	Gain
China	2285872	10694332164	0.00001	0.00367	6592902	66156528	Gain	Gain
Philippines	2265684	1439761734	0.00014	0.00729	14206360	17702724	Gain	Gain
Chile	2009299	705281173	0.00032	0.00039	2047996	467479	Gain	Loss
Marshall Isl.	1992232	264785476	0.00109	0.00035	132405	157083	Loss	Gain
Cook Isl. (New Zealand)	1960135	1571841	0.65916	0.00001	506437	21.8	Loss	Loss
India (mainland)	1630356	3138268241	0.00005	0.00238	2026000	12606863	Gain	Gain
Solomon Isl.	1597492	588690954	0.00038	0.00007	2895587	69713	Loss	Loss
South Georgia & San ...	1449532	8700818	0.07199	0.00000	0	26.2	Loss	Gain
France (New Caledonia)	1422543	130801479	0.00242	0.00053	8900265	116076	Loss	Loss
Viet Nam	1396299	1616696874	0.00010	0.00183	1491065	4996856	Gain	Gain

Norway	1395753	1142401126	0.00016	0.00091	11566816	1754596	Gain	Loss
Seychelles	1332031	31784907	0.01396	0.00033	1114165	17920	Loss	Loss
Fiji	1281122	169546952	0.00170	0.00016	259504	45459	Loss	Loss
Mauritius	1272787	67251290	0.00541	0.00007	1464693	8341	Loss	Loss
Madagascar	1198722	169513300	0.00168	0.00008	361154	23141	Loss	Loss
Taiwan	1149189	566041498	0.00037	0.00116	2233618	1103064	Gain	Loss
Argentina	1084386	812527743	0.00023	0.00661	10636240	9062028	Gain	Loss
South Africa	1066655	219220885	0.00118	0.00428	7721048	1580156	Gain	Loss
Azores Isl. (Portugal)	1056156	19953068	0.02357	0.00078	3004516	26114	Loss	Loss
Maldives	916189	506513914	0.00040	0.00000	0	0	Loss	Gain
Peru	906454	899263510	0.00019	0.00373	1126748	5653007	Gain	Gain
Pitcairn (UK)	836108	111140	2.00000	0.00000	0	0	Never	Gain
Ecuador (Galapagos Isl.)	835936	44349276	0.00819	0.15910	1518133	11896596	Gain	Gain
Somalia	830389	45892537	0.00784	0.00402	1418720	311218	Loss	Loss
Vanuatu	827891	62274746	0.00535	0.00008	1552356	8671	Loss	Loss
Colombia	817816	40714539	0.00907	0.09516	6254613	6532565	Gain	Gain
Cape Verde	796840	24693701	0.01683	0.00000	46863	26.1	Loss	Loss
United Kingdom	773676	1178934670	0.00013	0.06691	133801544	132993707	Gain	Loss
Iceland	772218	824516895	0.00021	0.00454	4679458	6313736	Gain	Gain
Tristan da Cunha Isl ...	754720	1690229	0.47418	0.00017	355335	492	Loss	Loss
Tuvalu	751797	36846142	0.01006	0.00005	447131	2971	Loss	Loss
Northern Marianas (USA)	749268	19562417	0.02217	0.00002	1930	621	Loss	Loss
Chile (Easter Isl.)	720395	8481587	0.06241	0.00000	1.36	36.6	Loss	Gain
Kermadec Isl. (New Z ...	678402	18599547	0.02304	0.01103	85977	345760	Loss	Gain
Tonga	664853	757240	2.00000	0.01509	2365297	19265	Never	Loss
Andaman & Nicobar I ...	659912	77748127	0.00383	0.00081	0	106788	Loss	Gain
Chagos Archipel., Br ...	638568	9438790	0.05298	0.00430	924727	68503	Loss	Loss
Bahamas	629293	69291997	0.00437	0.00415	7724328	484753	Loss	Loss
Palau	604289	46007372	0.00721	0.00043	3074975	33599	Loss	Loss
Crozet Isl. (France)	574558	1292128	0.61963		0		unknown	unknown
Costa Rica	572014	17084086	0.02455	0.01114	6879975	320973	Loss	Loss
Mozambique	571955	37015049	0.00934	0.03313	2543029	2067680	Gain	Loss
France (Kerguelen Isl.)	567732	9534078	0.05080	0.00000	0	0	Loss	Gain
Namibia	560152	310171607	0.00065	0.00000	0	0	Loss	Gain
Spain	551874	601671091	0.00028	0.01251	19438169	12686236	Gain	Loss
Falkland Isl. (UK)	550872	193000798	0.00117	0.00013	22296	42235	Loss	Gain
Yemen	544416	198976658	0.00113	0.00666	85955	2233762	Gain	Gain
Australia (Lord Howe ...	543346	11083702	0.04162	0.00984	431732	183915	Loss	Loss
Italy	537932	311236023	0.00064	0.17279	30633583	90668588	Gain	Loss
Oman	535912	204900908	0.00108	0.00900	1094645	3109032	Gain	Gain
Sri Lanka	530684	53828677	0.00574	0.00353	1456183	320133	Loss	Loss
Myanmar	520262	896792700	0.00017	0.00196	2716611	2969918	Gain	Gain
Amsterdam & St Paul ...	509015	1962912	0.35644	0.00072	0	2386	Loss	Gain
Chile (J. Fernandez, ...	502490	52579772	0.00583	0.00019	29322	16885	Loss	Loss
Angola	501050	191824218	0.00116	0.05808	4071490	18784636	Gain	Gain
Greece	494605	395639172	0.00047	0.00524	5642755	3494511	Gain	Loss
Australia (Macquarie ...	475847	1305099	0.58377	0.34241	93988	753439	Loss	Gain
Korea (South)	475469	989486809	0.00015	0.00713	2041326	11890890	Gain	Gain
Prince Edward Isl. (...	473380	383784	2.00000	0.00000	0	0	Never	Gain
Venezuela	471507	206481342	0.00104	0.04585	10053722	15961990	Gain	Gain
Brazil (Trindade & ...	468615	10250677	0.04423	0.00000	0	0	Loss	Gain
Australia (Cocos (Ke ...	467249	4350981	0.12900	0.00011	22892	842	Loss	Loss
Canary Isl. (Spain)	455397	193939815	0.00111	0.00165	1509716	539697	Gain	Loss
Madeira Isl. (Portugal)	454495	16083771	0.02499	0.00007	268168	2029	Loss	Loss
Bermuda (UK)	450370	628382	2.00000	0.00033	3307704	350	Never	Loss
Chile (Desventuradas ...	449805	47405474	0.00646	0.00000	0	0	Loss	Gain
Malaysia (Peninsula ...	447276	485624691	0.00035	0.02661	21667027	21786996	Gain	Gain
Saint Helena (UK)	444916	37633	2.00000	0.00000	197018	0.302	Never	Loss
Johnston Atoll (USA)	442635	2687160	0.23245	0.00029	4419	1331	Loss	Loss
Ascension Isl. (UK)	441658	2400456	0.26751	0.00000	0	0	Loss	Gain
Bouvet Isl. (Norway)	441163	0.9	2.00000	0.00013	32.4	0.000199	Never	Loss
Howland & Baker Isl ...	434921	4992302	0.10670	0.00060	0	5034	Loss	Gain
Clipperton Isl. (France)	431263	3158640	0.18869	0.00000	0	0	Loss	Gain
Australia (Norfolk Isl.)	431121	12137798	0.03507	0.00000	0	0	Loss	Gain
Norway (Svalbard Isl.)	426119	159320438	0.00140	0.54444	36498001	146244707	Gain	Gain
Australia (Heard & ...	417015	32470303	0.01017	0.15491	0	8480302	Gain	Gain
Ireland	410534	483248827	0.00035	0.00007	2209016	55669	Loss	Loss
USA (Wake Isl.)	407241	7588760	0.06218	0.00000	0	0	Loss	Gain
American Samoa	404391	5044339	0.10342	0.00050	586252	4263	Loss	Loss
Cuba	365448	108726150	0.00217	0.01250	3061128	2291629	Gain	Loss
Libya	355120	9649092	0.04451	0.00273	1214433	44437	Loss	Loss
Palmyra Atoll & Kin ...	352300	2852513	0.20376	0.01150	0	55301	Loss	Gain
France (Mozambique C ...	352117	9406608	0.04585	0.00000	0	0	Loss	Gain
France	334604	394308194	0.00042	0.00950	14436999	6314367	Gain	Loss
Panama	331465	100779952	0.00233	0.02083	6086088	3539178	Gain	Loss

Portugal	322197	121468768	0.00183	0.00494	5745893	1011164	Gain	Loss
New Zealand (Tokelau)	319031	220170	2.00000	0.00000	0	0	Never	Gain
Jarvis Isl. (USA)	316665	2330526	0.25542	0.00048	6394	1884	Loss	Loss
New Zealand (Niue)	316629	78006	2.00000	0.00000	137285	0.115	Never	Loss
France (Runion)	315058	4136782	0.12451	0.00001	700643	98.3	Loss	Loss
Nauru	308502	62556864	0.00415	0.00000	0	0	Loss	Gain
Equatorial Guinea	308337	3707632	0.14201	0.00426	1283007	26660	Loss	Loss
Thailand	306365	411391554	0.00039	0.02011	4248600	13949506	Gain	Gain
Jan Mayen Isl. (Norway)	292567	29051493	0.01069	0.00000	0	0	Loss	Gain
Australia (Christmas ...	277345	3655500	0.14077	0.00031	208427	1939	Loss	Loss
Morocco	272059	248203848	0.00072	0.00514	1780243	2152555	Gain	Gain
France (Tromelin Isl.)	270455	6397201	0.06950	0.00000	0	0	Loss	Gain
Denmark (Faeroe Isl.)	269866	308363023	0.00055	0.00000	0	0	Loss	Gain
Dominican Republic	269285	29872601	0.01011	0.11071	4579934	5576167	Gain	Gain
Jamaica	263283	46214990	0.00583	0.00839	2707330	653468	Gain	Loss
Egypt	261824	21432399	0.01521	0.17686	960473	6390980	Gain	Gain
France (Wallis & Fu ...	258269	9278632	0.04316	0.00000	0	0	Loss	Gain
Turkey (Black Sea)	255787	275927242	0.00062	0.01799	7169036	8371174	Gain	Gain
Liberia	246152	10487808	0.03659	0.00225	475969	39797	Loss	Loss
Tanzania	241541	19278087	0.01702	0.01745	1710822	567021	Gain	Loss
Honduras	240240	20218567	0.01601	0.01033	3133688	352200	Loss	Loss
Ghana	224908	82934991	0.00270	0.00000	0	0	Loss	Gain
Guam (USA)	221504	9705916	0.03926	0.00079	8989	12941	Loss	Gain
Pakistan	221435	336894391	0.00047	0.00985	441661	5597315	Gain	Gain
Saudi Arabia (Persia ...	219905	126260051	0.00159	0.01506	643101	3206112	Gain	Gain
Nigeria	216789	192676108	0.00093	0.00000	0	0	Loss	Gain
Gabon	193627	37911125	0.00691	0.01539	1867597	983727	Gain	Loss
Barbados	186107	527244	2.00000	0.00001	148449	10	Never	Loss
Puerto Rico (USA)	177685	18662253	0.01641	0.01568	8096022	493269	Loss	Loss
Cte d'Ivoire	174545	35156606	0.00740	0.00169	595724	100180	Loss	Loss
Sweden	170086	198703320	0.00084	0.03180	151466134	10653258	Gain	Loss
Sao Tome & Principe	165364	5336083	0.07709	0.00000	0	0	Loss	Gain
Comoros Isl.	164691	4589363	0.09298	0.00245	476699	18981	Loss	Loss
Iran	164051	325867283	0.00045	0.04577	1659250	25145278	Gain	Gain
Sierra Leone	159744	57490737	0.00392	0.00000	0	0	Loss	Gain
Senegal	157550	256383603	0.00060	0.00430	1134737	1858579	Gain	Gain
Mauritania	155422	272112304	0.00056	0.07854	695139	36033443	Gain	Gain
Turks & Caicos Isl. ...	154068	22999891	0.01220	0.00461	0	178782	Loss	Gain
Ukraine	144038	25897643	0.01034	0.02102	3071900	918021	Gain	Loss
Guyana	135900	145304993	0.00118	0.00000	0	0	Loss	Gain
France (French Guiana)	133949	19927104	0.01409	0.00059	287058	19694	Loss	Loss
Uruguay	132286	84170617	0.00232	0.00049	882908	69000	Loss	Loss
Samoa	131812	680532	0.95578	0.00079	464537	907	Loss	Loss
Algeria	128865	28936374	0.00875	0.00743	833525	362649	Loss	Loss
Suriname	128318	71198861	0.00284	0.00938	1737799	1126527	Gain	Loss
Nicaragua	127488	61752338	0.00338	0.00631	943345	656595	Gain	Loss
Cayman Isl. (UK)	119137	102438	2.00000	0.00087	6620809	151	Never	Loss
Guatemala	117743	11890022	0.02601	0.00157	998636	31511	Loss	Loss
Korea (North)	115649	186670921	0.00083	0.00000	0	0	Loss	Gain
Haiti	112025	16279701	0.01735	0.00000	0	0	Loss	Gain
Kenya	111999	2860471	0.15247	0.02725	1937028	131408	Loss	Loss
Guinea	109456	92581341	0.00196	0.00000	0	0	Loss	Gain
Antigua & Barbuda	107914	5748820	0.06313	0.00111	0	10770	Loss	Gain
Denmark	107579	346659443	0.00038	0.03747	31482989	21899655	Gain	Loss
Guinea-Bissau	106117	31149016	0.00760	0.02792	1447341	1466486	Gain	Gain
Tunisia	102362	146446726	0.00109	0.00063	430005	154496	Loss	Loss
Cyprus	98550	12728922	0.02285	0.00016	967838	3500	Loss	Loss
France (Guadeloupe)	95978	20254230	0.01270	0.00093	1664729	31620	Loss	Loss
El Salvador	93761	38600587	0.00564	0.00000	0	0	Loss	Gain
Anguilla (UK)	92178	864912	0.64769	0.00074	1844433	1076	Loss	Loss
Finland	90828	51254751	0.00392	0.01329	9734769	1148483	Gain	Loss
Sudan	88067	21853405	0.01130	0.00014	98166	5020	Loss	Loss
British Virgin Isl. (UK)	80117	2023876	0.21609	0.00063	3979160	2144	Loss	Loss
Eritrea	78703	4329718	0.08315	0.00000	0	0	Loss	Gain
Bangladesh	78538	196356430	0.00071	0.00583	562800	1929613	Gain	Gain
Trinidad & Tobago	77502	9753438	0.03001	0.00103	0	16962	Loss	Gain
Timor Leste	77256	17100051	0.01486	0.00000	0	0	Loss	Gain
Aruba (Leeward Nethe ...	68783	310469	2.00000			0		Never	unknown
Netherlands	63912	123944517	0.00119	0.01105	5974239	2308543	Gain	Loss
France (Mayotte)	63078	3268143	0.11182	0.00073	504076	4044	Loss	Loss
Germany	57259	69436840	0.00239	0.18629	20140131	21808761	Gain	Gain
United Arab Emirates	57194	119719077	0.00121	0.07440	779347	15016545	Gain	Gain
Croatia	56374	111077500	0.00132	0.00767	2539004	1437347	Gain	Loss
Malta	55556	15015987	0.01610	0.00024	598345	6053	Loss	Loss
Cambodia	47827	75262345	0.00207	0.04021	355622	5102012	Gain	Gain

France (Martinique)	47640	4117730	0.07809	0.03127	6444097	217100	Loss	Loss
Congo, R. of	40499	10004148	0.02472	0.02469	667456	416479	Loss	Loss
Estonia	39940	18101523	0.01174	0.03627	2478752	1106988	Gain	Loss
Saint Vincent & the ...	36314	415180	2.00000	0.00107	0	749	Never	Gain
Belize	35995	11187250	0.02087	0.07094	5407858	1338116	Gain	Loss
Bulgaria	35156	884748	0.49477	0.00041	166764	613	Loss	Loss
US Virgin Isl.	33744	7065866	0.03648	0.00512	2105885	60939	Loss	Loss
Latvia	32021	23338382	0.00809	0.00470	499467	184817	Loss	Loss
Qatar	31870	10324998	0.02238	0.00427	624452	74389	Loss	Loss
Poland	31600	31907446	0.00545	0.01928	1639363	1037445	Gain	Loss
Benin	30024	3537087	0.08414	0.00000	0	0	Loss	Gain
Dominica	28626	4891976	0.05543	0.00019	168927	1530	Loss	Loss
Israel	27346	5392390	0.04852	0.01884	3216120	171299	Loss	Loss
Grenada	26158	1450827	0.24763	0.00004	89379	103	Loss	Loss
Brunei Darussalam	25427	2925267	0.10234	0.00670	601285	33035	Loss	Loss
Georgia	22765	211080	2.00000	0.00164	303142	583	Never	Loss
Gambia	22630	27703810	0.00598	0.00704	334337	329004	Gain	Loss
Romania	20598	491163	0.90334	0.31368	3886846	259762	Loss	Loss
Lebanon	19196	8988387	0.02345	0.00026	145186	3947	Loss	Loss
Saint Lucia	15484	2901100	0.09135	0.00053	514069	2590	Loss	Loss
Togo	15375	2982050	0.08810	0.00059	120087	2943	Loss	Loss
Cameroon	14693	56889334	0.00218	0.26462	1261184	25380735	Gain	Gain
France (Saint Pierre ...	12334	387017	2.00000	0.00000	0	0	Never	Gain
Kuwait	12236	23491810	0.00631	0.02207	624798	873974	Gain	Gain
Netherlands Antilles ...	12169	255936	2.00000	0.01066	0	4600	Never	Gain
Channel Isl. (UK)	11658	37867645	0.00343	0.00000	0	0	Loss	Gain
Haiti (Navassa Isl.)	11494	3068268	0.07905	0.12850	384753	664753	Gain	Gain
Albania	11138	9906014	0.01812	0.02568	1415726	428860	Gain	Loss
Syria	10222	5352484	0.03829	0.00489	1168791	44142	Loss	Loss
Saint Kitts & Nevis	10201	922964	0.34444	0.00000	0	0	Loss	Gain
Bahrain	8884	47661302	0.00240	0.06545	602942	5259744	Gain	Gain
Montserrat (UK)	7582	98135	2.00000	0.00004	785522	6.55	Never	Loss
Montenegro	7415	10708858	0.01485	0.01621	380507	292681	Gain	Loss
Djibouti	6947	170801	2.00000	0.00187	331714	539	Never	Loss
Lithuania	6104	6780330	0.02505	0.09186	1577665	1050122	Gain	Loss
Belgium	3453	5100763	0.03101	0.00076	290193	6500	Loss	Loss
Gaza Strip	2584	3983356	0.03928	0.00000	0	0	Loss	Gain
China (Hong Kong)	2097	59922763	0.00126	0.28185	9725488	28475270	Gain	Gain
Congo (ex-Zaire)	1072	1521463	0.10499	0.00000	0	0	Loss	Gain
Singapore	823	25302203	0.00293	0.01369	481669	584067	Gain	Gain
Iraq	597	695604	0.24125	0.00000	0	0	Loss	Gain
Slovenia	186	687470	0.18291	0.00640	382617	7416	Loss	Loss
Jordan	95	256661	0.52986	0.42105	161262	182203	Loss	Gain
Bosnia & Herzegovina	14	1744323	0.02990	0.00000	0	0	Loss	Gain
Ecuador		36230183	3.00000		1101231		unknown	unknown
Morocco (Western Sahara)		255936	3.00000		0		unknown	unknown

F.11.1 Catch-to-stock ratios

In our economic analysis, we use the observed catch as a lower bound on the level of stock. The RAM database provides some information the typical relationships between catch and stock, which can be used to provide better bounds on the estimate.

Across all years and stocks, the ratio of catch to stock follows an exponential distribution, suggesting a wide variance of observed ratios (see figure F.7). Over time, this ratio of catch to stock has increased for many stocks (see figure F.8), although the shape of the distribution of ratios has remained similar.

The average ratio of catch to stock (excluding observations for which catch is greater than stock in a given year), is 0.1834 ± 0.0042.
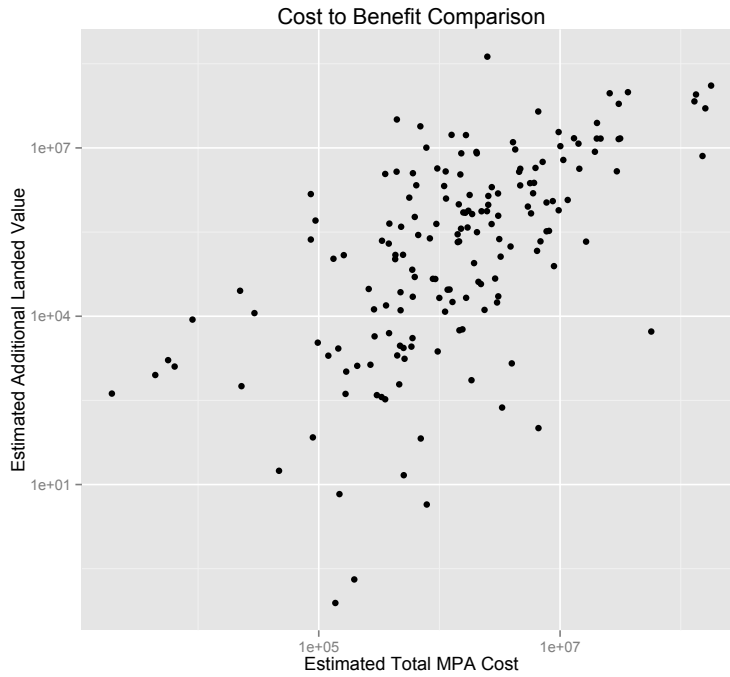
Figure F.6: Estimated costs and benefits by country. MPA costs use the full model from Balmford *et al.* (2004). Benefit estimates use our linear model.
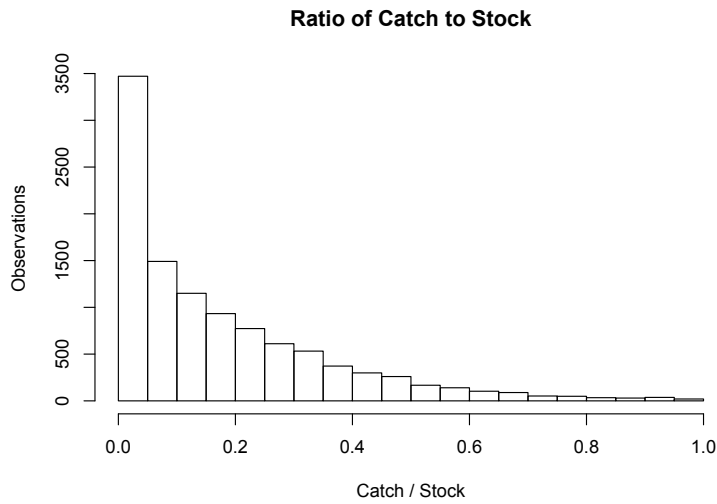

Figure F.7: A histogram of the ratios of catch to stock, across all years and stocks used in our analysis. The distribution of ratios follows an approximately exponential decay.
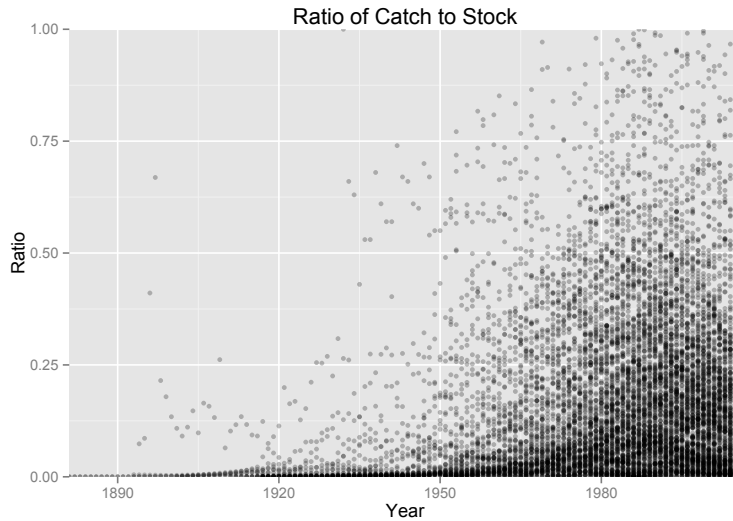
Figure F.8: The ratio of catch to stock for all years and stocks. Each point represent a different year-stock observation, with darker points produced by several points overlayed. A wide range of ratios is observed in all time periods since 1950, although the average ratio has decreased since 2000.

F.12 Additional robustness checks

The preferred specifications treat the growth rate effects of MPAs as additive with the stock-specific growth rate. For stocks with very low baseline growth rates, this predicts very large proportional increases in the growth rate. Another alternative theory is that MPAs increase growth rates proportionally.

As a proxy for the growth rate, we use the average proportional surplus (see figure F.9).

The results are very similar to the preferred specification, and if slightly easier to interpret at face value, fit the data slightly less well.

F.12.1 Temporal delay

Our model assumes that the benefits of an MPA can be observed in the following year, and will underestimate benefits that take years to build. Some studies find that MPA benefits are not realized for many years for slow-growing species, with the maturity age of a species representing the necessary wait (Roberts *et al.*, 2001).

This approach is motivated by an attempt to build a delay into the model, finding that it was most effective without the delay. This corroborates evidence that the benefits of many MPAs appear rapidly (Halpern and Warner, 2002).
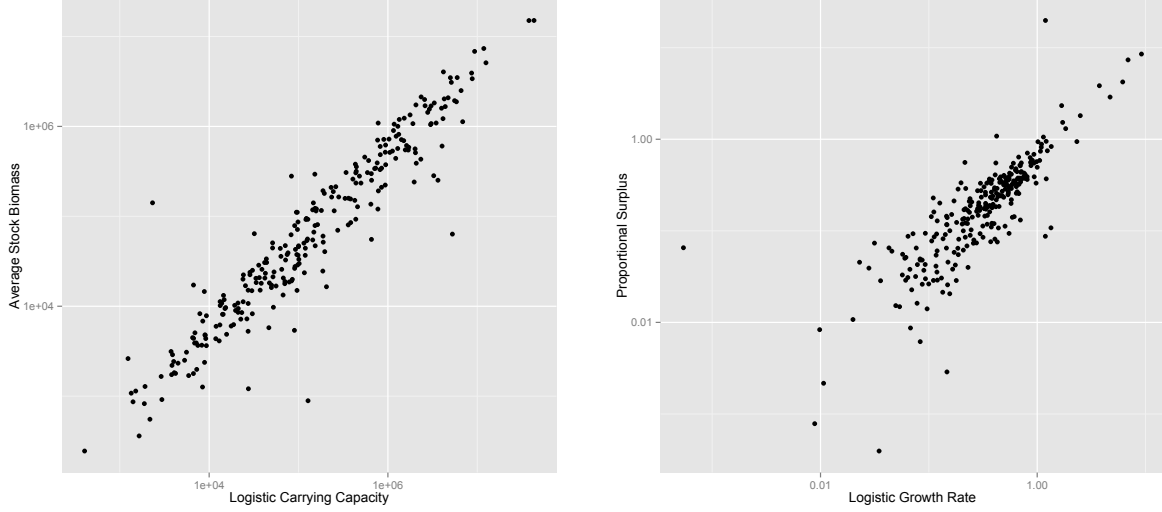
Figure F.9: Comparison between the estimated carrying capacity, K, and mean stock biomass, $\mathbb{E}S_t$ (left), and estimated growth rate, r, and average proportional surplus, $\mathbb{E}|\frac{S_{t+1}-S_t+C_t}{S_t}|$. The directly observed statistics match these derived values fairly closely.

	Timeless	Stock Trends	Year FE	Trends and Year
# MPA $S_t r_0$	−0.0083**	−0.0110***	−0.0085**	−0.0111***
	(0.0031)	(0.0027)	(0.0031)	(0.0027)
Sum MPA^2 / Area$^2 S_t r_0$	−146.7291***	−159.7456**	−155.6076***	−169.4580**
	(43.1178)	(53.9743)	(46.0673)	(56.4474)
# MPA $\bar{S} r_0$	0.0040	0.0066	0.0043	0.0063
	(0.0023)	(0.0074)	(0.0023)	(0.0074)
Sum MPA^2 / Area$^2 \bar{S} r_0$	205.3712***	227.7053***	213.1385***	235.5406***
	(31.1600)	(54.0216)	(33.8301)	(56.2433)
R^2	0.6122	0.6444	0.6150	0.6470
Adj. R^2	0.5483	0.5717	0.5487	0.5721
Num. obs.	10093	10093	10093	10093

***$p < 0.001$, **$p < 0.01$, *$p < 0.05$

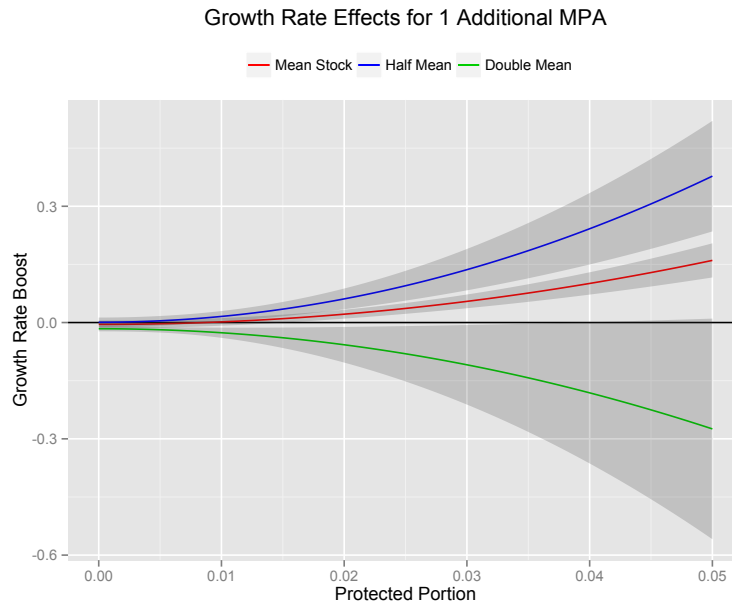Table F.12: Estimates of the main coefficients of interest for the relative estimates.

Figure F.10: The effects of a single MPA across possible sizes. The relative effect increases to a 15% in the growth rate (at the mean stock size), for an MPA accounting for 5% of an assessed region. This effect decreases for large stocks (green) and increases for small stocks (blue).

Appendix G

Distributed fishery model supplement

G.1 Plankton ocean currents estimates

A proper comparison between plankton diffusion and the larval drift estimate requires checking that for differences between the two ocean current velocity profiles. The northernmost reaches of Peru touch the high-velocity equitorial current. However, as shown in figure G.1, the velocities are only slightly higher than the $0.092\,\text{m/s}$ calculated for larvae as a whole.
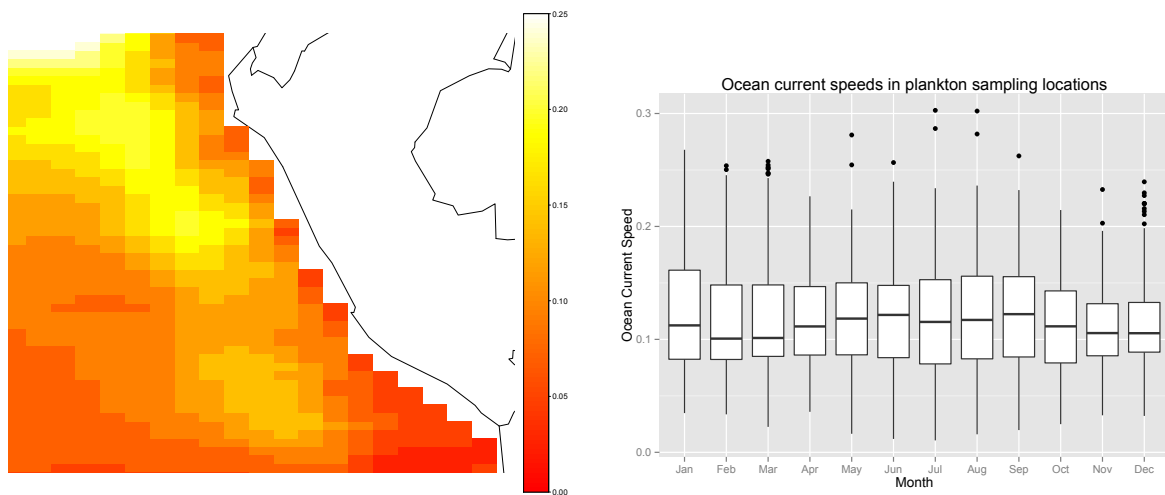

Figure G.1: Ocean currents in the plankton study region, across space and by month. **Left:** Colors denote the speed of ocean currents (in m/s), with the greatest speeds to the north, near the equator. **Right:** The speed of ocean currents by month. Each point represented in the boxplots is a grid-cell sampled in the plankton surveys, and each such grid cell is represented in every month for which the velocity is available. .

G.2 Plankton diffusion model parameters

Parameters estimated for the phytoplankton and zooplankton diffusion model in section 7.1.1.

	Phytoplankton	Zooplankton
β	0.19	0.10
F	2.0×10^7	2.1×10^7
$\hat{\gamma}(2)$	2.0×10^7	2.1×10^7
$\hat{\gamma}_\eta(2)$	2.0×10^7	2.1×10^7
$\gamma(2)$	0.066	0.035

Table G.1: Table of parameters for the variogram-based estimates of plankton diffusion. $\hat{\gamma}$ and $\hat{\gamma}_\eta$ are semivariances, while γ is the calculated diffusion parameter, each as a function of distance. The variogram values are highly uncertain, producing a wide variety of estimates at different times. However, the values at 2 degree, after the sill of the variogram, are very close to the estimated values for F.