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Three Essays on the Economics of Fisheries Management

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in

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by

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Three Essays on the Economics of Fisheries Management

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Abstract

Three Essays on the Economics of Fisheries Management

Rebecca Ellen Toseland

Problems arising from the common-pool nature of fisheries are well known (Gordon 1954). Recent research estimates the lost economic benefits from suboptimal fisheries production to be on the order of $50 billion annually (Arnason et al. 2009). Among the most promising management tools for improving fisheries performance are closed areas (marine reserves) and rights-based management (catch shares) (Worm et al. 2009). This collection of essays examines the emergence and performance of these two policy tools from an economics perspective.

The first chapter examines the determinants of the transition from command and control regulation to rights-based management in common-pool resources. I develop a conceptual framework describing a regulator's decision to adopt a rights-based management regime in a common-pool resource and empirically test the hypotheses advanced in the framework with a duration analysis of rights-based management program adoption in a group of federally managed Alaska fisheries. Consistent with the conceptual framework, I find that rent dissipation along input cost and product value dimensions increases likelihood of program adoption, and high transaction costs decrease likelihood of program adoption. I find mixed evidence that resource depletion increases probability of program adoption.

The second chapter examines the capacity of marine reserves to function as a cooperation mechanism in transboundary fisheries, stocks shared by two or more jurisdictions. Using a game theoretic model, we find that marine reserve implementation can achieve first-best equilibrium economic outcomes. We illustrate our theoretical predictions with a case study of the Chilean loco fishery.

The third chapter estimates the economic and biological outcomes of the implementation of a marine reserve network at the California Channel Islands in 2003 on the Santa Barbara Area red sea urchin fishery. The paper develops an integrated bioeconomic model of the fishery consisting of an age structured population model and a discrete choice model of fishermen behavior linked via the fishing mortality rate. We calibrate the model using a 15-year panel dataset of fishery data and use the calibrated model to simulate outcomes under a counterfactual no-reserve scenario. We find that eight years after reserve implementation, urchin biomass is 53.7% higher, fishery yield is 11.3% lower, and fishery revenue is 10.9% lower compared to simulated outcomes under a counterfactual no-reserve policy.

Professor Christopher Costello
Dissertation Committee Chair

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Chapter 1

Transition to Property Rights in Common-Pool Resources: Evidence from Alaska Fisheries

1.1 Introduction

Economists have long recognized that competition for common-pool resources results in rent dissipation, and that this "tragedy of the commons" (Hardin 1968) can be mitigated by the assignment of clearly defined property rights to the resource (Gordon 1954).[1] Nevertheless, the extent to which common-pool losses have been mitigated varies widely (Dietz et al. 2003). This paper expands our understanding of common-pool resource governance by delineating the determi-

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[1] This work was funded by the Pacific Fisheries Information Network, the NOAA Alaska Fisheries Science Center (AFSC), and the Property and Environment Research Center (PERC). I acknowledge generous data and research support from Ron Felthoven, Ben Fissel, Brian Garber-Yonts, and other staff members at the AFSC Economic and Social Sciences Research Program and Kristen Green and Kamala Carroll at the Alaska Department of Fish and Game. I also thank Joshua Abbott, Christopher Costello, Sam Cunningham, Robert Deacon, Jane DiCosimo, Gary Libecap, Doug Steigerwald, Wally Thurman, and the UCSB Econometrics Research Group.

nants of (property) rights-based management adoption in common-pool resources and applying the resulting framework to a marine fisheries empirical application.

The paper makes two principal contributions. First, I advance a conceptual framework of a regulator's decision to adopt a rights-based management program, a system in which exclusive rights to a resource are allocated among individuals or groups of resource users. The framework draws from existing literature on the evolution of management institutions in common-pool resources, which places institutional transition in a cost-benefit context, but my approach more precisely defines the relevant costs and benefits and introduces empirically testable hypotheses about rights-based management adoption. Second, I empirically test the conceptual framework hypotheses with a duration analysis of rights-based management program adoption using a panel dataset of federally managed Alaska fisheries. Consistent with the conceptual framework, I find that rent dissipation arising from inefficient production under command and control regulation increases the likelihood of rights-based management program adoption, and the presence of high transaction costs decreases the likelihood of adoption. Finally, I find mixed evidence that resource depletion increases the probability of transition to rights-based management.

A substantial body of empirical research addresses the performance of property rights institutions in common-pool resources. This research generally finds that well-designed and enforced property rights can substantially mitigate common-pool externalities relative to weak or insecure rights. For example, Hornbeck (2010) finds increases in land value and agricultural productivity among land owners on the American Plains in the late nineteenth century following the introduction and near-universal adoption of barbed wire, which allowed land owners to demarcate and enforce their land rights. Similarly, Goldstein and Udry (2008) show that individuals with more secure land tenure rights invest more in land fertility and have substantially higher agricultural output.[2] Bohn and Deacon (2000) find that insecure property rights increase rates of deforestation worldwide.[3] Pfeiffer and Lin (2012) empirically quantify losses from spatial externalities in a shared groundwater aquifer and show that these losses are internalized in cases where water rights are allocated at a spatial level that prevents competition among users for resource rents. Finally, Grafton et al. (2000), Newell et al. (2005), and Costello et al. (2008) find substantial gains in economic efficiency and resource sustainability in fisheries in which exclusive harvest rights are allocated among individual resource users.

Although empirical research indicates that rights-based management institutions can mitigate common-pool losses, they generally do not emerge as a

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[2] See Besley (1995), Banerjee et al. (2002), and Jacoby et al. (2002) for further emprical work documenting the relationship between property rights institutions and agricultural productivity.

[3] See Alston et al. (2000) and Araujo et al. (2009) for empirical evidence linking property rights insecurity to deforestation in the Brazilian Amazon.

first-response to common-pool resource problems. Instead, the typical government response is command and control regulation, prescriptive rules designed to constrain user extraction effort. Libecap (2008) argues that command and control regulation prevails initially because this approach provides some protection of the resource stock while avoiding political costs associated with rights allocation. Further, he observes that rights-based management institutions are typically implemented only after command and control regulation has failed to adequately control common-pool externalities and the expected gains from rights-based management are clear.

Libecap's observations are consistent with classic economic theory on property rights, which casts the emergence of property rights in common-pool resources in a cost-benefit framework. Demsetz (1967) argues that property rights will develop to internalize externalities when the benefits of establishing a rights-based system exceed the costs. Anderson and Hill (1975) also assert that the evolution of property rights institutions can be explained and predicted by a comparison of the benefits and costs of defining and enforcing property rights. Similarly, Cheung (1970) notes that, while there are potentially large gains from allocating exclusively delineated, well-enforced property rights to common-pool resource users, the costs of such arrangements are also high, and this explains their absence in many resources.

More recent work on the emergence of property rights in common-pool resources emphasizes the role of transaction costs arising from heterogeneity among resource stakeholders. Libecap (1989) provides a general analytical framework for analyzing the development of property rights institutions in common-pool resources that focuses on the transaction costs associated with assigning property rights among multiple, likely heterogeneous claimants. Boyce (2004) develops a model of management instrument choice in fisheries in which a regulator's decision to implement rights-based management hinges on her relative weighting of the welfare of multiple resource stakeholders. Grainger and Costello (2012) and Deacon et al. (2013) provide theoretical and empirical evidence that heterogeneity in harvesting costs among fishermen can lead to opposition to rights-based management in fisheries. Wiggins and Libecap (1985) and Libecap and Smith (2002) highlight the role of negotiation costs among heterogeneous resource users in delaying or preventing implementation of rights-based management arrangements in United States oil and gas reservoirs.

In this paper, I seek to contribute to the literature on the emergence of property rights in common-pool resources by developing a general and empirically testable conceptual framework for analyzing the transition from command and control regulation to right-based management that can be broadly applied to a range of empirical contexts. I then use the framework to examine the transition to rights-based management in a marine fisheries context. The paper proceeds as follows. Section 2 presents a conceptual framework of rights-based management adoption in which I discuss the evolution of management institutions in common-pool resources, introduce a cost-benefit framework describing a regulator's decision to transition a common-pool resource from a command and control regime to a rights-based management regime, and develop several empirically testable hypotheses. Section 3 introduces the institutional setting for the empirical application. Section 4 lays out the duration analysis empirical approach. Section 4 describes the dataset. Section 6 presents the main results and discusses the extent to which they do and do not support the conceptual framework hypotheses. Section 7 concludes with thoughts on the significance of the results for U.S. fisheries policy and further research opportunities.

1.2 Conceptual framework

1.2.1 Rent dissipation under open access

Institutional settings in common-pool resources can be broadly classified into three regimes that generally evolve in the following order: (1) open access, (2) command and control regulation, and (3) rights-based management (Libecap 2008).[4]

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[4] In this paper, I focus on command and control regulation and rights-based management initiated by a central regulator as possible responses to open access losses. Rent dissipation in

This pattern, shown in Figure 1.1, has been observed in fisheries and oil and gas reservoirs, among other resources (Sanchirico and Wilen 2007; Libecap and Smith 2002).[5] Because common-pool resources are characterized by non-excludability, an open access regime initially prevails in which users have unrestricted access to the resource. Under open access, competitive extraction among resource users leads to rent dissipation along three possible dimensions: (1) resource, (2) cost, and (3) value (Knapp and Murphy 2010).

Figure 1.1: The Evolution of Management Institutions in Common-Pool Resources

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common-pool resources can also be addressed through the emergence of decentralized property rights regimes, protected by tradition or custom (e.g., Acheson 1988). In general, these regimes are successfully implemented in a sub-group common-pool resources with specific characteristics (see Ostrom 1990, 1998).

[5] This institutional trajectory has also been observed in the response to environmental externalities resulting from the over-provision of public bads such as air pollution (see, for example, Stavins 2007). Common-pool resources and public goods (bads) share the characteristic of non-excludability, but differ in the characteristic of rivalry (common-pool resources are rival in consumption, public goods (bads) are not). For this reason, aspects of the net benefits framework describing adoption of rights-based management in common-pool resources are also applicable to a conceptual framework of transition to rights-based management for public goods (bads), but the framework is not directly transferrable. In particular, the costs of adoption also apply to public goods (bads), but the benefits of differ between the two categories.

The *resource dimension* of rent dissipation refers to forgone rents resulting from depletion of the resource stock to a suboptimal level. In the absence of property rights, rights to the resource are assigned according to the rule of capture in which the first user to establish possession of the resource owns the resource. Any units of the resource not extracted by one user may be extracted by another user, leaving no incentive to conserve the resource for use in future periods. Moreover, in the absence of individual rights to the resource, users capture the full benefit of each unit of the resource extracted while bearing only a fraction of the social cost of associated resource depletion. This disparity between the benefits received and the costs incurred from extraction leads to suboptimal resource depletion. In a fisheries context, the resource dimension of rent dissipation is characterized by over-extraction of the target species and wasteful fishing practices including use of gear that causes habitat damage, high discard rates (non-retained target species), and high bycatch rates (catch of non-target species).

The *cost dimension* of rent dissipation refers to forgone rents resulting from the cost of inefficient extraction effort and excess capital investment relative to the social optimum. Over-investment in labor and capital and inefficiency in how both input factors are applied to resource extraction results from competition among users to capture the resource before other users. In fisheries, the cost dimension of rent dissipation is often referred to as the "race to fish," characterized by inefficient investment in the number and capacity of vessels, wasteful fishing practices (e.g., gear loss), and fishing in unsafe conditions.

Finally, the *value dimension* of rent dissipation refers to forgone rents resulting from the failure to realize the full market value potential of the resource when it is sold as a lower-valued product. In fisheries, the value dimension of rent dissipation is a common outcome of the race to fish. For example the race to fish often leads to high volumes of fish caught over short time periods resulting in higher ratios of lower-valued frozen product to higher-valued fresh product (Homans and Wilen 2005).

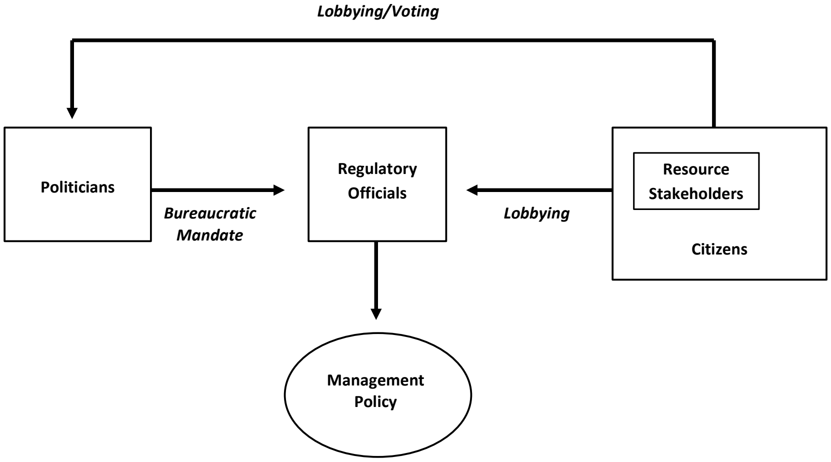
The extent to which rent dissipation along these three dimensions occurs in an open access regime depends critically on the value of the resource (Baland and Platteau 1996). When demand for the resource is low, the resource is of low value and rent dissipation is minimal. As the value of the resource rises due to higher consumer demand, competition for the resource among users increases and rent dissipation becomes more severe, leading to government intervention to correct market failure.

1.2.2 Government response to open access

Government intervention in common-pool resources to alleviate problems of rent dissipation is depicted in Figure 1.2 and proceeds as follows. Government is composed of politicians who are responsive to citizen demands through voting and lobbying (Becker 1983; Peltzman 1976). Resource stakeholders are a subset of all citizens broadly defined to include all groups affected by resource extraction (e.g., resource users (harvesters), entities in the supply chain, local communities). In representing the public interest and meeting resource stakeholder demands, politicians seek to maximize rents from the common-pool resources under their jurisdiction while considering the distribution of rents among users. To achieve these goals, politicians create regulatory agencies and issue bureaucratic mandates to these agencies. Bureaucratic mandates define objectives for resource management. Politicians then appoint regulatory officials (regulators) to set management policy consistent with the agency's bureaucratic mandate. Regulators accordingly seek to set management policy to satisfy the objectives of their bureaucratic mandate. At the same time, regulators face demands from resource stakeholders who lobby for management policy that enables them to maximize individual rents from resource use.

In forming management policy, regulators balance the demands of resource stakeholders with their bureaucratic mandate, both of which contribute to *transaction costs*. Transaction costs are the regulatory agency's administrative costs of designing, implementing, monitoring, and enforcing a new management policy. These costs include those associated with satisfying their bureaucratic mandate

Figure 1.2: Political Economy of Common-Pool Resource Management



and accounting for lobbying from users concerned about the relative distribution of rents resulting rom a policy change. The bargaining positions taken by various resource users depend on how they view their expected welfare to change under the new institutional arrangement relative to the status quo. In general, the more numerous and heterogenous the users and the greater the difference between the distribution of rents under the status quo and the projected distribution of rents under new policy, the larger the transaction costs (Libecap 1989).

Regulators commonly respond to open access losses initially by enacting command and control regulation. Command and control regulations are uniform standards aimed at constraining user extraction behavior to mitigate one or more dimensions of rent dissipation. Examples include placing a quota on the level of resource extraction permitted in a given time period, limiting entry, and issuing restrictions on production inputs. Command and control regulations are consistent with the regulator's objectives initially because they typically offer some mitigation of rent dissipation with relatively low transaction costs compared to rights-based management. Command and control regulations are often less politically contentious compared to rights-based approaches because uniform standards typically preserve relative rent distribution among users, and they are thus met with less opposition from resource users on distributional grounds (Libecap 2008). Rights-based management programs may also require more institutional infrastructure and incur larger administrative costs relative to command and control approaches.[6]

1.2.3 Transition from command and control regulation to rights-based management

In general, command and control regulation is not sufficient to solve common-pool problems in the long-term because users will continue to dissipate rents along unregulated margins (Homans and Wilen 1997). In contrast, rights-based management policies, which allocate secure rights to the resource among individuals

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[6] For example, the Magnuson-Stevens Act includes specific provisions for cost recovery to offset additional administrative costs resulting from catch share programs. See Magnuson-Stevens Fishery Conservation and Management Reauthorization Act, Pub. L. No. 109-479, (2007).

or groups of resource users, are more effective at mitigating rent dissipation by alleviating competition for rents among users and providing incentives for resource stewardship, but they may also have high transaction costs. Following previous theory on the emergence of property rights in common-pool resources, I argue that a regulator will adopt a rights-based management program if, from the perspective of the regulator, the net benefits of adoption are larger than the net benefits under the status quo command and control regime. This statement leads to the following four hypotheses about a regulator's decision to adopt a rights-based management program:

Resource Dimension Hypothesis: Rights-based management program adoption is more likely when the *resource dimension* of rent dissipation under the status quo regime is high.

Cost Dimension Hypothesis: Rights-based management program adoption is more likely when the *cost dimension* of rent dissipation under the status quo regime is high.

Value Dimension Hypothesis: Rights-based management program adoption is more likely when the *value dimension* of rent dissipation under the status quo regime is high.

The Resource, Cost, and Value Dimension Hypotheses imply that the higher the potential benefits for mitigating rent dissipation from replacing a command and control regime with a rights-based regime, the more likely rights-based management adoption will occur. The potential benefits from rights-based management adoption are directly related to the extent to which existing command and control regulations are able to mitigate rent dissipation. For instance, if a regulator is able to successfully implement and enforce a resource extraction quota under a command and control regime consistent with an optimal extraction path, the resource dimension of rent dissipation under the status quo regime and, thus, the benefits from transition to rights-based regime may be low. On the other hand, if this policy is unable to adequately halt wasteful resource degradation, there may be large gains from rights-based management along the resource dimension.

Transaction Costs Hypothesis: Rights-based management program adoption is more likely when the *transaction costs* of program adoption are low.

The Transaction Costs Hypothesis states that lower the transaction costs of the policy change, the more likely we are to observe a transition to rights-based management. The level of transaction costs may change over time based on factors such as the number and heterogeneity of resource users, availability of information about resource status, and the state of institutional infrastructure.

In delineating the categories of benefits and costs of the transition from command and control regulation to rights-based management, the framework is agnostic about the relative weight the regulator places on each; this is an empirical question. I now provide an empirical application to a group of federally-managed fisheries in Alaska to give context to the conceptual framework and to test the hypotheses about the determinants of rights-based management adoption.

1.3 Empirical application: The institutional setting

1.3.1 United States fisheries policy

In the late 1970s, the United Nations Convention on the Law of the Sea provided the legal framework for coastal nations to establish jurisdiction over marine resources up to 200 miles off their coastline through the establishment of exclusive economic zones (EEZs). This international agreement marked an important transition from open access to command and control regulation in many commercially important fisheries as coastal nations enacted national fisheries management legislation to establish their EEZ and form regulatory agencies responsible for managing domestic fisheries.

The Fishery Conservation and Management Act of 1976, later renamed the Magnuson-Stevens Fishery Conservation and Management Act, established the United States' 200-mile EEZ and created eight Regional Fishery Management Councils responsible for establishing management policy for stocks occurring in federal waters (3 to 200 miles offshore). Councils are composed of between 7 and 21 voting members and 4 non-voting members representing a range of bureaucratic and fishing industry viewpoints. Voting members are comprised of: (1) regulatory officials from related fisheries management agencies in the Council's jurisdiction, including the National Marine Fisheries Service and state fisheries management agencies, and (2) members appointed by the Secretary of Commerce based on gubernatorial nominations from states in the Council's jurisdiction. Appointed members are typically representatives from the fishing industry.

The Magnuson-Stevens Act is the principal component of the Councils' bureaucratic mandate. It defines each Council's required functions including the development of fishery management plans (FMPs), documents that establish management and conservation measures for each fishery or group of fisheries under its jurisdiction that requires management. The establishment of an FMP therefore represents the initiation of command and control regulation at the federal level. The Councils can then pass subsequent amendments to the FMP to modify existing command and control regulations or implement a rights-based management program.

The Magnuson-Stevens Act specifies ten National Standards for fishery conservation and management and requires all FMPs to be consistent with these standards. The National Standards include provisions for rent maximization, requiring management measures to "prevent overfishing while achieving, on a continuing basis, the optimum yield from each fishery for the United States fishing industry," and "where practicable, consider efficiency in the utilization of fishery resources."[7] At the same time, the National Standards require the Councils to consider the transaction costs of new policies, including distributional costs, mandating that management measures "where practicable, minimize costs" and "minimize adverse economic impacts on [fishing] communities."[8] The National Standards also state that if it "becomes necessary to allocate or assign fishing privileges among various United States fishermen, such allocation shall be... fair and equitable to all such fishermen." Further, the Magnuson-Stevens Act requires Councils to hold public hearings to "allow all interested persons an opportunity

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[7] Magnuson-Stevens Fishery Conservation and Management Reauthorization Act, Pub. L. No. 109-479, (2007).

[8] Ibid.

to be heard in the development of fishery management plans and amendments to such plans."[9]

In addition to upholding the Magnuson-Stevens Act, Councils must adhere to the requisites of several other statutes. Among these statues is Executive Order 12866 on regulatory planning and review, which requires regulatory agencies to "assess all costs and benefits of available regulatory alternatives, including the alternative of not regulating" and to "select those approaches that maximize net benefits (including potential economic, environmental, public health and safety, and other advantages; distributive impacts; and equity)."[10] Councils must also uphold the standards of the Endangered Species Act, the National Environmental Policy Act, which requires federal agencies to incorporate environmental considerations in their planning and decision-making, and the Regulatory Flexibility Act, which requires consideration of impact of proposed regulations on the needs and capabilities of small businesses and other small entities.

When considering a major policy change such as a transition from command and control regulation to rights-based management, Councils seek to satisfy their bureaucratic mandate by having Council staff prepare an Environmental Assessment/Regulatory Impact Review/Initial Regulatory Flexibility Analysis document, which lays out the regulatory alternatives and the capacity of each to fulfill

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[9] Ibid.

[10] Exec. Order No. 12866, 58 Fed. Reg. 190 (Oct. 4 1993).

the Council's bureaucratic mandate. In addition, there exist ample opportunities for resource stakeholders to lobby the Councils throughout the policy-making process through written comment or oral testimony at Council meetings.

1.3.2 Rights-based management adoption in Alaska fisheries

The North Pacific Fishery Management Council establishes management policy for federally controlled fisheries in Alaska. Consistent with the Councils' bureaucratic mandate, in managing the fisheries under its jurisdiction, the North Pacific Fishery Management Council has a stated commitment to providing "the maximum benefit to present generations of fishermen, associated fishing industry sectors, communities, consumers, and the nation as a whole" (NPFMC 2006).

For fisheries management purposes, federal waters in Alaska are divided into two main regions: (1) the Bering Sea and Aleutian Islands (BSAI), and (2) the Gulf of Alaska (GOA) (Figure 1.3). Within these two regions, three fishery management plans establish management measures for 98% of federally-managed Alaska fisheries by volume and 80% by value: (1) the Bering Sea and Aleutian Islands Groundfish Fishery Management Plan (BSAI Groundfish FMP), (2) the Gulf of Alaska Groundfish Fishery Management Plan (GOA Groundfish FMP), and (3) the Bering Sea and Aleutian Islands Crab Fishery Management Plan

(Crab FMP) (NPFMC 2012, Hiatt et al. 2011). Table A.1 in the Appendix lists fisheries included in this study and the FMPs to which they belong.[11] Fisheries are defined as species-management area combinations (e.g., Bering Sea pollock) with a regulated annual total allowable catch (TAC). Within fisheries, the TAC may be further allocated among multiple fishing sectors. A fishing sector is defined as a group of vessels of the same type using the same gear to fish. In Alaska, harvesting vessels are divided into two types: (1) catcher vessels that deliver catch to shoreside processors and (2) catcher/processor vessels that both harvest and process fish. The primary gear types are trawl, longline, pot, and jig.

In fisheries, rights-based management programs are commonly called catch shares.[12] When a catch share program is adopted, it may include only a particular sector or sectors within a fishery and thus only include a portion of the TAC. In the primary empirical specifications, I consider a fishery as having adopted catch shares if at least 50% of the fishery TAC is allocated to a catch share program, but I also consider models in which catch share adoption is defined as any percentage of the TAC allocated to a catch share program.

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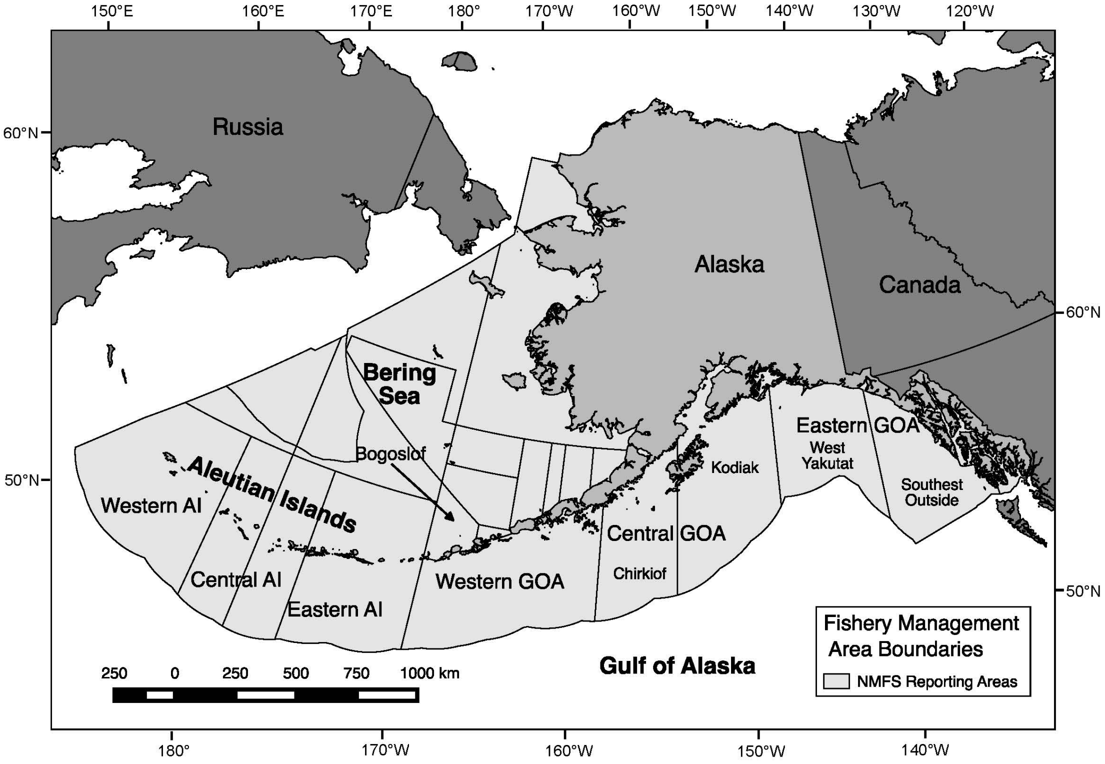
[11] The Scallop, Salmon, and Arctic FMPs are excluded from the analysis. The fisheries managed in these FMPs constitute a relatively small component of Alaska fishing activity in federal waters.

[12] Types of catch share programs include: individual transferrable quotas (ITQs), and their variants, in which shares of the TAC are allocated to individual users; cooperatives in which shares of the TAC are allocated to groups of users; and territorial user rights for fisheries (TURFs) in which spatial rights are allocated to individuals or groups (Hannesson 2004).

Figure 1.3: Alaska Fisheries Management Areas



A timeline of major management events affecting the fisheries in the study is shown in Figure 1.4. The three FMPs included in the study were adopted between 1978 and 1989. In 1996, the Council amended the FMPs to a implement a limited entry policy by first placing a moratorium on new vessel entry and subsequently introducing a formal license limitation program regulating vessel entry and exit in each fishery. Since limited entry was established, several fisheries have transitioned to rights-based management under four catch share programs summarized in Table

1.1.[13] Three of the programs were designed and adopted by the North Pacific Fishery Management Council and one was enacted by federal legislation.[14] The BSAI American Fisheries Act Pollock Cooperatives program implemented cooperatives in the Bering Sea pollock fishery in 1999 under the American Fisheries Act. The BSAI King and Tanner Crab Rationalization Program began in 2005 and created a program with ITQ and cooperatives elements for eight of the BSAI crab fisheries. The Central GOA Rockfish Pilot Cooperatives program, adopted in 2007, allocated 95% of the TAC of the three primary species (Pacific ocean perch, Northern rockfish, and pelagic shelf rockfish) and smaller portions of the TAC of several secondary species (thornyhead rockfish, shortraker and rougheye rockfish, Pacific cod, and sablefish) among several catcher and catcher/processor vessel cooperatives. Finally, the BSAI American Fisheries Act Non-Pollock Cooperatives program (commonly referred to as Amendment 80), adopted in 2008, allocated 90-100% of the TAC in the yellowfin sole, rock sole, flathead sole, Atka mackerel, and Aleutian Islands Pacific ocean perch fisheries as well as 13.4% of the Pacific cod TAC as catch shares to voluntarily formed fishing cooperatives.

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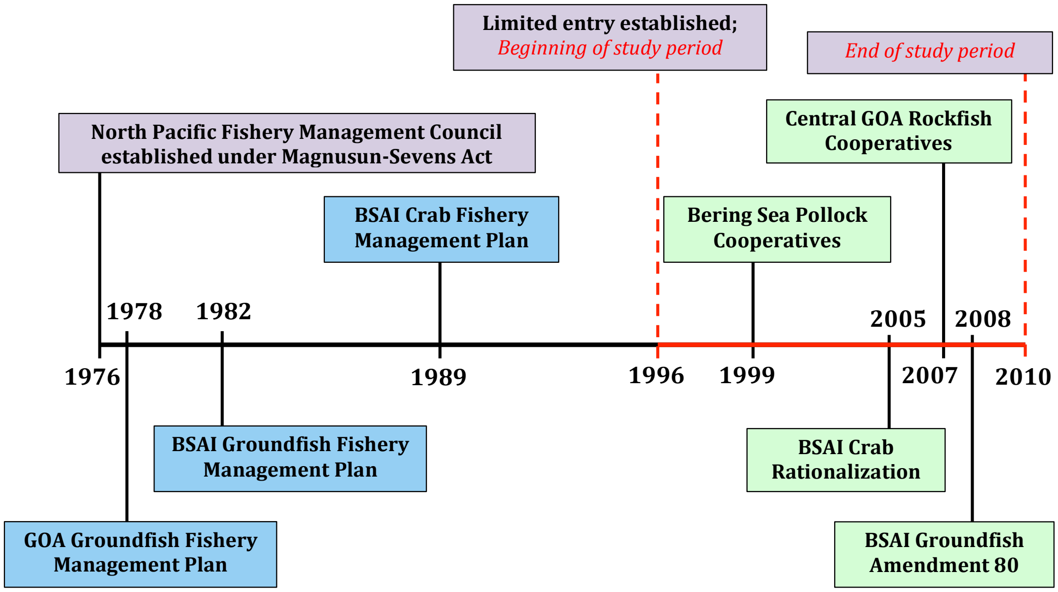
[13] I do not consider the 1992 Western Alaska Community Development Quota (CDQ) Program a catch share program for purposes of analysis. Although this program does allocate exclusive use rights to a portion of the TAC in certain fisheries in the BSAI Groundfish and Crab FMPs, these allocationa are small (<15%), and primarily for distributional purposes (to promote economic development and alleviate poverty in Western Alaska) rather than to improve aggregate fisheries outcomes. Except where noted, the data exclude CDQ allocations. I also exclude the 1995 Halibut and Sablefish IFQ program from the analysis.

[14] See Fina (2011) for a detailed qualitative summary of the design and implementation of catch share programs in Alaska.

Figure 1.4: Alaska Fisheries Management Timeline



1.4 Econometric framework

1.4.1 Duration analysis

I use duration analysis to empirically identify the determinants of catch share program adoption in the Alaska fisheries dataset. Duration analysis provides a natural empirical approach for economic applications in which the response variable is the time elapsed until a certain event occurs (see Kiefer (1988) and Wooldridge (2002) for an overview). The primary advantages of using duration models in this context are that right-censored observations, fisheries that never adopt a catch share program during the observation period, are handled in a straightforward way using maximum likelihood methods, and both time-constant and time-varying covariates can be introduced without any estimation or conceptual problems.

Duration models are estimated in terms of a hazard function, $\lambda(t)$, which represents the instantaneous probability of an event (catch share program adoption) conditional on the event not having occurred prior to time t. Let $f(t)$ be a continuous probability density of a random variable T, the number of years until catch share program adoption. The corresponding cumulative distribution function is given by $F(t)$. Equivalently, the distribution of T can be expressed by the survivor function,

$$S(t) = 1 - F(t) = Pr(T > t), \tag{1.1}$$

the probability that catch shares are adopted after time t. The survivor function equals 1 at $t = 0$ and weakly decreases towards 0 as t goes to infinity. That is, we assume that all fisheries eventually adopt catch shares. Analytically, the hazard function is defined as

$$\lambda(t) = \lim_{h \to 0} = \frac{\Pr(t \leq T < t + h | T \geq t)}{h} = \frac{f(t)}{S(t)}. \tag{1.2}$$

To introduce fishery-level characteristics that can vary over the observation period, one must parameterize the hazard rate as a function of these characteristics. Let $\mathbf{x}(\mathrm{t})$ denote the vector of regressors that affect the probability of catch share program adoption at time t. For $t \geq 0$, let $\mathbf{X}(\mathrm{t})$ be the covariate path up through time t. The conditional hazard function is defined as

$$\lambda[t|\mathbf{X}(t)] = \lim_{h \to 0} = \frac{\Pr[t \leq T < t + h | T \geq t, \mathbf{X}(t+h)]}{h}. \tag{1.3}$$

The most widely used formulation of the conditional hazard function in Equation 1.3 is the proportional hazards model with time-varying covariates

$$\lambda[t|\mathbf{x}(t)] = \lambda_0(t, \boldsymbol{\alpha})\phi[\mathbf{x}(t), \boldsymbol{\beta}], \tag{1.4}$$

where $\lambda_0(\cdot) > 0$ is a baseline hazard common to all fisheries, $\phi[\cdot]$ is a nonnegative function that multiplicatively shifts the baseline hazard, and $\boldsymbol{\alpha}$ and $\boldsymbol{\beta}$ are a parameter vectors. The most common choice of $\phi[\cdot]$ is the exponential form, $\phi[\cdot]$ $= \exp[\mathbf{x}(t)\boldsymbol{\beta}]$. Thus, a fundamental assumption of proportional hazard models, known as the *proportional-hazards assumption*, is that the baseline hazard is the same for all fisheries, but its shape is multiplicatively shifted by fishery-level characteristics.

I estimate Equation 1.4 using a Cox proportional hazards model (Cox 1972) specification

$$\lambda[t|\mathbf{x}(t)] = \lambda_0(t)\text{exp}[\mathbf{x}(t)\boldsymbol{\beta}], \tag{1.5}$$

The Cox model is a semiparametric model in which the baseline hazard is left unspecified and estimation is conducted with a partial likelihood approach. Compared to a correctly specified fully parametric model, the $\boldsymbol{\beta}$-estimator is inefficient, but the efficiency loss is small, and avoids problems of inconsistency in a misspecified parametric model (Cameron and Trivedi 2005). Therefore, the Cox model is an advantageous approach for the current case in which there is no obvious theoretical assumption about the functional form of the baseline hazard.

1.4.2 Unobserved heterogeneity

An important identification issue is the possible presence of unobserved heterogeneity among groups of fisheries. That is, one might be concerned that the hazard rate of catch share adoption varies among groups of fisheries in ways that are unaccounted for by the model covariates. Specifically, when fisheries are harvested concurrently (e.g., species are caught together in the same trawl tows) or fished by the same fishing fleet, regulators often implement management policy changes for the entire group of fisheries.

I account for this issue in two ways. First, I consider models with management area and fishery management plan fixed effects. These models indirectly capture group effects among fisheries by controlling for fisheries located in the same geographic region and for fisheries grouped together and managed collectively by the North Pacific Fishery Management Council. Second, I estimate a *shared frailty* model in which a latent random effect for fishery groups enters the hazard function multiplicatively.[15] To estimate a Cox model with shared frailty, the data are organized into $i = 1, ..., n$ groups with $j = 1, ..., n_i$ fisheries in the ith group. For the jth fishery in the ith group, the hazard function becomes

$$\lambda_{ij}[t|\mathbf{x}_{ij}(t)] = \gamma_i \lambda_0(t) \exp(\mathbf{x}_{ij}\boldsymbol{\beta}). \tag{1.6}$$

where γ_i is the group-level random effect, which is assumed to have mean 1 and variance θ and is estimated along with the other parameters in the model. When $\theta = 0$, the shared frailty model simply reduces to the standard Cox proportional hazards model. Table A.2 in the Appendix enumerates the groups and provides rationales for the groupings. Generally, fisheries are assigned to the same group if they jointly adopted a catch share program or are harvested by the same fishing fleet (NPFMC 2012).

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[15] Shared frailty is a term used in duration analysis to describe regression models with random effects.

1.5 Data

To test the hypotheses advanced in the conceptual framework in a fisheries context, I assemble a panel dataset of 68 federally-managed fisheries in Alaska observed annually from 1996-2010. The data come from several sources produced by NOAA's National Marine Fisheries Service including Stock Assessment and Fishery Evaluation Reports (SAFEs), Catch Reports, Harvest Specification Tables, Status of Fisheries Reports, Weekly Production Reports, and groundfish landings tickets as well as the Alaska Department of Fish and Game's Shellfish Observer Program Fishery Management Reports, and crab fish ticket and eLandings data. A summary of the variables included in the analysis appears in Table 3.5 along with the expected effect of each variable on the likelihood of catch share program adoption.

Four variables measure the resource dimension of rent dissipation. The first two variables, stock status and overfishing status, are the primary indicators used to assess the biological status of federally managed fisheries (NOAA 2011). Stock status is defined as the ratio of stock biomass, B, to the maximum sustainable yield biomass, B_{MSY}, (or equivalent) as defined for each fish stock in the stock assessment reports.[16] The higher this ratio, the healthier the fish stock and the

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[16] In some cases, a single fish stock is managed as multiple fisheries, and in these cases, I assign the same stock status to each fishery.

smaller the implied resource dimension of rent dissipation. A stock is generally considered healthy if this metric is at least 1 and "overfished" if it is less than 0.5.[17] Stock status is only defined for 47 fisheries in the sample; the remaining fisheries lack adequate stock assessment data to measure this reference point. Overfishing status is calculated as the percentage of the overfishing level (OFL) defined in the stock assessment reports realized by the aggregate catch. A higher percentage implies a larger resource dimension of rent dissipation. In particular, if this percentage exceeds 100, the fishery is considered "subject to overfishing" in that year. The principal difference between overfishing status and stock status is that overfishing status is a short-term measure of the fishing mortality rate determined entirely by harvesting activity within a single year. Stock status, in contrast, is a long-term measure of stock level, which can be affected by both the fishing mortality rate as well as other factors including natural population cycles, habitat degradation, climate change (i.e., ocean temperature and acidification), and water pollution.

I also include two additional variables that measure widely cited indicators of ecological stewardship in fisheries and rationales for catch share program adoption, the discard rate and the bycatch rate (Essington 2010; Brinson and Thunberg

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[17] See stock assessment reports for further detail: `http://www.afsc.noaa.gov/REFM/Stocks/assessments.htm` (groundfish) and `http://www.npfmc.org/safe-stock-assessment-and-fishery-evaluation-reports/` (crab).

2013). The discard rate is the percentage of non-retained total catch. A higher discard rate implies a larger resource dimension of rent dissipation because discards contribute to the overall fishing mortality rate. I measure the bycatch rate indirectly by including a bycatch closure indicator that takes the value '1' if the fishery is closed during the fishing season to avoid exceeding regulated non-target species catch limits.

Two metrics proxy for the cost and value dimensions of rent dissipation: (1) season length, the number of days the fishery is open to commercial fishing in a given year, and (2) TAC exceeded, a binary variable indicating whether the aggregate catch was greater than the fishery TAC.[18] These metrics were recently used to assess the *ex post* economic performance of U.S. catch share programs (Brinson and Thunberg 2013). Shorter season lengths and TAC overages both indicate a more extreme race to fish, potentially resulting in inefficient capital investment and harvesting effort and lower-valued product, and they therefore imply larger cost and value dimensions of rent dissipation.

Three variables proxy for the transaction costs of catch share program adoption by measuring the number and heterogeneity of resource users. The first variable is the number of active vessels in a fishery, a proxy for the number of resource users.[19]

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[18] TAC overages typically occur when fishing occurs at speed such that fishery managers are not able to effectively close the fishery immediately when the TAC is acheived.

[19] An ideal measure of the number of resource users would be the number of firms active in each fishery. Instead, I assume that the number of vessels is highly correlated with the number of firms.

The second variable is the number of fishery sectors, determined from the North Pacific Fishery Management Council Fishing Fleet Profiles and Groundfish Species Profiles (NPFMC 2011, NPFMC 2012). The third variable is the Gini coefficient (calculated using the Deaton formula and expressed as a percentage), which measures the degree of inequality in the distribution of revenues among active vessels (Deaton 1997). A value of 0% represents a perfectly uniform distribution of revenues among vessels, and value of 100% represents maximal inequality (i.e., one vessel captures all the revenue). The number of sectors and the Gini coefficient measure the level of heterogeneity among users. The larger the value of each of these metrics, the higher the implied transaction costs of catch share program adoption.

Finally, in addition to the variables of interest, I include several control variables. Aggregate catch and average ex-vessel price control for the size and value of the fishery to account for the effects of market conditions on catch share adoption.[20] Management area and fishery management area fixed effects control for unobserved effects across geographic regions and management groupings.

Figure 1.5 presents a Kaplan-Meier survival function estimate, a graphical summary of catch share program adoption among fisheries in the sample. The Kaplan-Meier estimator is a nonparametric estimate of the survival function $S(t)$

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[20] I convert nominal prices to real prices using the Producer Price Index for unprocessed and packaged fish.

where discreet jumps indicate catch share program adoption. At the end of the observation period, the probability of survival is still quite high, 73.5%, reflecting the fact that a majority of fisheries in the sample have not yet adopted catch share programs.

Figure 1.5: Kaplan-Meier estimate for Alaska fisheries data

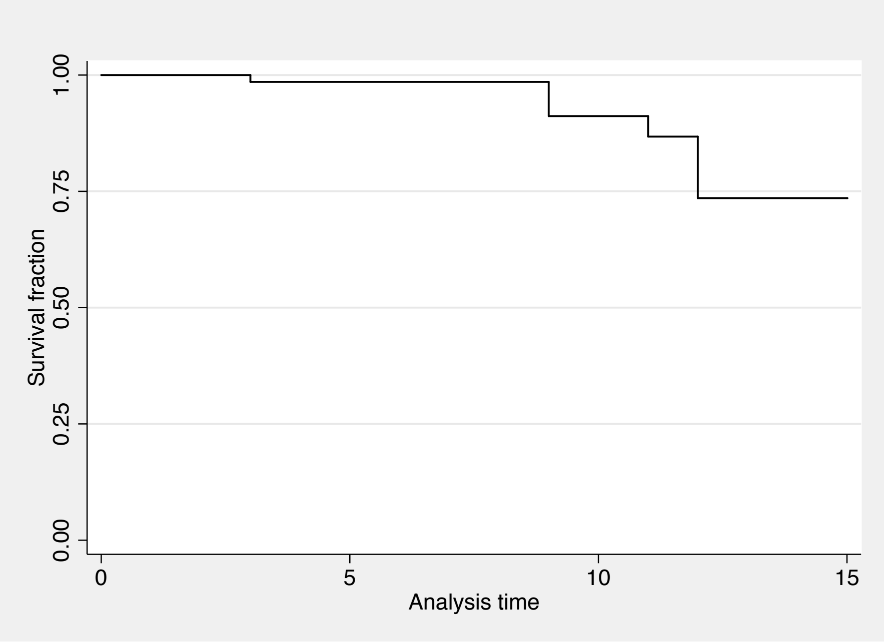

Table 1.3 reports descriptive statistics and differences in means between fisheries that do and do not adopt catch shares during the observation period, calculated using 1996 data, the first year of the analysis period. Columns (1) through (3) report means and standard errors for the entire sample and for fisheries that did and did not adopt catch shares during the observation period,

respectively. Column (4) reports the difference in the means of the catch share and non-catch share fisheries along with the standard error from a t-test of the null hypothesis that this difference is equal to zero. Table 1.3 is consistent with the Cost Dimension, Value Dimension, and Transaction Costs Hypotheses. Fisheries that adopt catch shares tend to have shorter season lengths and are more likely to have TAC overages prior to the transition to rights-based management compared to fisheries that do not adopt catch shares, supporting the idea that catch share program adoption is more likely if the cost and value dimensions of rent dissipation are large. Similarly, fisheries that adopt catch shares tend to have fewer active vessels, fewer sectors, and smaller Gini coefficients relative to fisheries that do not adopt catch shares, supporting the idea that catch share program adoption is more likely if transaction costs are low.

In contrast, Table 1.3 gives limited evidence in support of the Resource Dimension Hypothesis. Although fisheries that adopt catch shares tend to have lower stock status metrics and higher overfishing status metrics prior to the transition compared to fisheries that do not adopt catch shares, neither difference is statistically significant. Additionally, discard and bycatch closure rates are lower among fisheries that transition to catch shares. More generally, the data indicate that resource depletion is not of high concern in most Alaska fisheries, regardless of whether they adopt catch shares. The mean stock status in the full sample is 1.59

(1.49 for fisheries that adopt catch shares), well above the overfished threshold of 0.5, and the mean overfishing indicator is 37.8% (44.8% for fisheries that adopt catch shares), well below the overfishing threshold of 100%. These findings are not necessarily surprising in the Alaska context because catch limits for fisheries in this region have historically been set conservatively to maintain sustainable stocks, and command and control regulations are generally well-enforced (NPFMC 2011).

1.6 Empirical results

1.6.1 Main results

Table 1.4 reports hazard ratios and *t*-statistics corresponding to estimated coefficients from Cox proportional hazards models of catch share program adoption for the full sample. Hazard ratios are exponentiated coefficient estimates, and they measure the change in the probability of catch share program adoption for a one-unit change in the corresponding covariate. A hazard ratio greater (less) than one indicates an increase (decrease) in the probability of catch share program adoption. For example, in Column (3), a 1% increase in overfishing status increases the probability of catch shares adoption by 2.9%, and 1% increase in the discard rate decreases the probability of catch shares adoption by 5.2%.

Columns (1) through (3) report estimates from standard Cox proportional hazards models. Columns (4) through (6) present results from shared frailty versions of models in columns (1) through (3) using the fishery groups described in Section 4.2. Column (3), which controls for unobserved heterogeneity with fishery management plan and management area fixed effects, is the preferred specification. Wald tests reject the null hypotheses that fishery management plan and management area fixed effects are jointly equal to zero at the 5% and 1% levels respectively. The magnitude and statistical significance of the hazard ratios corresponding to the variables of interest are generally similar across columns (1) through (3), but the variation that does exist suggests the importance of accounting for unobserved heterogeneity among fisheries.

The shared frailty models in columns (4) through (6) provide evidence that controlling for unobserved heterogeneity using area and management plan fixed effects is a reasonable approach. The estimated frailty variance, $\hat{\theta}$, in columns (5) and (6) is essentially zero and likelihood ratio tests fail to reject the null hypothesis that θ equals zero, which reduces the model estimates to the standard Cox model estimates reported in columns (2) and (3). In column (4), likelihood-ratio test of the null hypothesis that θ equals zero rejects at the 5% level. I use the Akaike Information Criterion (AIC) to compare the models in columns (3) and (4). The AIC is lower for column (3), which, taken together with models (4) and (5), suggests that management plan and area fixed effects are a reasonable approach to controling for unobserved differences among fisheries.

Focusing on the preferred specification in column (3), there is strong evidence supporting the Cost and Value Dimension and the Transaction Costs hypotheses from the conceptual framework. Consistent with the Cost and Value Dimension hypotheses, a one day decrease in season length increases the probability of catch share program adoption by 3.1%. Similarly, the estimated hazard ratios for the number of vessels, the number of sectors, and the Gini coefficient support the Transaction Costs Hypothesis – greater numbers of and heterogeneity among resource users decrease the likelihood of catch share program adoption. An additional vessel decreases the probability of catch share program adoption by 0.6% (although this value is not statistically significant), an additional sector by 65%, and a one percentage increase in the Gini coefficient by 4.1%. Column (3) gives mixed evidence with respect to the Resource Dimension Hypothesis. Although an increase in the overfishing metric and a bycatch closure both increase the probability of catch share program adoption, a 1% increase in the discard rate *lowers* the probability of catch share program adoption by 5.5%.

One potential issue with the empirical specifications in Table 1.4 is that stock status is omitted because only 47 of the 68 fisheries sampled have formal stock assessments that produce stock status data. To assess the extent to which leaving out stock status leads to omitted variable bias, I examine a restricted sample of fisheries with stock status data. In column (1) of Table 1.5, I reestimate model (3) from Table 1.4 with the restricted sample. The estimated hazard ratios on the variables of interest are generally similar in magnitude and statistical significance to the specification in column (2) that includes stock status, which supports the idea that omitting stock status does not lead to serious omitted variable bias.

Table 1.5 reveals important additional information about the Resource Dimension Hypothesis. In column (2), the estimated hazard ratio for stock status indicates that healthier stocks are more likely to adopt catch shares while fisheries that have higher overfishing status and experience bycatch closures are *also* more likely to adopt. A potential explanation for this mixed evidence for the Resource Dimension Hypothesis is as follows. In regions such as Alaska where harvest levels are set conservatively under command and control regulation to protect the resource stocks, resource managers may be particularly diligent about protecting and maintaining the long-term health of high-value stocks – the same fisheries that are likely to adopt catch shares. In these situations, a more likely vector along which to observe the resource dimension of rent dissipation is in short-term metrics of resource pressure such as annual overfishing status and bycatch rates, which can be high even while maintaining long-term stock health if thresholds for these indicators are set conservatively enough under command and control regulation. Moreover, in this empirical context, command and control regulation appears to be doing an adequate job of preventing long-term resource depletion. Thus, the resource dimension gains from catch share program adoption are likely small, and those that do exist are those associated with mitigating short-term (flow) metrics of resource depletion such as overfishing status and bycatch rates.

I also consider the implications of redefining catch share adoption, so that a fishery is considered to have adopted catch shares regardless of the percentage of the overall TAC that is allocated to a catch share program. This redefinition designates four additional fisheries as having adopted catch shares: 1) the BSAI Pacific cod fishery in the Amendment 80 Cooperatives and 2) the Pacific cod, thornyhead rockfish, and shortraker and rougheye rockfish fisheries in the Central GOA Rockfish Pilot Cooperatives Program. Table 1.6 re-estimates models (1) through (4) in Table 1.4 with this definition of catch shares adoption. Comparatively low pseudo R-squared values from the standard Cox models and high log-likelihood values in all model specifications indicate that the catch shares adoption definition used in Table 1.4 has significantly more explanatory power. From a policy perspective, this result is not surprising. These fisheries were not the main impetus for their respective catch share programs. Rather, species from these fisheries tend to be caught along with the species of the fisheries that were the primary focus of these catch share programs, and they are the main target of other other sectors within the fishery for which catch shares have not (yet) been adopted.

Finally, the proportional-hazards assumption is a central requirement for identification in Cox models. To verify the proportional-hazards assumption, I implement a test based on Schoenfeld residuals for all reported models (Cleves et al. 2010). I find no evidence that the proportional-hazards assumption is violated.

1.6.2 Predicting catch share program adoption

A limitation of the Cox proportional hazards approach is that, since the baseline hazard is not estimated, one cannot use the model to predict future catch share program adoption among fisheries that do not transition to rights-based management during the observation period. A fully parametric model, in contrast, can be used for prediction. Table 1.7 presents a ranked list of fisheries by predicted catch share adoption year from the group of fisheries that did not adopt a catch share program during the observation period, based on a Weibull regression model estimated with the covariates of column (3) in Table 1.4. The Weibull proportional hazards model assumes a baseline hazard of the form $h_0(t) = pt^{(p-1)}\exp(\beta_0)$, which allows for a variety of monotonically increasing or decreasing shapes depending on the value of the estimated parameter p, and it was selected over other common parametric specifications (e.g., exponential, Gompertz) because it had the lowest

AIC score. Table 1.8 reproduces column (3) of Table 1.4 for comparison to the Weibull specification. The hazard ratio estimates from the Weibull model are generally (although not always) of the same direction and of similar magnitude to those of the Cox model.

Predicted years of catch share program adoption were constructed by computing the expected value of the survival time (the integral of the survival function) and plugging in the estimated model parameters and year 2010 covariate values. For each fishery, this produces an estimate of the number of years until catch share program adoption based on conditions at the end of the observation period. Although the estimated year of adoption is based on the assumption of time-constant covariates based on 2010 values, the rank ordering is meaningful if the relative magnitudes of covariate values among fisheries are consistent over time.

Of the 15 fisheries with the shortest predicted times to catch share program adoption listed in Table 1.7, the North Pacific Fishery Management Council is in the process of considering catch share program adoption for 8 of these under a proposed program for the Central Gulf of Alaska groundfish fisheries caught primarily by a fleet of trawl vessels (NPFMC 2014). Additionally, while the Council has not adopted a federally-regulated catch share program BSAI Pacific Cod, an industry-created voluntary cooperative program began in this fishery in August 2010 (Brinson and Thunberg 2013).

These results serve two purposes. First, they provide some indication of internal validity of the empirical model. Fisheries currently under consideration for catch share programs are generally those predicted to adopt catch shares next by the model. Second, they offer policy recommendations for the North Pacific Fishery Management Council insofar as they identify fisheries with appropriate characteristics for future catch share program development and implementation given observed patterns of catch share program adoption in Alaska.

1.7 Conclusion

An emerging literature on the performance of rights-based management policies indicates that these management tools can improve both economic and ecological outcomes in common-pool resources relative to open access and command and control regulation regimes. This paper seeks to explain the determinants of the transition from command and control regulation to rights-based management in common-pool resources through a general conceptual framework of rights-based management adoption and a specific empirical application to a group of federally managed Alaska fisheries.

Consistent with the recent literature on fishery management reform (see, for example, Grainger and Parker 2013), I find that while production inefficiency motivates the transition to rights-based management, transactions costs arising from heterogeneity resource users leads to delay in catch share program adoption. I also find a nuanced relationship between resource depletion and the transition to property rights in the empirical context studied.

There are at least two vectors along which this line of research could be extended and improved. First, the empirical analysis focuses on transaction costs arising from resource users narrowly defined (vessels owners). However, other resource stakeholders including vessel crew members, processors, non-commercial fishery sectors, local communities, and environmental groups often exert important political influence on catch share program adoption, and I do not explicitly account for this in the empirical analysis. For example, the BSAI Crab Rationalization Program was the result of a six year design process undertaken by the North Pacific Fishery Management Council that explicitly attempted to accommodate a variety of stakeholders including vessel owners, captains, crew, processors, and local communities (Fina 2011). Future work could place more emphasis on quantifying the influence of multiple stakeholders in the transition from command and control regulation to rights-based management.

Second, a key finding of the empirical analysis is that production inefficiency under command and control regulation and transactions costs are clearer determinants of the transition to rights-based management relative to resource depletion.

This result is not necessarily surprising in the Alaska context where fishery managers have a long history of commitment to and success in resource conservation, but it would be useful to examine the extent to which this result holds in other institutional contexts.

Finally, the results of this study offer important insights relevant to United States fisheries management policy. NOAA's 2010 Catch Share Policy calls for "the consideration and adoption of catch shares wherever appropriate in fishery management and ecosystem plans and their amendments, and will support the design, implementation, and monitoring of catch share programs," but there has been little formal guidance on which fisheries to focus catch share adoption efforts (NOAA 2010). By systematically identifying characteristics of fisheries where catch shares have been successfully adopted, this analysis provides a basis for distinguishing fisheries in which regulators might target future catch share program adoption efforts.

Table 1.1: Summary of Catch Share Fisheries

Program	Year	Number	Fisheries
BSAI American Fisheries Act Pollock Cooperatives	1999	1	BS pollock
BSAI King & Tanner Crab Rationalization Program	2005	5	Bristol Bay red king crab BS snow crab BS Tanner crab Pribilof Islands red and blue king crab St. Matthew Island blue king crab
Central GOA Rockfish Pilot Cooperatives	2007	3	Central GOA Pacific ocean perch Central GOA Northern rockfish Central GOA pelagic shelf rockfish
BSAI American Fisheries Act Non-Pollock Cooperatives (Amendment 80)	2008	9	BSAI yellowfin sole BSAI rock sole BSAI flathead sole Eastern AI/BS Atka mackerel Central AI Atka mackerel Western AI Atka mackerel Eastern AI Pacific ocean perch Central AI Pacific ocean perch Western AI Pacific ocean perch

Notes: The BSAI Crab Rationalization Program includes three additional fisheries not in the analysis due to limited data availability: Western AI golden king crab, Eastern AI golden king crab, and Western AI red king crab.

Table 1.2: Variables Summary

Category	**Variable**	**Description**	**Expected sign**
Resource dimension of rent dissipation	Stock status	B/B$_{MSY}$ (or equivalent)	-
	Overfishing status	(Aggregate catch/OFL)×100%	+
	Discard rate	Percentage of total catch not retained	+
	Bycatch closure	‘1’ if fishery closed due to bycatch	+
Cost/value dimension of rent dissipation	Season length	Number of days fishery is open	-
	TAC Exceeded	‘1’ if aggregate catch > TAC	+
Transaction costs	Vessels	Number of active vessels	-
	Sectors	Number of sectors	-
	Gini coefficient	Measure of vessel-level revenue equality	-
Controls	Average price	Average ex-vessel price (2010$/lb)	
	Aggregate catch	Aggregate catch (thousand mt)	
	FMP FE	Fishery management plan fixed effects	
	AREA FE	Management area fixed effects	

Table 1.3: Baseline Descriptive Statistics and Differences in Means

	All Fisheries (1)	Catch Share Fisheries (2)	Non-Catch Share Fisheries (3)	Difference (4)
Resource Dimension				
Stock status[†]	1.59	1.49	1.65	-0.16
	(0.09)	(0.16)	(0.12)	(0.20)
Overfishing status	37.78	44.77	35.16	9.60
	(3.09)	(6.11)	(3.54)	(6.88)
Discard rate	21.99	21.89	22.02	-0.13
	(2.12)	(3.63)	(2.59)	(4.85)
Bycatch closure	0.29	0.17	0.34	-0.17
	(0.06)	(0.09)	(0.07)	(0.13)
Cost/Value Dimension				
Season length	157.5	66.3	190.3	-124.1***
	(15.30)	(17.72)	(17.68)	(31.42)
TAC Exceeded	0.21	0.39	0.14	0.25**
	(0.05)	(0.12)	(0.05)	(0.11)
Transaction Costs				
Vessels	120.6	107.1	125.6	-18.42
	(15.03)	(17.30)	(19.51)	(34.25)
Sectors	2.34	1.50	2.64	-1.14***
	(0.18)	(0.12)	(0.23)	(0.39)
Gini coefficient	75.24	61.83	80.07	-18.24***
	(2.15)	(5.81)	(1.62)	(4.38)
Controls				
Average price	0.77	1.33	0.57	0.77**
	(0.15)	(0.46)	(0.10)	0.32
Aggregate catch	28.48	81.10	9.54	71.55*
	(16.55)	(60.53)	(4.89)	(36.75)
Fisheries	68	18	50	

Notes: For each variable, I report the mean and standard error for the full sample and fisheries that do and do not adopt catch share programs using 1996 data, the first year of the sample. The last column shows the difference in the means of catch share and non-catch share fisheries with the standard error of the difference. * $p < 0.10$, ** $p < 0.05$, *** $p < 0.01$.

[†] Stock status is available for 47 of the fisheries sampled.

Table 1.4: Cox Proportional Hazards Model

	Standard Model			Shared Frailty Model		
	(1)	(2)	(3)	(4)[§]	(5)[†]	(6)[#]
Overfishing status	1.008	1.045***	1.029**	0.990	1.045**	1.029
	(0.72)	(2.93)	(2.40)	(-0.54)	(2.02)	(1.15)
Discard rate	0.953**	0.959***	0.948***	0.942	0.959	0.948
	(-2.29)	(-2.60)	(-3.33)	(-1.22)	(-1.20)	(-1.27)
Bycatch closure	16.98**	46.92***	17.95**	2.493	46.92***	17.95*
	(2.31)	(2.96)	(2.29)	(0.71)	(2.76)	(1.78)
Season length	0.967***	0.965***	0.969***	0.982**	0.965***	0.969***
	(-3.23)	(-3.18)	(-4.41)	(-2.09)	(-3.27)	(-3.33)
TAC Exceeded	0.851	0.477	0.732	1.148	0.477	0.732
	(-0.23)	(-1.58)	(-0.59)	(0.15)	(-0.91)	(-0.35)
Vessels	0.992**	0.997	0.994	0.999	0.997	0.994
	(-2.15)	(-1.64)	(-1.27)	(-0.09)	(-0.75)	(-0.91)
Sectors	0.259**	0.252***	0.346***	0.299	0.252**	0.346**
	(-2.34)	(-4.19)	(-3.46)	(-1.39)	(-2.41)	(-2.23)
Gini coefficient	0.956***	0.955***	0.959***	0.970	0.955**	0.959
	(-3.32)	(-2.96)	(-2.91)	(-1.31)	(-2.12)	(-1.61)
Average price	2.611***	0.912	1.416	1.846	0.912	1.416
	(3.76)	(-0.38)	(0.93)	(1.29)	(-0.18)	(0.57)
Aggregate catch	1.026***	1.016***	1.013***	1.012	1.016**	1.013**
	(4.15)	(3.99)	(5.79)	(1.05)	(2.44)	(2.36)
FMP FE		YES	YES		YES	YES
Area FE			YES			YES
Observations	914	914	914	914	914	914
Fisheries	68	68	68	68	68	68
Catch Share Programs	18	18	18	18	18	18
Log-likelihood	-36.06	-31.85	-28.13	-34.51	-31.85	-28.13
AIC	92.12	87.71	82.26	89.01	87.71	82.26
Pseudo R-squared	0.507	0.564	0.615			

Hazard ratios reported; *t*-statistics in parentheses; * $p < 0.10$, ** $p < 0.05$, *** $p < 0.01$.

Efron approximation for tied failures.

[§] The estimated frailty variance is $\theta = 7.15$ (SE = 6.46). The likelihood-ratio test of $H_0 : \theta = 0$ has p-value = 0.039.

[†] The estimated frailty variance is $\theta = 2.55\text{E-}18$ (SE = 2.08E-14). The likelihood-ratio test of $H_0 : \theta = 0$ has p-value = 0.500.

[#] The estimated frailty variance is $\theta = 2.08\text{E-}18$ (SE = 3.99E-14). The likelihood-ratio test of $H_0 : \theta = 0$ has p-value = 0.500.

Table 1.5: Cox Proportional Hazards Model: Assessed Fisheries

	(1)	(2)
Stock status		11.21***
		(6.54)
Overfishing	1.023*	1.035***
	(1.84)	(3.02)
Discard rate	0.973	0.968
	(-1.30)	(-1.51)
Bycatch closure	11.90**	4.613***
	(2.50)	(2.72)
Season length	0.979***	0.973***
	(-2.69)	(-4.11)
TAC Exceeded	0.853	0.600
	(-0.40)	(-1.43)
Sectors	0.467***	0.552***
	(-2.96)	(-2.84)
Vessels	0.995*	0.990***
	(-1.73)	(-4.10)
Gini coefficient	0.971**	0.976*
	(-2.35)	(-1.94)
Average price	1.246	0.978
	(0.83)	(-0.11)
Aggregate catch	1.014***	1.019***
	(8.50)	(7.29)
FMP FE	YES	YES
Area FE	YES	YES
Observations	612	612
Fisheries	47	47
Catch Share Programs	18	18
Log-likelihood	-36.93	-34.48
Pseudo R-squared	0.444	0.481

Hazard ratios reported; *t*-statistics in parentheses.

* $p < 0.10$, ** $p < 0.05$, *** $p < 0.01$.

Breslow approximation for tied failures.

Table 1.6: Cox Proportional Hazards Model: Catch Shares Redefinition

	Standard Model			Shared Frailty Model
	(1)	(2)	(3)	(4)§
Overfishing status	0.999	1.007	0.995	0.979
	(-0.09)	(0.82)	(-0.54)	(-1.11)
Discard rate	0.981**	0.980*	0.977	0.961
	(-2.09)	(-1.83)	(-1.56)	(-1.11)
Bycatch closure	1.111	1.357	0.890	0.261
	(0.14)	(0.39)	(-0.11)	(-1.27)
Season length	0.991*	0.991*	0.988**	0.993
	(-1.83)	(-1.89)	(-2.06)	(-1.12)
TAC Exceeded	1.538	0.957	1.071	0.727
	(0.83)	(-0.10)	(0.15)	(-0.39)
Vessels	0.999	1.002	1.001	1.008*
	(-0.24)	(1.04)	(0.62)	(1.87)
Sectors	0.942	0.867	0.999	4.956**
	(-0.30)	(-0.93)	(-0.01)	(2.26)
Gini coefficient	0.984	0.992	0.985	0.976
	(-1.55)	(-0.75)	(-1.09)	(-1.24)
Average price	1.817***	0.886	1.225	1.129
	(3.61)	(-0.53)	(0.82)	(0.30)
Aggregate catch	1.012	1.007***	1.007***	1.008
	(1.48)	(4.97)	(4.63)	(0.98)
FMP FE		YES	YES	
Area FE			YES	
Observations	899	899	899	899
Fisheries	68	68	68	68
Catch Share Programs	22	22	22	22
Log-likelihood	-64.69	-60.39	-51.72	-56.85
AIC	149.4	144.8	129.4	133.7
Pseudo R-squared	0.269	0.318	0.416	

Hazard ratios reported; *t*-statistics in parentheses; * $p < 0.10$, ** $p < 0.05$, *** $p < 0.01$.
Efron approximation for tied failures.

§ The estimated frailty variance is $\theta = 10.75$ (SE = 5.84). The likelihood-ratio test of $H_0 : \theta = 0$ has p-value <0.01.

Table 1.7: Predicted Transitions to Catch Shares

Rank	**Fishery**	**Predicted Adoption Year**	**Considering Catch Shares?**
1	Norton Sound red king crab	2007	No
2	Central GOA (Kodiak) pollock	2010	Yes
3	Central GOA (Chirkiof) pollock	2010	Yes
4	Central GOA Pacific cod	2011	Yes
5	BS Pacific Ocean perch	2011	No
6	AI Greenland turbot	2011	No
7	BSAI Pacific cod	2013	No*
8	AI pollock	2022	No
9	Central GOA rex sole	2024	Yes
10	BSAI arrowtooth flounder	2024	No
11	Central GOA deep-water flatfish	2027	Yes
12	Central GOA shallow-water flatfish	2027	Yes
13	Central GOA flathead sole	2028	Yes
14	Central GOA arrowtooth flounder	2031	Yes
15	Pribilof Islands golden king crab	2032	No

Notes: Predicted mean year of catch share program adoption computed from estimated model parameters from a Weibull regression model and 2010 covariate values.

*Voluntary cooperative began in August 2010.

Table 1.8: Weibull vs. Cox Models

	Cox (1)	Weibull (2)
Overfishing	1.029**	0.975
	(2.40)	(-1.64)
Discard rate	0.948***	0.964
	(-3.33)	(-1.64)
Bycatch closure	17.95**	8.233*
	(2.29)	(1.93)
Season length	0.969***	0.969**
	(-4.41)	(-2.09)
TAC Exceeded	0.732	0.877
	(-0.59)	(-0.28)
Vessels	0.994	1.001
	(-1.27)	(0.27)
Sectors	0.346***	0.489**
	(-3.46)	(-2.57)
Gini coefficient	0.959***	0.969
	(-2.91)	(-1.54)
Average price	1.416	1.227
	(0.93)	(0.33)
Aggregate catch	1.013***	1.014***
	(5.79)	(4.45)
FMP FE	YES	YES
Area FE	YES	YES
Observations	914	914
Fisheries	68	68
Catch Share Programs	18	18
Log-likelihood	-28.13	2.667
Pseudo R-squared	0.615	

Hazard ratios reported; *t*-statistics in parentheses;
* $p < 0.10$, ** $p < 0.05$, *** $p < 0.01$.
Efron approximation for tied failures.

Chapter 2

Marine Reserves as a Cooperation Mechanism in Transboundary Fisheries

2.1 Introduction

An important challenge in achieving sustainable fisheries is the establishment of effective management in transboundary fisheries.[1] Broadly defined, the term transboundary fishery refers to a stock shared by two or more jurisdictions at any political scale. Possible management contexts include a stock that occurs in the Exclusive Economic Zones (EEZs) of two or more countries or a stock managed with Territorial User Rights for Fisheries (TURFs), spatial rights to fishing grounds. Transboundary stocks are vulnerable to overexploitation and economic inefficiency because spatial connectivity across jurisdictional boundaries

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[1] This chapter is co-authored with Chris Costello, Bren School of Envrionmental Science and Management, UC Santa Barbara.

creates an externality in resource extraction. In this paper, we investigate whether the establishment of marine reserves, areas in which fishing is prohibited, in transboundary fisheries can mitigate spatial externalities and achieve first-best economic outcomes.

Economic theory of transboudary fisheries is well known. In the absence of coordination among jurisdictions sharing a fish stock, competition among jurisdictions for rents leads to suboptimal resource extraction and economic outcomes (Levhari and Mirman 1980). Further, the ability of jurisdictions to effectively bargain with one another to promote cooperation significantly improves equilibrium outcomes (Munro 1979). Applied work supports theoretical predictions. Theoretical and empirical research on specific transboundary fisheries such as the Arcto-Norwegian cod stock finds economic losses from non-cooperation between countries consistent with generalized theory (see Sumaila 1997, Armstrong and Sumaila 2000, Armstrong and Sumaila 2001). Additionally, a recent global study indicates that transboundary fish stocks shared by multiple countries are systematically more exploited compared to those completely controlled by a single nation (McWhinnie 2009).

The efficient management of a transboundary fishery requires coordination between jurisdictions to achieve socially optimal harvest levels (Munro et al. 2004). One approach to cooperation is the implementation of cooperative harvest agreement between the jurisdictions sharing the fish stock. Some transboundary fisheries have been effectively managed this way. For example, the Pacific halibut stock shared between the United States and Canada has been jointly managed via international treaty since 1923. Under the treaty, both countries set annual harvest limits in their respective jurisdictions based on the recommendations of the International Pacific Halibut Commission (IPHC), a bilateral fisheries management organization. The implementation of rights-based management programs in both the U.S. and Canadian portions of the fishery during the 1990s further improved management outcomes (Grafton et al. 2000, Brinson and Thunberg 2013).

Cooperative harvest agreements can achieve socially optimal catch levels, but may prove costly to negotiate, monitor, and enforce (Barrett 2003). Apart from costs of negotiating an initial agreement, cooperative harvest agreements require active management that can be costly to maintain. The IPHC, for example, has an annual budget of $4.9 million for ‘core activities’, not including research and stock assessments required to make annual harvest allocation decisions (McCreary and Brooks 2012). Cooperative harvest agreements also require institutional resources and infrastructure to monitor and enforce harvest allocations. For these reasons, cooperative harvest agreements are not widely observed in practice.

This paper considers an alternative approach to coordination in transboundary fisheries, the implementation of marine reserves. Our objective is to determine whether marine reserve implementation can result in first-best economic outcomes equivalent to cooperative resource extraction. While we do not explicitly model costs of implementing a cooperative harvest agreement vs. marine reserves, the implication is that if marine reserves can produce economic outcomes equal or close to those resulting from cooperative extraction that there may be contexts in which marine reserves are a practical coordination mechanism.

The paper proceeds as follows. In Section 2 we provide background on the use of marine reserves as a fishery management tool. In Sections 3-6, we develop a game theoretic model of resource extraction in a transboundary fishery shared by two jurisdictions and compare steady-state equilibrium fishery profit levels under three management scenarios: 1) *cooperative extraction* in which each jurisdiction chooses a harvest rate to maximize total fishery profits, 2) *non-cooperative extraction* in which each jurisdiction chooses a harvest rate to maximize own profits, and 3) *marine reserve implementation* in which each jurisdiction commits an equal fraction of fishery area to marine reserves and harvests non-cooperatively in the remaining open area. Under a set of common biological assumptions, we show that when marginal harvest costs are independent of stock density, optimally-sized marine reserves achieve economic outcomes equivalent to those obtained under cooperative extraction. We also find that for a range of stock dependent harvest cost functions, optimally-sized marine reserves can improve economic outcomes relative to the non-cooperative equilibrium. In Section 7, we illustrate our theoretical predictions with a numerical application to the Chilean loco fishery. Section 8 offers conclusions and ideas for further work.

2.2 Marine reserves as a fisheries management tool

Marine reserves have emerged as a widely-used management tool to protect marine ecosystems and improve fishery outcomes. A rapidly expanding body of theoretical and empirical literature in ecology and economics investigates outcomes associated with establishing marine reserves. The ecology literature focuses primarily on the efficacy of marine reserves for improving biological outcomes including stock biomass and density, organism size, and species diversity both inside (Halpern 2003, Lester et al. 2009) and outside (Halpern et al. 2010) reserves and generally finds positive outcomes of varying magnitudes.

A significant bioeconomic literature uses integrated population and economic models to examine the circumstances under which establishing marine reserves is an economically optimal fishery management strategy. Conceptually, reserves are economically optimal when the value derived from biological spillover from closed areas outweighs the value of fishing these areas. This line of research focuses on the effects of introducing marine reserves on fishery yields and profits under various status quo management regimes including open access (e.g., Sanchirico and Wilen 2001b, Sanchirico 2005), limited entry (e.g., Holland and Brazee 1996, Sanchirico and Wilen 2002b, Smith and Wilen 2003, Sanchirico 2004), sole ownership (e.g., Sanchirico et al. 2006, Costello and Polasky 2008). An important conclusion from this literature relevant to our work is that marine reserves are more likely to improve economic outcomes in fisheries where the stock is overexploited under the status quo management regime.

Several existing papers consider the economic outcomes of establishing marine reserves in a transboundary fishery. Ruijs and Janmaat (2007) develop a differential game using a bioeconomic metapopulation model of a transboundary fishery to analyze the effect of marine reserve implementation on fishery biomass and profits. The analysis relies on a two-country, six-patch numerical example in which each jurisdiction implements a marine reserve in one patch under the following management regimes: (i) cooperation, in which jurisdictions jointly choose the combination of marine reserve placements and effort levels open patches to maximize profits, and (ii) non-cooperation, in which jurisdictions choose marine reserve placement and effort levels individually to optimize their own net benefits given the effort levels of the other nation. The main result is that the non-cooperative regime always results in lower profits and biomass levels compared to the cooperative regime regardless of biological growth and connectivity scenarios. Costello and Kaffine (2009) present a numerical model of a stylized California kelp bass fishery managed with 48 hypothetical TURFs and compare outcomes of marine reserve implementation in a cooperative and non-cooperative system to cooperative and non-cooperative outcomes in the absence of marine reserves. They find that whether marine reserves can increase profits and biomass depends critically on the level of coordination already occurring between TURF owners. Finally, Molina et al. (2014) modify the two jurisdiction model of Levhari and Mirman (1980) to study the effects of different types of spatial connectivity between jurisdictions on equilibrium marine reserve outcomes.

Our paper contributes to the existing literature on marine reserves and transboundary fisheries in two important ways. First, we examine how the structure of harvest costs, particularly the degree to which marginal harvest costs depend on underlying stock density, affects the benefits from marine reserve implementation. Second, we investigate how the number of jurisdictions sharing the stock affects the potential gains from implementing marine reserves in a non-cooperative setting. The next section introduces our theoretical model, which extends the model of Hastings and Botsford (1999) to a game theoretic setting.

2.3 A basic model of a transboundary fishery

A fish stock spans the boundaries of two jurisdictions, $j \in \{A, B\}$.[2] Let $\alpha \in (0,1)$ be the proportion of total fishery area that lies in jurisdiction A. For ease of exposition, we normalize the area of the fishery to 1 so that the number and density of adults in the entire fishery are equal. We make the following biological assumptions:

Assumption 1. *Adults are sedentary.*[3]

Assumption 2. *Larvae mix uniformly throughout the fishery.*

Assumption 3. *The density of juveniles that are successfully recruited to the adult population at any location is given by the function $f(d)$ where d is the density of juveniles attempting to settle, $f'(d) > 0$, $f''(d) < 0$, and $f(0) = 0$.*

The implication of Assumptions 1 and 2 is that all spatial connectivity in the stock occurs via larval dispersal. In the ecology literature, the larval dispersal structure described in Assumption 2 is called a common larval pool. Under Assumption 2, larvae have an equal probability of setting at any location in the fishery regardless of origin. Assumption 3 describes the nature of density-dependence in the population dynamics. In particular, the density of juvenile

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[2] Here we model a two jurisdiction scenario. We plan to extend the model to $J > 2$.

[3] For our purposes, 'sedentary' does not necessarily mean immobile. Rather, following Hilborn et al. (2004), we define sedentary organisms as "those whose movements are short-range when compared to the spatial scale of the fishing process (fleet displacement) and/or pelagic larval dispersal."

individuals that successfully reach the adult stage at a given location is an increasing concave function of the density of juveniles transitioning to the adult stage at that location.

The stock grows according to the following biological rules. Fish reach maturity at age k, at which point they produce m settling juveniles annually. Adult fish are subject to an annual natural survival rate s. The density of adults vulnerable to harvest in period t is the sum of the density of juveniles reaching maturity in the current period and the density of adults that escaped harvest in the previous period and survive natural mortality. Under Assumptions 2 and 3, the density of juveniles successfully reaching the adult population in period t is uniform throughout the fishery and is a function of the sum of the number of adults in jurisdiction A in time $t-k$, $\alpha n_{A,t-k}$, and the number of adults in jurisdiction B in time $t-k$, $(1-\alpha)n_{B,t-k}$, multiplied by the number of juveniles produced by each adult. We assume that each jurisdiction harvests a fixed fraction $H_j \in [0,1]$ of adults in its area. The density of adults at the beginning of period $t+1$ in jurisdiction j, $n_{j,t+1}$, is equal to the fraction of the stock that escapes harvest multiplied by the density of adults in period t

$$n_{j,t+1} = (1-H_j)[f(m(\alpha n_{A,t-k} + (1-\alpha)n_{B,t-k})) + sn_{j,t}]. \tag{2.1}$$

In the analysis that follows, we implicitly assume a zero discount rate and focus on steady-state equilibrium outcomes. In equilibrium, the density of adults in jurisdiction A satisfies

$$n_A = (1 - H_A)[f(m(\alpha n_A + (1 - \alpha)n_B)) + sn_A], \tag{2.2}$$

and the density of adults in jurisdiction B satisfies

$$n_B = (1 - H_B)[f(m(\alpha n_A + (1 - \alpha)n_B)) + sn_B]. \tag{2.3}$$

Since the area of the fishery is 1, the number and density of adults in the entire fishery is the same. Therefore, total fishery stock density, n, is the sum of the number of adults in jurisdiction A and the number of adults in jurisdiction B given by

$$n = \alpha n_A + (1 - \alpha)n_B. \tag{2.4}$$

Equilibrium yield in each jurisdiction is found by multiplying the density harvested by the proportion of total fishery area contained in the jurisdiction. Equilibrium yield in jurisdiction A is given by

$$Y_A = \alpha H_A[f(m(\alpha n_A + (1 - \alpha)n_B)) + sn_A], \tag{2.5}$$

and equilibrium yield in jurisdiction B is given by

$$Y_B = (1-\alpha)H_B[f(m(\alpha n_A + (1-\alpha)n_B)) + sn_B]. \tag{2.6}$$

We assume that the fishery under consideration is small relative to the total market and that the quality of the resource is spatially homogeneous, implying an exogenous ex-vessel price p that is identical across jurisdictions. We also assume that jurisdictions have identical marginal fishing costs. Annual profits in jurisdiction A are given by

$$\pi_A = pY_A - \alpha \int_{n_A^{post}}^{n_A^{pre}} \frac{\theta}{n} dn, \tag{2.7}$$

where $n_A^{pre} = f(m(\alpha n_A + (1-\alpha)n_B)) + sn_A$ and $n_A^{post} = n_A$ are the pre- and post-harvest adult densities in jurisdiction A. Annual profits in jurisdiction B are given by

$$\pi_B = pY_B - (1-\alpha) \int_{n_B^{post}}^{n_B^{pre}} \frac{\theta}{n} dn, \tag{2.8}$$

where $n_B^{pre} = f(m(\alpha n_A + (1-\alpha)n_B)) + sn_B$ and $n_B^{post} = n_B$ are the pre- and post-harvest adult densities in jurisdiction B. The cost function captures harvesting costs associated with the stock density level and the amount harvested. The parameter $\theta \geq 0$ captures the degree to which harvest costs depend on stock density. This ‘stock effect’ can be interpreted as the density at which it becomes unprofitable to fish.[4] Multiplying by the proportion of fishery area in each jurisdiction scales costs to the number of fish harvested. For some results, we will also invoke the following assumption about harvest costs:

Assumption 4. *Marginal harvest costs are independent of stock density.*

This assumption implies that the stock does not become more costly to harvest as it becomes more scarce. We represent this assumption by setting $\theta = 0$.

2.4 Benchmark management scenarios

Before considering the implementation of marine reserves, we introduce two benchmark management scenarios: cooperative extraction and non-cooperative extraction. We use outcomes under these management scenarios as a basis of comparison for marine reserve implementation. Throughout, we assume both jurisdictions have complete, symmetric information, and there is no uncertainty.

2.4.1 Cooperative extraction

We first consider a *cooperative extraction* management scenario in which jurisdictions successfully form a cooperative harvest agreement. The objective of

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[4] Clark and Munro (1975) establish the concept of the marginal stock effect in which the unit harvest cost is inversely proportional to the stock size.

the agreement is to maximize the total economic value of the fishery. Under cooperative extraction, jurisdictions implement a harvest rates that maximize total fishery profits subject to the evolution of stock in both jurisdictions. The objective function is given by

$$\pi^*(H_A, H_B) = \max_{H_A \in [0,1], H_B \in [0,1]} \pi_A(H_A, H_B) + \pi_B(H_A, H_B), \tag{2.9}$$

which is subject to the biological constraints Eqs. 2.2 and 2.3. Differentiating with respect to the harvest fractions gives the following necessary conditions for an interior solution

$$\frac{\partial \pi_A(H_A, H_B)}{\partial H_j} + \frac{\partial \pi_B(H_A, H_B)}{\partial H_j} + \lambda_j((f(m(\alpha n_A + (1-\alpha)n_B)) + sn_j) = 0 \quad \forall j \tag{2.10}$$

where λ_j is the Lagrange multiplier for the equation of motion for stock in jurisdiction j. Because jurisdictions coordinate harvest decisions to maximize the joint profits of both jurisdictions, the spatial externality created by larval dispersal across jurisdictional boundaries is fully internalized.

2.4.2 Non-cooperative extraction

We also consider the benchmark management scenario of *non-cooperative extraction* in which each jurisdiction chooses a harvest rate to maximize its own profits taking the harvest behavior of the other jurisdiction as given. The objective function of jurisdiction j is given by

$$\pi_j^{NC}(H_j, H_{-j}) = \max_{H_j \in [0,1]} \pi_j(H_j, H_{-j}), \tag{2.11}$$

which is subject to the biological constraints Eqs. 2.2 and 2.3. Differentiating with respect to harvest fraction gives the following necessary condition for an interior solution

$$\frac{\partial \pi_j(H_A, H_B)}{\partial H_j} + \lambda_j((f(m(\alpha n_A + (1-\alpha)n_B)) + sn_j) = 0 \tag{2.12}$$

where λ_j is the Lagrange multiplier for the equation of motion for stock in jurisdiction j.

2.4.3 Benchmark results

Lemma 1. *Under Assumptions 1-4, there exists a unique total fishery stock density n^* that maximizes total fishery profits under cooperative extraction, and any pair of jurisdiction harvest rates that obtain n^* will maximize total fishery profits.*

Proof. Under Assumption 4, total fishery profits are the sum of jurisdiction revenues

$$\pi = pY_A + pY_B. \tag{2.13}$$

Substituting Eqs. 2.5 and 2.6 in for Y_A and Y_B gives

$$\begin{aligned}\pi &= p\alpha H_A[f(m(\alpha n_A + (1-\alpha)n_B)) + sn_A] \\ &\quad + p(1-\alpha)H_B[f(m(\alpha n_A + (1-\alpha)n_B)) + sn_B]. \qquad (2.14)\end{aligned}$$

Under cooperative extraction, jurisdictions choose harvest rates, H_A and H_B, jointly to maximize Eq. 2.14 subject to Eqs. 2.2 and 2.3. Solving Eqs. 2.2 and 2.3 for H_A and H_B and substituting into Eq. 2.14 gives the following unconstrained optimization problem for jurisdictions choosing stock densities to maximize total fishery profits:

$$\begin{aligned}\pi^* &= \max_{n_A, n_B} \quad p\alpha[f(m(\alpha n_A + (1-\alpha)n_B)) + sn_A - n_A] \\ &\quad + \; p(1-\alpha)[f(m(\alpha n_A + (1-\alpha)n_B)) + sn_B - n_B]. \qquad (2.15)\end{aligned}$$

Rearranging the total fishery profit equation gives

$$\pi = p[f(m(\alpha n_A + (1-\alpha)n_B)) + s(\alpha n_A + (1-\alpha)n_B) - (\alpha n_A + (1-\alpha)n_B]. \quad (2.16)$$

Using Eq. 2.4, we can now rewrite total fishery profits as a function of total fishery stock density,

$$\pi(n) = p[f(mn) + sn - n]. \qquad (2.17)$$

Eq. 2.17 establishes that we will obtain identical total fishery profits from every pair of jurisdiction harvest rates, (H_A, H_B), that solve Eqs. 2.2 and 2.3 for a pair of jurisdiction stock densities, (n_A, n_B), that result in the same total stock density, n, in Eq. 2.4. That is, the total fishery stock density rather than the location of harvest determines total fishery profits. Eq. 2.17 is a strictly concave function of n. Therefore, $n^* = argmax\{\pi(n)\}$ is a unique maximum, and any pair of jurisdiction harvest rates that result in a pair of jurisdiction stock densities that gives n^* will maximize total fishery profits. $\square$

Lemma 1 establishes that under the assumption of stock independent harvesting costs, the *location* of harvest does not affect total fishery profits; there are multiple combinations of jurisdiction harvest rates that maximize fishery profits. Let $C = \{(H_A^*, H_B^*) : n_A^* = (1 - H_A^*)[f(m(\alpha n_A^* + (1-\alpha)n_B^*)) + sn_A^*]$, $n_B^* = (1 - H_B^*)[f(m(\alpha n_A^* + (1-\alpha)n_B^*)) + sn_B^*]$, $n^* = \alpha n_A^* + (1-\alpha)n_B^*\}$ be the set of jurisdiction harvest rate pairs that obtain the profit maximizing total fishery stock density, n^*, under cooperative extraction. To verify that non-cooperative extraction will not attain maximal fishery profits, we must show that the pair of harvest rates jurisdictions choose under non-cooperative extraction does not belong to the set of fishery profit maximizing harvest rate pairs. To prove this result requires the following lemma:

Lemma 2. *H_B^* is strictly decreasing in H_A^*, $\frac{\partial H_B^*}{\partial H_A^*} < 0$.*

Proof. Substituting the jurisdiction stock density constraints into the total fishery stock density constraint and solving for H_B^* gives the following equation relating the fishery profit maximizing harvest rates in jurisdiction A and jurisdiction B

$$H_B^* = \frac{\alpha(1 - H_A^*)[f(m(\alpha n_A^* + (1-\alpha)n_B^*)) + sn_A^*] + (1-\alpha)[f(m(\alpha n_A^* + (1-\alpha)n_B^*)) + sn_B^*] - n^*}{(1-\alpha)[f(m(\alpha n_A^* + (1-\alpha)n_B^*)) + sn_B^*]}. \tag{2.18}$$

The partial derivative of Eq. 2.18 with respect to the profit maximizing harvest rate in jurisdiction A is negative

$$\frac{\partial H_B^*}{\partial H_A^*} = \frac{-\alpha[f(m(\alpha n_A^* + (1-\alpha)n_B^*)) + sn_A^*]}{(1-\alpha)[f(m(\alpha n_A^* + (1-\alpha)n_B^*)) + sn_B^*]} < 0. \tag{2.19}$$

Therefore, H_B^* is a strictly decreasing function of H_A^*. ☐

Lemma 2 establishes that given a pair of jurisdiction harvest rates that maximize total fishery profits, if jurisdiction A deviates from its chosen harvest rate, then jurisdiction B must also change its harvest rate to maintain maximal fishery profits. That is, for any harvest rate in jurisdiction B, there is only one corresponding harvest rate in jurisdiction A for which fishery profits are maximized.

Proposition 1. *For any transboundary fishery satisfying Assumptions 1-4, total fishery profits under cooperative extraction are strictly greater than total fishery profits under non-cooperative extraction ($\pi^* > \pi^{NC}$).*

Proof. Let $(H_A^*, H_B^*) \in C$ be a pair of jurisdiction harvest rates that maintain the profit-maximizing total stock density, n^*, under cooperative extraction. Let (H_A^{NC}, H_B^{NC}) be a pair of jurisdiction harvest rates that emerge under non-cooperative extraction. By Lemma 2, to prove that total fishery profits under cooperative extraction are greater than total fishery profits under non-cooperative extraction it is sufficient to show that $H_A^{NC} \neq H_A^*$ when the harvest fraction in jurisdiction B under non-cooperative extraction is fixed at the cooperative extraction level ($H_B^{NC}{=}H_B^*$). From the first order necessary conditions for cooperative and non-cooperative extraction, $H_A^{NC}{=}H_A^*$ if and only if $\frac{\partial \pi_B(H_A, H_B^*)}{\partial H_A} = 0$ but $\frac{\partial \pi_B(H_A, H_B^*)}{\partial H_A} < 0$ for all $H_A > 0$, so $H_A^{NC} \neq H_A^*$. $\square$

Under cooperative extraction, jurisdictions will implement a pair of harvest rates from C to maximize fishery profits.[5] Proposition 1 shows that under non-cooperative extraction, jurisdictions will choose a pair of harvest rates that are not an element of the set of fishery profit maximizing jurisdiction harvest rate pairs. Thus, the total fishery profits that emerge under non-cooperative extraction will be lower than the profits that emerge under cooperative extraction.

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[5] Though not formally proven here, under cooperative extraction, first-best profits are obtained.

2.5 Marine reserve implementation

Thus far, we have shown that total fishery profits under non-cooperative extraction will fail to reach the first-best economic outcome that is obtained under cooperative extraction. We now examine fishery profits under an alternative coordination mechanism called the *marine reserve implementation* management scenario in which jurisdictions each agree to protect an equal proportion, $r \in (0, 1)$, of their fishery area in a no-take marine reserve. Each jurisdiction then non-cooperatively chooses a harvest rate in its remaining open fishery area to maximize own profits. In this scenario, the density of adults in jurisdiction j at time $t + 1$ is given by

$$n_{j,t+1} = (1 - H_j)[f(m(\alpha(1 - r)n_{A,t-k} + (1 - \alpha)(1 - r)n_{B,t-k} + rn_{R,t-k})) + sn_{j,t}]. \tag{2.20}$$

As in Eq. 2.1, the density of adults in period $t+1$ in jurisdiction j, is expressed as the fraction of the stock that escapes harvest in period t multiplied by the density of adults in period t. Now the density of juveniles successfully recruited to the adult population, $f(\cdot)$, is a function of the sum of the number of adults in harvestable area of jurisdiction A in time $t - k$, $\alpha(1 - r)n_{A,t-k}$, the number of adults in harvestable area of jurisdiction B in time $t - k$, $(1 - \alpha)(1 - r)n_{B,t-k}$, and the number of adults in marine reserves in time $t - k$, $rn_{R,t-k}$, multiplied by the number of juveniles produced by each adult, m. In equilibrium, the density of adults in jurisdiction A satisfies

$$n_A = (1 - H_A)[f(m(\alpha(1-r)n_A + (1-\alpha)(1-r)n_B + rn_R)) + sn_A]. \tag{2.21}$$

Similarly, the density of adults in jurisdiction B satisfies

$$n_B = (1 - H_B)[f(m(\alpha(1-r)n_A + (1-\alpha)(1-r)n_B + rn_R)) + sn_B]. \tag{2.22}$$

The adult density in marine reserves in equilibrium satisfies

$$n_R = f(m(\alpha(1-r)n_A + (1-\alpha)(1-r)n_B + rn_R)) + sn_R. \tag{2.23}$$

Total fishery stock density is now given by the sum of the number of adult fish in jurisdiction A, the number of adult fish in jurisdiction B, and the number of fish in marine reserves

$$n = \alpha(1-r)n_A + (1-\alpha)(1-r)n_B + rn_R. \tag{2.24}$$

As in the non-reserve scenarios, yield in each jurisdiction is found by multiplying the harvest rate by the fishery area contained in the jurisdiction. Equilibrium yield in jurisdiction A is given by

$$Y_A = \alpha(1-r)H_A[f(m(\alpha(1-r)n_A + (1-\alpha)(1-r)n_B + rn_R)) + sn_A], \quad (2.25)$$

and equilibrium yield in jurisdiction B is given by

$$Y_B = (1-\alpha)(1-r)H_B[f(m(\alpha(1-r)n_A + (1-\alpha)(1-r)n_B + rn_R)) + sn_B]. \quad (2.26)$$

Profits in the marine reserve are given by

$$\pi_A = pY_A - \alpha(1-r)\int_{n_A^{post}}^{n_A^{pre}} \frac{\theta}{n} dn, \quad (2.27)$$

and

$$\pi_B = pY_B - (1-\alpha)(1-r)\int_{n_B^{post}}^{n_B^{pre}} \frac{\theta}{n} dn, \quad (2.28)$$

which accounts for the fact that harvest costs now must be scaled by the fishery area in each jurisdiction outside reserves. In this scenario, the objective function for jurisdiction j is given by

$$\pi_j^R(H_j, H_{-j}, r) = \max_{H_j \in [0,1]} \pi_j(H_j, H_{-j}, r) \quad (2.29)$$

which is subject to the biological constraints Eqs. 2.21, 2.22 and 2.23.

Proposition 2. *For any transboundary fishery satisfying Assumptions 1-4, there exists an optimal reserve fraction, r^*, such that total fishery profits under marine reserve implementation equal total fishery profits under cooperative extraction ($\pi^{R^*} = \pi^*$).*

Proof. By Lemma 1, total fishery stock density determines total fishery profits. Thus, equivalence in total fishery profits between marine reserve implementation and cooperative extraction requires a reserve size that induces countries to non-cooperatively choose harvest rates in their remaining open area that result in n^*. Suppose $H_A^R = H_B^R = 1$ so that jurisdictions harvest all adult biomass in their respective jurisdictions in each period so that there is no post-harvest stock density outside reserves in either jurisdiction, $n_A = n_B = 0$. Post-harvest fishery stock density now consists entirely of reserve biomass

$$n = rn_R, \tag{2.30}$$

and the equation for equilibrium reserve density reduces to

$$n_R = f(mrn_R) + sn_R. \tag{2.31}$$

Substituting n into Eq. 2.31 and solving for r gives the following expression for reserve size in terms of total fishery stock density and model parameters

$$r = \frac{n(1-s)}{f(mn)}. \tag{2.32}$$

Let r^* be the reserve fraction that solves Eq. 2.32 for n^*. To confirm that $(H_A^R, H_B^R, r) = (1, 1, r^*)$ is an equilibrium in the marine reserve implementation scenario, we must show two things. First, we must show that $0 < r^* < 1$. That is, there is some non-zero reserve fraction that gives n^* in Eq. 2.32. The numerator of Eq. 2.32 is the equilibrium natural mortality rate of individuals, and the denominator of Eq. 2.32 is equilibrium recruitment rate of individuals. Since both expressions are positive for any $n > 0$ and the equilibrium recruitment rate of individuals must be greater than the equilibrium natural mortality rate of individuals when there is fishing mortality in the fishery, $0 < r^* < 1$. Second, we must show that a harvest rate equal to 1 is a best response for each jurisdiction at r^*. At the fishery profit-maximizing stock density, n^*, jurisdiction-level profits are increasing in own harvest rate, $\frac{\partial \pi_j(H_j, H_{-j})}{\partial H_j} > 0$. Thus, neither jurisdiction has an incentive to lower its harvest rate and $H_j = 1$ is a best-response for each jurisdiction. $\square$

2.6 Stock dependent harvest costs

We have shown that, if harvest costs are independent of stock density (Assumption 4), marine reserves can achieve first-best equilibrium profits in a transboundary fishery in which jurisdictions choose harvest rules non-cooperatively (Proposition 2). The assumption of stock independent harvest costs is in many cases unrealistic (Clark 2010). We now relax Assumption 4 and consider the case where the unit costs of harvest are inversely proportional to stock density ($\theta > 0$).

When $\theta = 0$, harvest costs could be ignored, so the system we are analyzing had the helpful property that aggregate profit depends only on the *number* of fish harvested, not on the *location* of that harvest. This property will no longer hold for $\theta > 0$, because the density of fish in a given jurisdiction affects harvest costs. Thus, the problem becomes more challenging, and relies on carefully accounting for the spatial dynamics of stock density. We begin with a result linking stock density and reserve size.

Lemma 3. *Holding yield constant, density is a decreasing function of reserve size.*

Proof. For jurisdiction A who faces reserve size r, density is given by: $n_A = f(mn) + sn_A - \frac{Y}{1-r}$. Since constant yield implies constant n (see Equation 2.17 in the proof to Lemma 1), under the assumption of constant yield, we needn't consider changes in n. Rearranging gives $n_A = \frac{f(mn) - \frac{Y}{1-r}}{1-s}$. Taking the derivative gives: $\frac{\partial n_A}{\partial r} = -\frac{Y}{(1-s)(1-r)^2} < 0$. ☐

This result will prove useful because it establishes that, if you hold yield in the entire fishery constant, density will decline as the reserve size is increased. This insight immediately destroys the ability of reserves to reproduce first-best profits in a non-cooperative fishery in which harvest costs are increasing as stock density declines.

Proposition 3. *When $\theta > 0$, it is not possible to produce first-best profits with any reserve ($r > 0$).*

Proof. Consider a single owner who chooses yield Y (or equivalently, harvest rate H) under reserve size r, which produces equilibrium out-of-reserve density n_A. Steady state profit is

$$\pi(Y, r) = pY - \int_{n_A}^{n_A + Y/(1-r)} \frac{\theta}{n} dn. \tag{2.33}$$

Given r, the owner will choose Y to maximize 2.33. Denote the maximized value by $\pi^*(Y(r), r)$, where we explicitly note that optimal harvest will depend on reserve size. We wish to determine how π^* changes with r, as follows

$$\frac{d\pi^*(Y(r), r)}{dr} = \frac{\partial \pi}{\partial r} = \frac{\partial pY}{\partial r} - \frac{\partial \int_{n_A(r)}^{n_A(r)+Y/(1-r)} \frac{\theta}{n} dn}{\partial r}. \tag{2.34}$$

The first equality is due to the envelope theorem, the second equality is simply taking the partial derivative and invoking Equation 2.33. Invoking Leibniz's rule, denoting $n'_A \equiv \frac{\partial n_A(r)}{\partial r}$, this simplifies to

$$\frac{d\pi^*(Y(r),r)}{dr} = \theta \left[n'_A \left(\frac{1}{n_A} - \frac{1}{n_A} + \frac{Y}{1-r} \right) - \frac{\frac{Y}{(1-r)^2}}{n_A + \frac{Y}{1-r}} \right]. \tag{2.35}$$

Invoking Lemma 3, so $n'_A < 0$, this expression is clearly negative. □

Proposition 3 establishes that for any $\theta > 0$, we can no longer rely on optimally-sized marine reserves to deliver first-best profits. In other words, when $\theta = 0$ the negative consequences of non-cooperation could be completely mitigated with a reserve. But when $\theta > 0$ they cannot. Whether a reserve can improve upon the outcome of non-cooperation (when $\theta > 0$) is the subject of the next two results.

Proposition 4. *For sufficiently small* θ*, non-cooperative profit is maximized with a reserve* $r > 0$.

Proof. Consider the optimally-sized reserve (under non-cooperative management), for $\theta = 0$. By Proposition 2 this reserve delivers first-best profit. Now consider a small increase in θ holding r constant. The increase in θ causes, by continuity, a marginal decrease in profit. But this second-order effect cannot fully offset the first order difference identified in Proposition 1. □

Proposition 4 shows that for a sufficiently small stock effect, an optimally sized reserve can always improve upon the non-cooperative solution. This is good news and accords with intuition. The proof essentially relies on the fact that there is a discrete loss from non-cooperative behavior. That loss is entirely offset by optimally-sized marine reserves when $\theta = 0$ and partially offset when θ is small.

Proposition 4 suggests that even with a stock effect, marine reserves can be designed to benefit transboundary fisheries. But the logic underpinning the proof to that result breaks down for large values of θ. To examine what happens under large values of θ, under which the *location* of harvest becomes increasingly important, we define a threshold value of θ. Define by $\hat{\theta}_j$ the size of cost parameter θ such that for $\theta > \hat{\theta}_j$, jurisdiction j finds it optimal to cease harvesting (because harvesting would entail a loss in profit), when $r = 0$. First we will show that the "large" patch can withstand larger values of the cost parameter.

Lemma 4. $\hat{\theta}_A < \hat{\theta}_B \Leftrightarrow \alpha < .5$

Proof. For patch A the marginal cost of the first unit of yield is: $\frac{\theta}{n_A(1-r)\alpha}$ and the marginal benefit of the first unit of yield is p. The value of θ that drives the marginal profit of the first unit of yield to zero is: $\hat{\theta}_A = pn_A(1-r)\alpha$. INCOMPLETE ☐

This gives rise to the following proposition:

Proposition 5. *For sufficiently large θ, adding a reserve always decreases non-cooperative profit.*

Proof. Let $r = 0$ and without loss of generality, let $\alpha < .5$, so $\hat{\theta}_A < \hat{\theta}_B$ (see Lemma 4), and let $\theta = \hat{\theta}_A$. Under that circumstance, $H_A = 0$ and $H_B > 0$ and jurisdiction B acts as a sole owner who can harvest efficiently in the fraction $1 - \alpha$ of the ocean he controls. Consider the effects on profit of adding a reserve of size $r > 0$. When the reserve is imposed, owner B either: (1) remains a sole owner or (2) is joined by owner A and the non-cooperative game commences. In case (1) profits decline by the proof to Proposition 3. Profits in case (2) cannot exceed profits in case (1) because r is fixed and case (1) is managed by a sole owner while case (2) is a non-cooperative game. This argument also applies for any $\theta > \hat{\theta}_A$ (and, by continuity, for some $\theta < \hat{\theta}_A$). Thus, for sufficiently large θ, implementing any reserve $r > 0$ will decrease joint profits. $\square$

Taken together, Propositions 4 and 5 suggest that for sufficiently small values of the stock effect parameter, a reserve can be designed to increase profits in a non-cooperative transboundary fishery[6] but that for large values of the stock effect parameter, the optimal reserve size is zero. In the latter case, adding a reserve would only decrease profit.

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[6] Though never to the first-best level, provided $\theta > 0$ (see Proposition 3).

2.7 Case study: The Chilean loco fishery

In this section, we illustrate our theoretical predictions with a stylized numerical application to the Chilean loco (*Concholepas concholepas*) fishery. In particular, we investigate the extent to which coordination among TURF owners can improve economic outcomes through implementation of either a cooperative harvest agreement or marine reserves. The loco is an abalone-like species of shellfish native to Chile. Loco are sedentary and live close to shore on rocky substrate up to depths of 40 meters, giving Chile exclusive access to the resource (Guisado and Castilla 1983). There is a high-value artisanal fishery for loco in Chile with landings of 2,252 metric tons in 2012 valued at approximately $45 million (IFOP 2012; SUBPESCA 2014).

In 1999, following successive periods of open access, fishery closure, and a failed quota-based management system, the Chilean government implemented a TURFs policy.[7] Under the management system, fishing jurisdictions called *caletas*, coastal areas that serve as operational bases for local artisanal fishing fleets, are granted exclusive spatial harvest rights to adjacent fishing grounds (Orensanz et al. 2005).[8] Although caletas are largely able to effectively manage the resource within their

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[7] In Chile, TURFs are known as "Areas de Manejo y Explotación de Recursos Bentónicos" (AMERBs) or sometimes simply as "management areas" (*áreas de manejo*).

[8] Although fishing for loco is prohibited outside TURFs, there is evidence that illegal fishing in background areas does occur. We do not explicitly account for illegal fishing in this case study. Instead, we assume all fishery area is contained within TURFs.

own TURFs, there is concern that a spatial mismatch between the small scale of TURFs and the medium to large scale of larval dispersal is now resulting in a "tragedy of the larval commons" in which spatial connectivity creates an incentive for caletas to set TURF harvest quotas too high relative to the social optimum (San Martín et al. 2010).

To examine this claim, we present results for a stylized model of the 56 caletas holding loco TURFs in Regions IV-VI, an 800 km stretch of coastline in central Chile (Figure 2.1). We focus on this area because: 1) it represents an economically important portion of the overall fishery, comprising 26% of overall landings between 2000-2010 and 82% of historical yield (González et al. 2006; SUBPESCA 2012); 2) key biological model parameters are known (see Tam et al. 1996); 3) it contains a loco subpopulation biologically disconnected from the rest of the stock and with roughly common larval pool dispersal characteristics (Garavelli et al. 2014).

We use numerical simulation methods to solve for steady state equilibrium outcomes under the three management scenarios. Table 3.5 reports the model parameters. For simplicity, we assume all TURFs are of equal size ($\alpha = 1/J$), although there is some variation in TURF size.[9] We represent the recruitment function $f(\cdot)$ with the Beverton-Holt stock-recruitment model (Beverton and Holt

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[9] Information about TURF areas can be found here: `http://www.sernapesca.cl/index.php?option=com_remository&Itemid=246&func=fileinfo&id=912`.

1957) given by

$$f(n) = \frac{an}{b+n} \tag{2.36}$$

where n is the adult stock density and a and b are parameters.[10]

Table 2.1: Model Parameters

Symbol	Description	Value	Source
J	Number of jurisdictions (caletas)	10, 20, 56	This paper
α	Proportion of fishery area in each jurisdiction	1/J	This paper
k	Age at maturity	4	Zagal et al. 2001
m	Number of settling juveniles	1	This paper
s	Adult survival rate	0.75	SUBPESCA 2008
a	1st Beverton-Holt parameter	5.12x10^7	Tam et al. 1996
b	2nd Beverton-Holt parameter	2.30x10^7	Tam et al. 1996
p	Average price per loco (USD)	1.5	Rosas et al. 2014
θ	Stock effect cost parameter	0 - 5x10^7	This paper

Given the paramterization in Table 3.5, a vector of harvest rates that generates steady state cooperative profits π^* can be found using constrained nonlinear optimization procedures. Recalling the symmetry among caletas, to find the Nash Equilibrium vector of harvest rates that gives steady-state non-cooperative profits π^{NC}, an initial vector of steady-state harvest rates H^{NC} is posited. Taking other caletas' steady-state harvest rates H^{NC}_{-j} as given, caleta j chooses a steady-state harvest rate H^{NC}_j to maximize own steady-state profits. Fixed-point iteration is used on the resulting vector of harvest rates until the harvest rate choice by

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[10] The paramter a is the maximum number of recruits produced, and b is the spawning stock needed to produce recruitment equal to $a/2$. The Beverton-Holt model is a commonly used recruitment function in the marine ecology literature, and it meets the specifications of our theoretical model by setting $m = 1$.

caleta j constitutes a best-response to the harvest-rate choice by all other caletas. The same procedure is used to find equilibrium profits for the marine reserve implementation scenario π^R, with the addition that caleta j also takes the reserve size r as given when determining the best-response harvest rate.

Figure 2.2 reports equilibrium profits, yields, harvest rates, and stock biomass for the stylized model. The cost parameter $\theta = 58\text{x}10^5$ was selected to make non-cooperative yield in the absence of marine reserves approximately equal to a five-year average of landings between 2006-2010 (SUBPESCA 2012). In the absence of marine reserves, annual non-cooperative profits are 25.8% of cooperative profits. If caletas agreed to implement optimally-sized marine reserves equal to 32.5% of fishery area, this would increase fishery profits to 96.3% of the estimated profits under cooperative harvesting behavior. Thus, although the marine reserve policy does not deliver first-best profits, optimally sized reserves can come quite close to the cooperative level. Implementing optimally-sized marine reserves also increases estimated steady state biomass levels by 362%.

Figure 2.3 compares steady state profit outcomes in the three management scenarios under different values of the stock effect cost parameter θ. First, note that as θ increases, the economic loss from non-cooperation decreases ($\pi^* - \pi^{NC} \downarrow$) because the incentive jurisdictions have to increase harvest rates to compete for rents is increasingly offset by the cost of harvest at lower stock densities. Figure

2.3 verifies our main theoretical results about the stock effect. When harvest costs are independent of stock density ($\theta = 0$), steady state cooperative profits π^* equal steady state non-cooperative profits with optimally-sized reserves π^{R^*} (Proposition 2). However, when there is a stock effect ($\theta > 0$), first-best profits cannot be achieved with marine reserves, $\pi^{R^*} < \pi^*$ (Proposition 3). There also exists a range of stock effect parameter values, $0 < \hat{\theta} < 3.2\text{x}10^7$, for which marine reserves can improve on non-cooperative profits without marine reserves, $\pi^{R^*} > \pi^{NC}$ (Proposition 4). Finally, if the stock effect is too large $\hat{\theta} \geq 3.2\text{x}10^7$, adding a marine reserve always decreases non-cooperative profit (Proposition 5). That is, for $\theta \geq \hat{\theta}$, the optimal reserve size under non-cooperation is zero and $\pi^{NC} = \pi^{R^*}$. However, in this range, non-cooperative profits are 90% or more of cooperative profits, so the loss from non-cooperation is relatively small. In summary, we find that there exists a broad range of cost parameters for which a marine reserve policy can improve on non-cooperative economic outcomes, and for stock effect levels at which marine reserves cannot improve upon the non-cooperative outcome, losses from non-cooperation are comparatively small.

Finally, examine the effects of decreasing the number of caletas owning TURFs. Figure 2.4 compares results from allocating the fishery area among 10 or 20 caletas instead of 56. As the number of caletas decreases, the loss in profits from non-cooperative behavior declines. Non-cooperative behavior with 20 and 10 caletas results in 45.6% and 63.6% of cooperative profits respectively. Regardless of the number of caletas, the optimal marine reserve fraction remains the same. Thus, the value of coordination among jurisdictions either through the enactment of a cooperative harvest agreement or marine reserve implementation increases in the number of jurisdictions sharing the resource.

2.8 Discussion

Transboundary fisheries present a significant challenge to the achievement of sustainable and economically efficient marine fisheries. We address the potential for marine reserves to be used as a cooperation mechanism to produce first-best economic outcomes in transboudary fisheries. Our theoretical and numerical results suggest that coordinated implementation of marine reserves can significantly improve equilibrium profits, particularly when the number of jurisdictions sharing the resource is large and the stock effect in harvest costs is low. These results support global empirical evidence provided by McWhinnie (2009) that the condition of transboundary fisheries worsens as the number of countries sharing a fishery increases. Transboundary marine reserves currently exist in regions diverse as Southeast Asia, Central America, and East Africa, but their primary objective is to meet marine conservation objectives (UNEP-WCMC 2008). Our work suggests that marine reserves could also be used to improve economic outcomes in transboundary fisheries.

Several limitations of our modeling approach deserve emphasis. First, our results rely on a set of common, but far from universal, biological assumptions. Further work should be done to assess the robustness of our model under a richer set of population dynamics such as age structure and other types of adult and larval connectivity. Second, future work could extend the economic framework to address a broader range of economic assumptions, such as asymmetric harvest costs among jurisdictions and consider a fully-dynamic framework with non-zero discount rates. Third, we do not explicitly consider the transaction costs of either the cooperative extraction or marine reserve policy approaches. The costs of designing, monitoring, and enforcing both policy approaches will greatly affect the practicality of their implementation in a transboundary fishery. We leave formal consideration of these costs to future work.

As increasing attention is paid to recovering economic losses from mismanaged fisheries (e.g., Beddington et al. 2007; Arnason et al. 2009), effective approaches to reforming fisheries management must be developed an implemented. Our work suggests that marine reserves present a promising management approach to improve outcomes in transboundary fisheries.

Figure 2.1: Regions of Chile

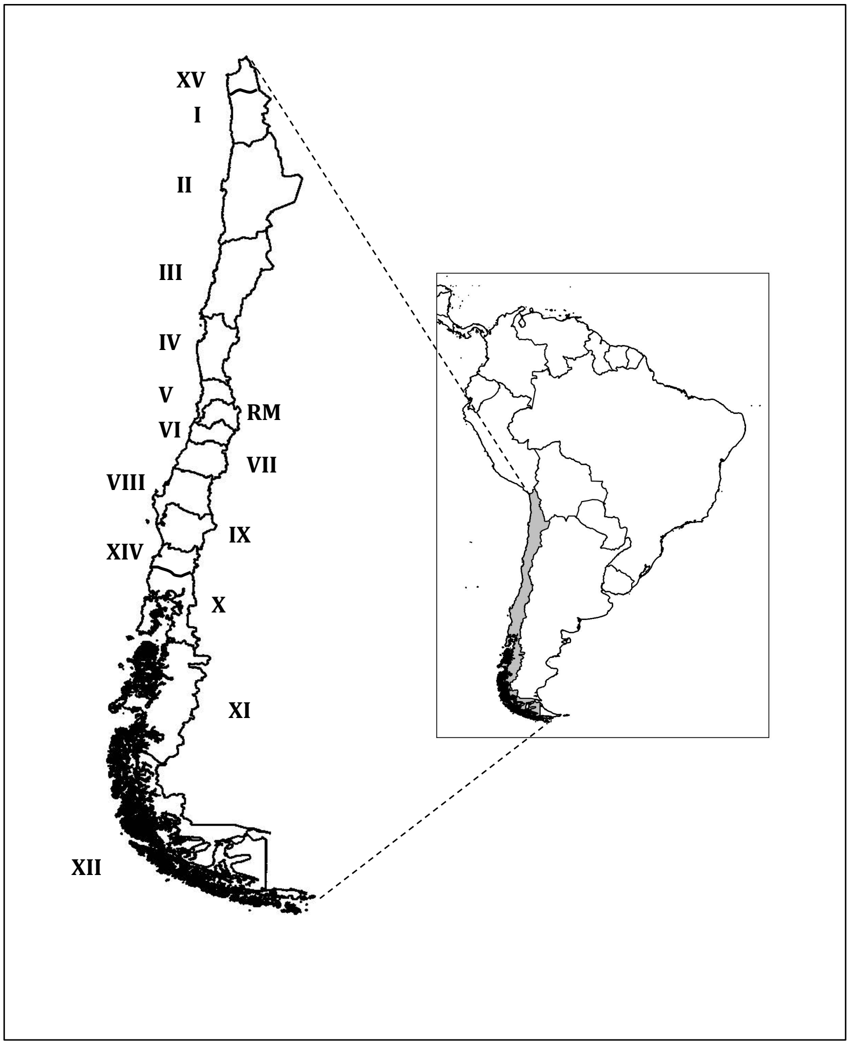
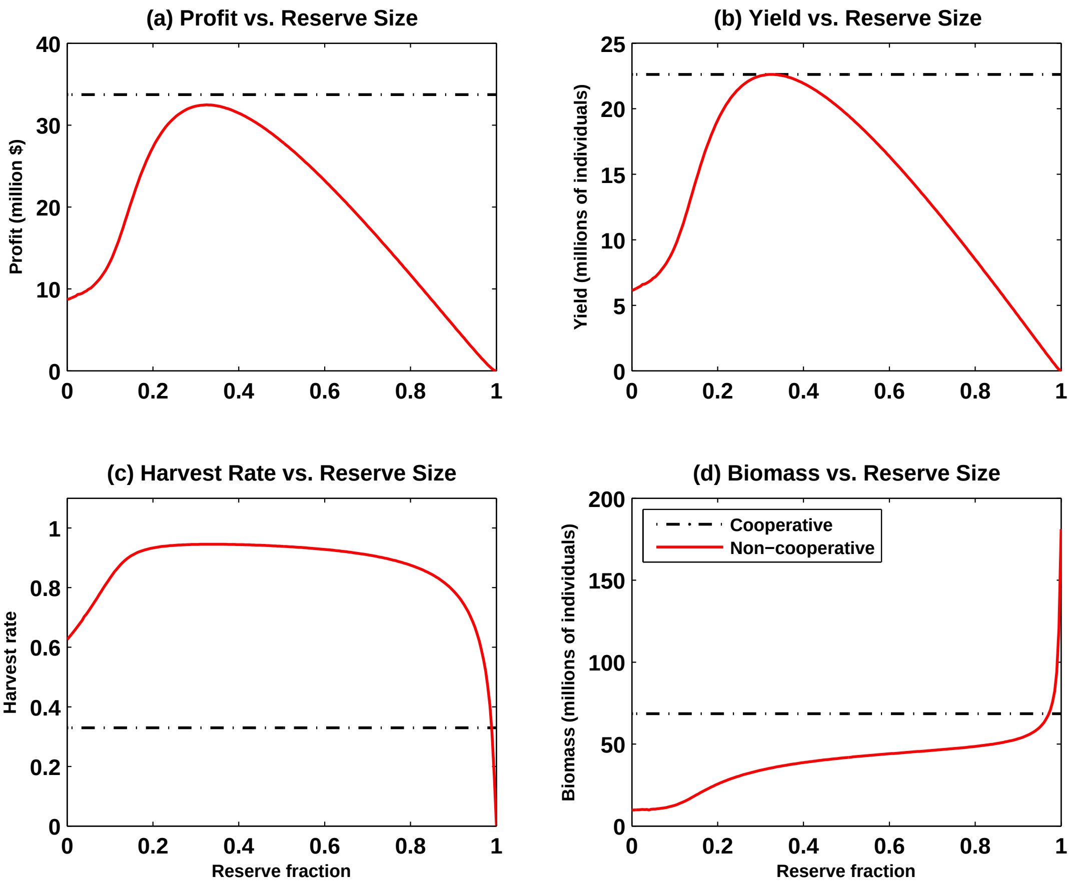


Figure 2.2: Main Results



Notes: Panels (a)-(d) show results for J = 56 and θ = 5x10^5. Cooperative outcomes in the absence of marine reserves are represented by black dashed lines. Non-cooperative outcomes are represented by solid red lines.

Figure 2.3: Varying the Cost Parameter

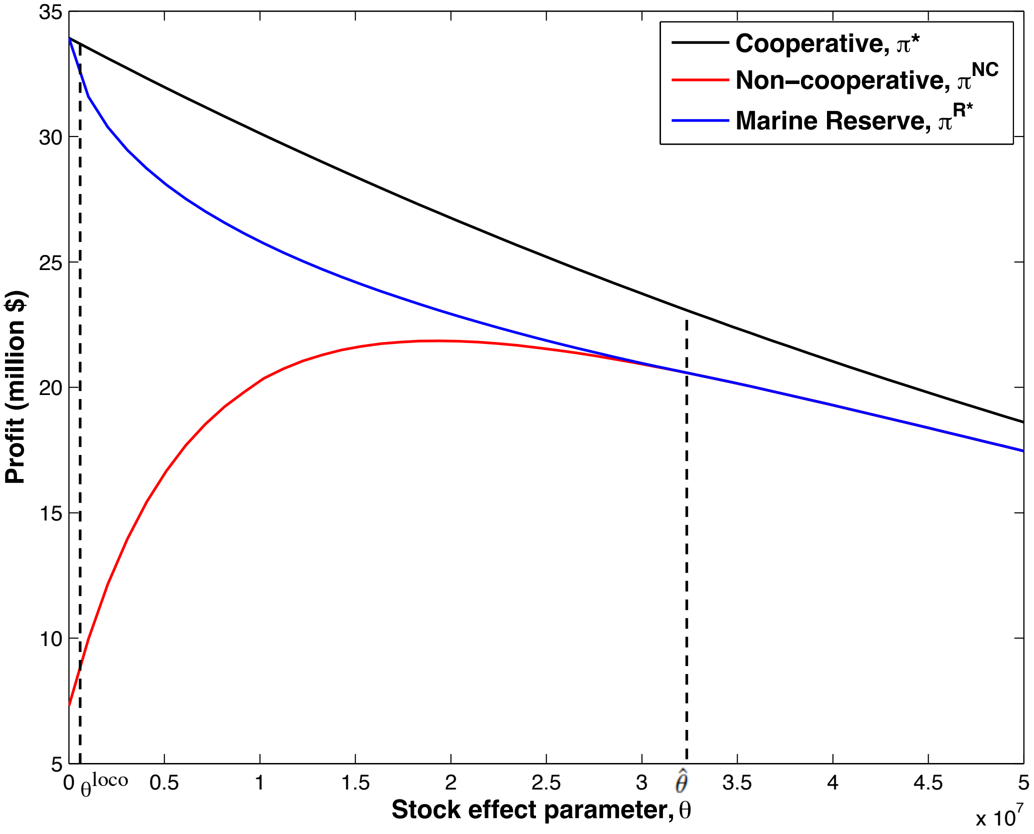
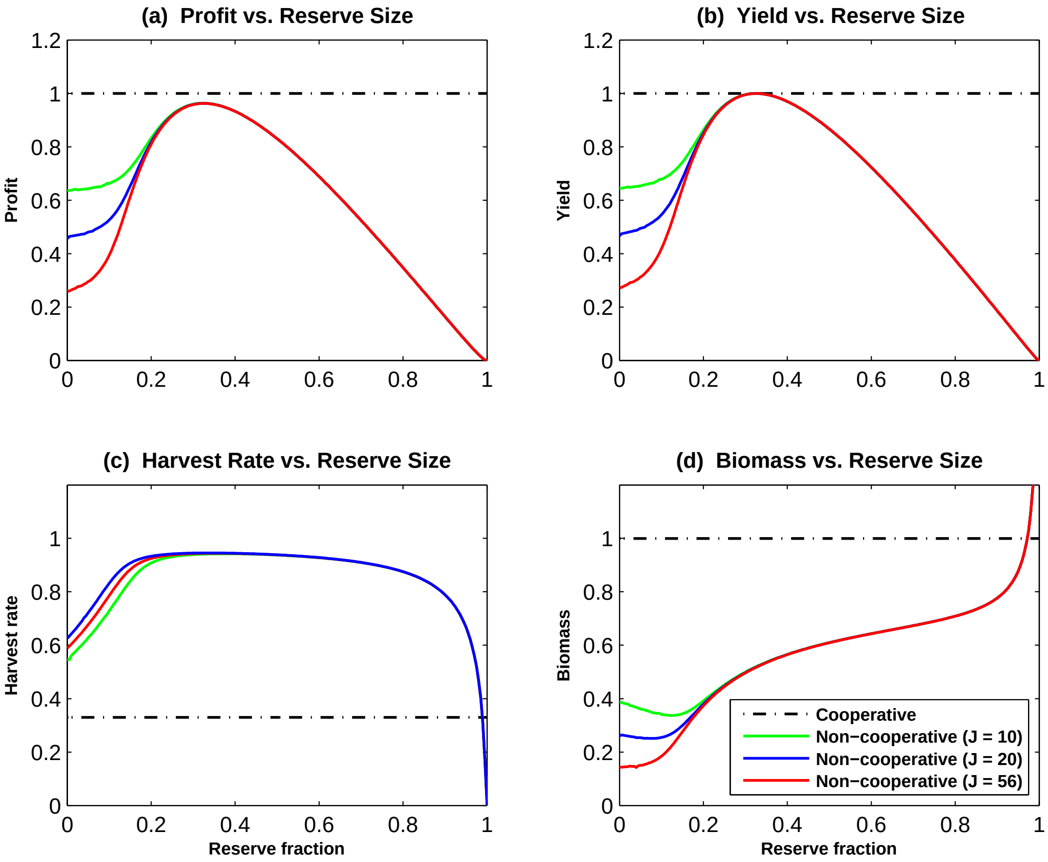


Figure 2.4: Reducing the Number of Jurisdictions



Notes: Panels (a)-(d) show results for J = 10, 20, and 56 for $\theta = 5\text{x}10^5$. Cooperative outcomes in the absence of marine reserves are represented by black dashed lines. Non-cooperative outcomes are represented by solid green (J = 10), blue (J = 20), and red (J = 56) lines. In panels (a), (b), and (d), non-cooperative results are plotted as a fraction of the corresponding cooperative levels.

Chapter 3

Do Marine Reserves Improve Fishery Outcomes? Evidence from the Santa Barbara Area Red Sea Urchin Fishery

3.1 Introduction

Spatial management policies are increasingly used for marine resource management and conservation. No-take marine reserves, areas of the ocean in which all extractive activities including fishing are prohibited, are a fundamental spatial management policy instrument.[1] Marine reserves now figure prominently into many marine resource management plans, and they are increasingly viewed as an important management tool within a suite of policy alternatives to achieve

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[1] Marine reserves are a special case of a broader class of spatial management tools called marine protected areas (MPAs). MPAs are defined as an area of the ocean where human activities are restricted to conserve marine life or habitats, whereas marine reserves strictly prohibit all extractive uses including fishing.

marine conservation and fishery management goals (see e.g., Beddington et al. 2007; Worm et al. 2009).

As the prevalence of marine reserves increases, evaluation of *ex post* effectiveness along biological and economic dimensions becomes increasingly important. A rich literature at the intersection of ecology and economics examines predicted outcomes from marine reserves in theoretical and *ex ante* empirical settings, but few papers establish *ex post* causal effects of marine reserve policies on both ecological and economic outcomes.[2]

This paper contributes to this literature by developing an integrated bioeconomic model to evaluate the biological and economic outcomes of a marine reserve network implemented in 2003 at California's Northern Channel Islands on the Santa Barbara area red sea urchin fishery. We find that eight years after marine reserve implementation, red sea urchin biomass is 53.7% higher, fishery yield is 11.3% lower, and fishery revenue is 10.9% lower compared to simulated outcomes under a counterfactual no-reserve policy.

The concept of marine reserves as a marine policy tool first gained prominence in the ecology literature in the early 1990s as marine ecologists and conservation biologists cited the potential for closed areas to achieve marine conservation goals, including protection of critical habitat and marine populations (Agardy 1994).

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[2] See Conrad and Smith (2012) for a synthesis of existing theoretical and empirical work on spatial fisheries management and marine reserves.

Later, economists such as Hannesson (1998) noted that marine reserves could also offer economic benefits to fisheries if the biomass spillover resulting from a reserve policy more than offset the opportunity cost of the area lost to fishing.

Marine reserves entered international policy discourse on approaches to marine resource conservation and management following the adoption of Agenda 21 at the 1992 UN Conference on Environment and Development, which called upon nations to "to pursue the protection and sustainable development of the marine and coastal environment and its resources" and encouraged the use of protected areas to achieve this objective (UNEP 1992). In 2002, the World Summit on Sustainable Development in Johannesburg further increased the visibility of marine reserves on the global marine policy agenda by adopting a Plan of Implementation that called for "the establishment of marine protected areas consistent with international law and based on scientific information, including representative networks by 2012" (United Nations 2002). In response to these international calls for action, marine reserves have been implemented in over 60 countries worldwide and on all seven continents (PISCO 2011).

The idea that marine reserve policies could confer both ecological and economic benefits has resulted in a substantial body of bioeconomic literature that models potential marine reserve policy outcomes along both ecological and economic dimensions. This literature is primarily focused on identifying "win-win" scenarios in which both ecological and economic indicators improve following marine reserve implementation. While the prediction that marine reserves can improve biological outcomes is a near universal finding of this line of research, the effect of marine reserve implementation on economic indicators is mixed. The state of the resource under the status quo management regime, the spatial distribution of fishing costs, and the underlying biological structure of the resource emerge as important factors in determining whether economic conditions improve after reserve implementation (Sanchirico et al. 2006).

Theoretical contributions to the literature include Sanchirico and Wilen (2001a) who model marine reserve implementation under open access conditions and find that it is possible to increase both aggregate biomass and fishery harvests under certain ecological configurations, namely when the resource exhibits source-sink dispersal dynamics and the source is designated as a marine reserve. Sanchirico and Wilen (2002a), model marine reserve implementation in a limited-entry setting and find that win-win outcomes can be produced in cases where the fishery is operating near open-access effort levels prior to reserve creation, sufficient level of spatial dispersal exist in the underlying resource population, and the area of the fishery designated as a reserve has high costs of fishing. Costello and Polasky (2008) model an optimal management setting and show cases in which implementing marine reserves can result in first-best economic outcomes under a number of bioeconomic conditions.

In addition to the theoretical literature on marine reserves, several papers implement *ex ante* empirical analyses of marine reserve policies by calibrating bioeconomic models to specific fisheries to predict the effects of marine reserve creation. Smith and Wilen (2003) develop an integrated bioeconomic model parameterized to the Northern California red sea urchin fishery and find that while marine reserve creation unequivocally improves biological outcomes regardless of reserve location, equilibrium yield fails to improve under any simulated reserve policy that realistically models fisherman behavior. Using a similar modeling approach, Kahui and Alexander (2008) find comparable results for a New Zealand abalone fishery. In contrast, Valderrama and Anderson (2007) find that temporally rotating marine reserves are part of an economically optimal management strategy for the heavily-exploited U.S. Atlantic sea scallop fishery.

There are comparatively few *ex post* evaluations of marine reserve polices that develop a causal link between marine reserve creation and both biological and economic outcomes. Most *ex post* marine reserve studies have appeared in the ecology literature and focus on conservation benefits inside reserves (e.g., biomass, organism size, and species diversity) and biomass spillovers to adjacent fisheries (Halpern 2003, Lester et al. 2009, Halpern et al. 2010).

To our knowledge, there are only two existing *ex post* empirical studies of marine reserve policies that establish a causal relationship between marine reserve implementation and fishery harvests. Smith et al. (2006) and Smith et al. (2007) examine the introduction of two marine reserves in the Gulf of Mexico multi-species reef fish fishery. Using panel data methods to evaluate the effects of both marine reserves on total harvest, Smith et al. (2006) find that yields after marine reserve implementation are negative and trending downward. Smith et al. (2007) use a structural modeling approach with Bayesian estimation to evaluate the effects of one of the marine reserves on gag grouper and find no statistically significant change in yields after marine reserve implementation.

This paper contributes to this emerging empirical literature on *ex post* marine reserve policy evaluation. We calibrate an integrated bioeconomic model of fishermen behavior and population dynamics in the Santa Barbara area red sea urchin fishery.[3] We use the calibrated model to evaluate biomass, yield, and revenue outcomes resulting from the implementation of a marine reserve network at the Northern Channel Islands in 2003. We accomplish this by comparing observed outcomes under the reserve policy to simulated outcomes from a counterfactual no-reserve scenario in which the entire fishery remains open to fishing.

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[3] We adapt the modeling approach of Smith and Wilen (2003) to our empirical setting.

The urchin fishery provides an ideal setting to conduct an *ex post* empirical analysis of marine reserve implementation. It is relatively well-studied, and ample data describing both the fishery and the surrounding physical environment facilitate detailed modeling of the empirical setting. In addition, the timing of marine reserve policy implementation in 2003 enables us to leverage eight years of post-implementation data in our analysis.

Our work advances the literature on *ex post* marine reserve policy evaluation in the following ways. First, we carry out the first *ex post* reserve analysis that explicitly links an economic model of individual fisherman behavior with a detailed population model to estimate policy outcomes. Second, our microeconometric model allows us to gain insight into the effect of reserves on individual fishermen behavioral responses as well as aggregate changes. Third, our simulation-based approach enables us to compare outcomes in the same location with and without marine reserves.

There are a couple of key limitations to the Smith et al. (2006) and Smith et al. (2007) Gulf of Mexico marine reserve analyses that we are able address in our work. First, the authors are only able to observe outcomes in the first 4.5 years after marine reserve creation. Gag grouper and many of the other species in the reef fish fishery are slow-growing species that are regulated with minimum size limits. If marine reserves ultimately result in biomass spillovers to open areas, these gains may not be fully realized in the time horizon evaluated. Our post-reserve data allow us to observe longer run policy outcomes. Second, the coarse nature of National Marine Fisheries Service (NMFS) data requires the authors to rely on an estimation strategy that treats NMFS statistical areas with reserves as treatment areas and NMFS statistical areas without reserves as control areas. This identification strategy requires the strong assumption that the areas are not biologically connected. Our work is able to explicitly model biological connectivity between discrete patches.

The remainder of the paper proceeds as follows. Section 2 describes the empirical setting including the Santa Barbara area red sea urchin fishery and the marine reserve network implemented at the Northern Channel Islands in 2003. Section 3 describes the empirical approach including the economic model of fishermen behavior, the biological metapopulation model, and the link between the two. Section 4 introduces the data sources and presents the empirical results. Section 5 concludes.

3.2 Empirical setting: The Santa Barbara area red sea urchin fishery and Channel Islands Marine Protected Areas Network

The red sea urchin (*Strongylocentrutus franciscanus*) is a long-lived invertebrate species inhabiting rocky substrate in kelp forests along the Pacific Coast from Baja California to Alaska.[4] Red sea urchins are harvested for their roe (gonads) and marketed as the sushi product uni.

A key feature of urchin fisheries is that the unit price is largely determined by product quality, which is characterized by texture, freshness, color, and taste. The quantity and quality of roe found in urchins has a cyclical component based on the organism's annual reproductive cycle. In the Santa Barbara area, urchin gonadosomatic index (GSI), the ratio of gonad mass to total body mass, generally peaks from September-December, and quality typically peaks in December (Teck et al. 2015). Figure 3.1 shows recent monthly average prices in the fishery with peak prices typically occurring from September-December. In addition to the annual reproductive cycle, food availability is a key determinant of roe quantity and quality. Giant kelp (*Macrocystis pyrifera*) is the primary food source for

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[4] Background information in this section is largely drawn from CDFW (2004).

California urchins, and fishermen use the presence of kelp beds as a visual indicator of productive fishing locations.

In California, the northern fishery (extending from south to north from Half Moon Bay to the Oregon border) and southern fishery (extending from south to north from the Mexican border to Point Conception) are managed separately by the California Fish and Game Commission and the California Department of Fish and Wildlife (CDFW). The southern fishery can be further subdivided into the San Diego, Los Angeles, and Santa Barbara area fisheries, with little overlap in fishing grounds occurring between the Santa Barbara area fishery and fishing activity to the south. The Northern Channel Islands (San Miguel, Santa Rosa, Santa Cruz, and Anacapa), the primary fishing grounds for the Santa Barbara area fishery, are widely regarded as producing the highest quality uni worldwide, and this has led to a consistently high-value fishery.

The Santa Barbara area fishery is based out of three ports: Santa Barbara, Oxnard, and Ventura. Figure 3.2 displays annual landing trends by port. Since the mid-1990s, annual landings in Ventura have been less than 5 percent of total landings. Most fishing activity occurs at the Northern Channel Islands with some additional fishing along the mainland from Point Conception to Malibu and at San Nicholas and Santa Barbara Islands.

The urchin fishery is a dive fishery. Divers harvest urchins from kelp beds at depths of 30 meters or less, using a picking rake to dislodge urchins from hard substrate and place them in mesh bags. They conduct their work using a "hookah rig," a scuba regulator attached to a long hose hooked up to an air compressor on the boat. Diver teams typically consist of 1-3 divers and a tender who monitors the air compressor and air hoses and assists the divers in lifting bags of urchins onto the boat. Boats are small, averaging less than 30 feet, and are designed for speed and the ability to carry large loads of urchins in potentially inclement weather conditions. Fishing trips are typically single-day, although a small percentage of multi-day trips occur.[5]

Weather conditions are a crucial determinant of both diver participation in the fishery and location choice on a given day. Both wind and wave conditions affect whether divers are able to reach fishing sites, particularly those located at the Channel Islands, and affect the fishing process once divers have reached a location. Anecdotally, divers report a lower propensity to go fishing, and a higher propensity to dive in more sheltered areas if they do fish, in the presence of high winds, high waves, and long wave periods (the time elapsed between wave crests). Wave direction is a key determinant of location choice. When waves are arriving from the north or northwest, divers are more likely to fish along coastlines facing

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[5] Multi-day trips are typically no more than two days in duration.

south, and when waves are arriving from the south, they are more likely to fish in locations facing north.

Commercial urchin fishing in the Santa Barbara area began in 1971 as part of a National Marine Fisheries Service program to develop fisheries for underutilized marine species. Ex-vessel prices and the number of divers increased throughout the 1970s and 1980s as demand for uni in the Japanese market increased, and landings grew steadily with the exception of years featuring strong El Niño events.[6] Warmer ocean temperatures during these periods depressed kelp populations, leading to lower product quality and harvests.

In the late 1980s, concern about the sustainability of the fishery led the Fish and Game Commission to establish more formal fishery regulations. These regulations included a moratorium on the issuance of new diver permits in 1987, a formal restricted access program implemented in 1989, and an effort reduction program established in 1990 that currently requires 10 permits to be retired for each new entrant to reach a capacity goal of 300 divers statewide.

In addition to the limited entry program, the Fish and Game Commission also uses a minimum size limit and temporal closures to manage the fishery, but there is no total allowable catch (TAC). The structure of the temporal closures

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[6] Historically, up to 90 percent of processed uni was exported to the famed Tsukiji fish market in Japan. More recently, a rise in domestic demand has resulted in a decrease in export rates to around 20 percent (Teck et al. 2015).

is somewhat unusual. Instead of the common method of opening the fishery on the same date each year and closing the fishery once a TAC has been reached, the fishery has regulated open and closed days that vary by month. Under the current regulation, the fishery is closed on Friday, Saturday, and Sunday from June through October.[7] Table 3.1 provides an overview of the major management actions implemented in the fishery.

On April 9, 2003, the California Fish and Game Commission established a network of Marine Protected Areas (MPAs) in state waters around the Channel Islands.[8] The Channel Islands Marine Protected Area Network closed 13% of Santa Barbara area urchin fishing grounds at the Channel Islands and 10% of fishing grounds area-wide.[9]

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[7] This management approach evolved to satisfy commercial divers in the late 1980s, who believed that consistent delivery of product to the Japanese market was essential to maintain competitiveness with other urchin fisheries.

[8] As established in 2003, the Channel Islands Marine Protected Area Network consisted of ten marine reserves where all commercial and recreational harvest was prohibited and two marine conservation areas that allowed limited take of lobster and pelagic fish. For the purpose of our study, all 12 areas are marine reserves because they prevent all take of urchin. In July 2007, the federal govenment extended some of the MPA boundaries from three nautical miles (the limit of state jurisdiction) to six nautical miles, and established one new MPA, the Footprint Marine Reserve, in federal waters south of Anacapa and Santa Cruz islands. This policy change did not impact the urchin fishery because all urchin fishing occurs in nearshore State waters. Additionally, in a separate initative in 1999, the State government passed the California Marine Life Protection Act (MLPA) requiring the reevaluation of all existing MPAs and the potential design of new MPAs that together function as a statewide network. Design and implementation of a statewide MPA network under the MLPA was carried our at a regional level. The South Coast Region (from Point Conception to the Mexican border) policy went into effect after our analysis period on January 1, 2012. The policy retains the 2003 Channel Islands MPA network evaluated in this paper and adds some additional MPAs at mainland locations. Further information on the MPA regulations resulting from MLPA implementation can be found here: `http://www.dfg.ca.gov/marine/mpa/index.asp`.

[9] We define urchin fishing grounds, or fishable area, as the area of hard-bottom substrate at a depth of <30 meters. Area-wide refers to all fishing locations in Figure 3.5, and Channel Islands

The policy was the result of a four-year process of public meetings and discussion and scientific analysis led by a multi-stakeholder Marine Reserves Working Group (MRWG) consisting of representatives from the federal and state agencies with jurisdiction in the planning region, commercial and recreational fishermen, and representatives from environmental groups (Airamé and Ugoretz 2008). In addition, a 15-member Science Advisory Panel and a five-member socioeconomic panel were formed to assist the MRWG in their analysis of potential MPA network sizes and configurations. The MRWG based its recommendations the ability of the resulting MPA network to fulfill five goals including: (1) the protection of ecosystem biodiversity, (2) the attainment of sustainable fisheries, (3) the achievement of long-term socioeconomic viability, (4) the protection of natural and cultural heritage, (5) provision of educational opportunities to increase awareness and stewardship of marine resources (Airamé et al. 2003). The multi-stakeholder process thus led to the establishment of an MPA network that attempted to maximize several management objectives including, but not exclusively, sustainable fisheries management.[10]

Figure 3.3 presents aggregate trends in the Santa Barbara area urchin fishery before and after marine reserve implementation. During the pre-reserve period,

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refers to Anacapa, Santa Cruz, Santa Rosa, San Miguel, San Nicholas, and Santa Barbara Islands (fishing locations 8-22 in Figure 3.5).

[10] In addition, the red urchin was only one of several fisheries including spiny lobster, abalone, and rockfish considered in the determining the design of the MPA network.

the number of divers, fishing trips, yields, and ex-vessel prices were all on a downward trajectory, which is attributable to a weak Japanese market, competition from other urchin fisheries worldwide, and strong Niño events along the California coastline. After the reserve policy went into effect, prices continued to decline before stabilizing in 2007. The number of divers continued to decline, but at a slower rate. The number of fishing trips and aggregate yields increased from 2002-2004, but are relatively stable in the post-reserve period.

These aggregate trends do not immediately reveal the effect of the marine reserve policy on fishery outcomes. In the analysis that follows, we calibrate an integrated bioeconomic model of urchin diver behavior and urchin population dynamics using an 11 year dataset (January 2000 - June 2011) spanning the pre- and post-reserve period. We use the calibrated model estimate the effect of marine reserve implementation on fishery biomass, yield, and revenue outcomes. The next section presents our modeling approach.

3.3 Bioeconomic model

Our bioeconomic model consists of a discrete choice fishing behavior model and a spatially explicit population model linked via a mapping between fishing trips estimated from the fishing behavior model and the fishing mortality rate applied in the population model.

3.3.1 Fishing behavior model

The economic model captures urchin divers' daily fishing decisions. Based on our knowledge of the fishery, we assume fishing decisions take the nested structure in Figure 3.4. In particular, on each day that the fishery is open (choice occasion), a diver decides *whether* to fish (the participation-level decision), and, conditional on participation, *where* to fish (the location-level decision).

We model an urchin diver's participation and location decision jointly in a random utility framework. Following McFadden (1978), we assume that on each choice occasion a diver will choose the utility maximizing alternative. The utility of diver i choosing alternative j on choice occasion o is given by

$$\begin{aligned} U_{ijo} &= V_{ijo} + \epsilon_{ijo} = f(\boldsymbol{X}_{io}, \boldsymbol{Z}_{ijo}; \boldsymbol{\beta}) + \epsilon_{ijo} \qquad (3.1) \\ & i = 1, ..., I;\ j = 0, ..., J;\ o = 1, ..., O \end{aligned}$$

where $\boldsymbol{X}_{io}$ is a vector of choice-specific variables that may vary over divers and choice occasions but not across alternatives, $\boldsymbol{Z}_{ijo}$ is a vector of alternative-specific variables that may vary over divers, choice occasions, and fishing locations, and

$\boldsymbol{\beta}$ is a vector of unknown parameters. We estimate the model using a Repeated Nested Logit (RNL) model similar to the framework described in Herriges and Phaneuf (2002). We assume that the vectors $\boldsymbol{\epsilon}_{i\cdot o} \equiv (\epsilon_{i0o}, ..., \epsilon_{ijo})$ are independently and identically distributed across individuals and choice occasions and are drawn from a generalized extreme value (GEV) distribution.

The corresponding choice probabilities associated with diver i choosing alternative j on choice occasion o are given by

$$Pr_{ijo} = \begin{cases} 1 - F_{io} & j = 0, \\ \\ F_{io} Pr_{ijo|fish} & j = 1, ..., J \end{cases} \tag{3.2}$$

where $j = 0$ denotes the choice not to fish,

$$F_{io} = \frac{\left[\sum\limits_{k=1}^{J} \exp(V_{iko}/\theta)\right]^{\theta}}{\left[\sum\limits_{k=1}^{J} \exp(V_{iko}/\theta)\right]^{\theta} + \exp(V_{i0o})} \tag{3.3}$$

is the probability that diver i chooses to fish on choice occasion o, and

$$Pr_{ijo|fish} = \frac{\exp\left(V_{ijo}/\theta\right)}{\sum\limits_{k=1}^{J} \exp(V_{iko}/\theta)} \quad j = 1, ..., J \tag{3.4}$$

is the probability that diver i chooses to fish in location j conditional on the decision to fish on choice occasion o. In Equations 3.3 and 3.4, θ is the dissimilarity parameter for the fish nest, which measures the degree of correlation among unobserved portions of utility for fishing location alternatives. This parameter must be between zero and one for the model to be consistent with random utility maximization, and the model reduces to a multinomial logit for the case where $\theta = 1$.

3.3.2 Population model

The population model tracks urchin population dynamics. We develop a metapopulation model consisting of P discrete age-structured subpopulations (patches) linked by a larval dispersal matrix. We assume adults are sedentary and that all dispersal between patches occurs via larval movement. The subpopulation in each patch has a size structure described by a von Bertalanffy growth function such that the length L of an individual of age a is given by

$$L_a = L_\infty(1 - e^{-ka}) \tag{3.5}$$

where a is a monthly time index from 1 to a_{max} and L_∞ and k are parameters representing the terminal size of an individual and organism growth rate respectively.[11] Allometric parameters α and β convert length to weight as follows

$$w_a = \alpha L_a^{\beta} \tag{3.6}$$

Individuals become vulnerable to fishing mortality once they reach the regulated fishery-wide minimum size limit L_{MSL}. Accounting for both natural and fishing mortality, the number of individuals A in patch p that survive the transition from age a to age $a+1$ is given by

$$A_{p,a+1} = \begin{cases} A_{p,a}e^{-m} & \text{if } L_a < L_{MSL} \\ \\ A_{p,a}e^{-(m+f_p)} & \text{if } L_a \geq L_{MSL} \end{cases} \tag{3.7}$$

where m is the instantaneous natural mortality rate and f_p is a patch-specific instantaneous fishing mortality rate. Fishery yield Y for a given month is the summed biomass of harvested individuals from all age classes in all patches and is given by

$$Y = \sum_{p=1}^{P} \left[\frac{f_p}{m+f_p} (1 - e^{-(m+f_p)}) \sum_{a=1}^{a_{max}} w_a A_{p,a} \right] \quad \forall\, L_a \geq L_{MSL} \tag{3.8}$$

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[11] We assume that no individuals live beyond a_{max}. In addition, all biological model parameters could be specified to be patch-specific, but based on available biological information and previous modeling work in this fishery, we specify all parameters to be spatially constant with the exception of the fishing mortality rate and one parameter in the settler-recruitment relationship.

In Equation 3.8, patch-level yield is computed by first calculating patch-level vulnerable biomass (the sum over all age classes of the biomass of individuals satisfying the minimum size limit). Vulnerable biomass is then multiplied by the fraction of individuals subject to either natural mortality or fishing mortality, $1-e^{-(m+f_p)}$, and the proportion of mortality attributed to fishing mortality, $\frac{f_p}{m+f_p}$, to determine patch-level yield. Patch-level yield is then summed over all patches to determine total fishery yield.

The metapopulation model also accounts for reproduction through computation of egg production and larval dispersal, settlement, and survival. Egg production is assumed to occur in January, and it is determined after adult survival has been computed.[12] We assume an exponential relationship between organism length and egg production for mature individuals. Egg production in patch p is given by the sum of egg production from mature individuals in each age class as follows

$$e_p = \sum_{a=1}^{a_{max}} \gamma x^{\delta} A_{p,a} \quad \text{where x} = \begin{cases} L_a & \text{if } L_a \geq L_{mat} \\ 0 & \text{if } L_a < L_{mat} \end{cases} \tag{3.9}$$

where γ and δ are fecundity parameters. After spawning occurs, the resulting larvae enter the water column and are distributed across the system by oceano-

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[12] There is evidence that spawning occurs throughout the winter months (Leet et al. 2001), but for modeling purposes, we assume all egg production occurs in January.

graphic currents before attempting to settle on rocky substrate and recruit to the adult population. The probability of larvae dispersing from one patch to another in the system is given by the P × P larval dispersal matrix **D**, where each element D_{ij} represents the probability of a larvae originating in patch i being transported to patch j. The spatial distribution of larvae attempting to settle after dispersal is given by

$$\mathbf{l} = \mathbf{e}\mathbf{D} \tag{3.10}$$

where **e** is a 1 × P vector of larvae originating from each patch as computed in Equation 3.9, and **l** is a 1 × P vector containing the number of larvae attempting to settle in each patch post-dispersal. The number of larvae that successfully settle and recruit to the adult population in patch p is determined by a Beverton-Holt settler-recruitment function

$$r_p = \frac{\eta l_p}{1 + \nu_p l_p} \tag{3.11}$$

where r_p denotes the number of recruits in patch p, η is a parameter representing the maximum settler survival rate, and ν_p is a patch-specific parameter that sets the maximum recruitment in a patch. Upon successful settlement, recruits in each patch become the first age class.

3.3.3 Linking the fishing behavior and population models

We link the fishing behavior and population models via the following system of equations relating $Trips_{jt}$, the number of predicted fishing trips to location j in month t, to the resulting patch-specific fishing mortality rate f_{pt}

$$Trips_{jt} = \sum_{n=1}^{N} \sum_{o=1}^{O_t} d_{nt} Pr_{njo} \tag{3.12}$$

$$Trips_{pt} = \sum_{j=1}^{J} h_{pjt} Trips_{jt} \tag{3.13}$$

$$f_{pt} = q_t Trips_{pt} \tag{3.14}$$

Equation 3.12 calculates $Trips_{jt}$ by summing over all ports and choice occasions the product of d_{nt}, the number of active divers in port n during month t and P_{njo}, the probability of a diver from port n choosing to fish in location j on choice occasion o, as specified in Equation 3.4. Equation 3.13 maps predicted trips to locations in the fishing behavior model $Trips_{jt}$ to predicted trips to patches in the population model $Trips_{pt}$ by weighting trips to a given fishing location by h_{pjt}, the proportion of habitat area available to fishing in location j in month t that occurs in patch p.[13] Equation 3.14 scales fishing trips to a fishing mortality rate via a catchability coefficient q_t, which is potentially time-variant.

3.4 Data and empirical results

3.4.1 Estimating the fishing behavior model

The choice set

Our discrete choice modeling approach requires us to define the set of fishing location alternatives available to divers. We do this using the CDFW Commercial Landing Receipt Database, which contains individual records of urchin landings commonly referred to as "fish tickets." Fish ticket information includes date and port of landing, diver identification number, vessel identification number, buyer identification number, catch, unit price, and the CDFW statistical reporting block where the fish were caught. Our fishing location choice set consists of 22 alternatives that cover 98% of total fish tickets recorded during our study period. Each alternative corresponds to a either single CDFW reporting block, or, in lightly fished areas, two or three adjacent blocks aggregated to reflect natural

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[13] This mapping allows us to use different geometries in the fishing behavior and population models. In particular, we track subpopulations inside and outside of reserves, which occur at a finer spatial sclae than we observe fishing behavior. Habitat area is computed using the kelp area dataset described in Section 3.4.1.

habitat boundaries. Figure 3.5 presents a map of the study space including the location of fishing ports, marine reserves, and the fishing location choice set.

Data

We construct a dataset of 434,590 diver-choice occasions spanning January 2000 - June 2011 to estimate the fishing behavior model. Choice occasions correspond to the days the fishery is open under CDFW regulations. There are 3,032 open days during our study period. We observe 60,082 individual fishing trips over this period, meaning that, on average, divers fish on 13.8% of choice occasions.

Table 3.2 presents diver trends during our study period. During this time, the number of active divers in the fishery fell from 209 to 108. The average number of patches fished increased moderately from 5.0 in 2000 to 6.2 in 2011. There was a more pronounced increase in average trips taken – 24.6 in 2000 to 44.5 in 2011 – including a jump between 2002-2004, coinciding with marine reserve implementation but more likely attributable to an increase in the number of regulated fishery open days in 2003.

Data for the explanatory variables come from several sources. Choice-specific variables include weather data on wind speed, wave height, and wave period, which come from the National Data Buoy Center.[14] We aggregate the weather

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[14] Information on buoys including maps and downloadable data can be found here: `http://www.ndbc.noaa.gov`.

data to 12-hr averages between midnight and noon on each choice occasion. We define a weekend dummy variable, which takes the value 1 on Friday, Saturday, and Sunday, to capture divers' lower observed propensity to fish on those days (see Table 3.3).

Alternative-specific variables include great circle distance from each port to the centroid of fishable area in each patch. We define fishable area as the hard substrate area within 30m bathymetric contours at the patch level, and inside and outside marine reserves where relevant. We use several shapefiles provided by the CDFW Marine Region GIS Unit including nearshore bathymetric contours and statistical reporting block and marine protected area boundaries to determine fishable area. We construct a measure of expected revenue from fish ticket data. We define expected revenue as the product of a 90-day backward average of patch-level catches and a 30-day backward average of fishery-wide prices. Kelp area data come from the UCSB Landsat Kelp Canopy Dataset, a high spatial resolution time series of giant kelp (*Macrocystis pyrifera*) canopy area.[15] The data were collected by the Landsat 5 TM satellite sensor on a 16-day repeat cycle with a 30m spatial resolution. We aggregate the raw data to a monthly timescale and to the patch level, including accounting for kelp area inside and outside reserves where relevant.

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[15] See `http://www.icess.ucsb.edu/~kyle/Site/Landsat.html` and Cavanaugh et al. (2011) for a detailed description of the Landsat Kelp Canopy Dataset.

Table 3.4. presents descriptive statistics for variables in the discrete choice model. Further information on the fishing behavior model dataset appears in Appendix B.1.1.

Nested logit estimates

Table 3.6 reports the results of our preferred nested logit specification using the full sample of 434,590 diver choice occasions. Variables that are choice-specific include the three weather variables (wind speed, wave height, and wave period) and the weekend dummy variable. These variables, along with the constant term, enter the model at the participation-level. Variables that are alternative-specific include distance to each patch, expected revenue, and kelp area.

All coefficient estimates are statistically significant at the 1% level and are of expected sign. Specifically, we find the utility of fishing is lower in poor weather conditions and on the weekend. We also find that the utility of fishing a patch is decreasing in distance and increasing expected revenue and kelp area. Since the dissimilarity parameter θ is between zero and one (and significantly different from one), we conclude that the model is consistent with random utility maximization and that the nested logit specification is superior to a multinomial logit specification.

Appendix Figure B.1 presents monthly observed trips and predicted trips from the fishing behavior model. Appendix Figure B.2 presents the corresponding residuals. Appendix Figure B.3 presents monthly observed trips and predicted trips for a selection of individual patches. Our trip estimates are noisy. There is also some evidence that we tend to over-predict trips before and under-predict trips after the marine reserve policy.

In future work, we will attempt to improve the fishing behavior model predictions. We may be able to improve the functional form specification by incorporating a wave direction variable, to account for the fact that divers likely choose fishing locations that protect them from waves. We could also use marine fuel prices to create a dynamic measure of travel costs instead of using distance as a static proxy measure. Finally, there is some evidence of measurement error in our ex-vessel price data (see Appendix B.1.1), which we will work to resolve.

3.4.2 Population model

The metapopulation model requires three inputs: 1) a set of patches, 2) a set of parameter values, and 3) an initial population matrix.

Patches

The population model tracks urchin subpopulations in the reserve and non-reserve areas of the location alternatives in the fishing behavior model. Figure 3.6 presents the 32 patches in the population model. Patches 1-22 are composed of the areas in the fishing location alternatives presented in Figure 3.5 that are open to commercial fishing throughout the study (non-reserve patches), and patches 23-32 are composed of areas that become marine reserves in April 2003 (reserve patches).

There is a 1-to-1 mapping between fishing location alternatives that do not contain a marine reserve (i.e., 1-7, 12, 15, 16, 20, and 22) and corresponding patches in the population model. That is, in Equation 3.13, $h_{pjt} = 1$ for $j = p$ and zero otherwise.

There is a 1-to-2 mapping between fishing location alternatives where marine reserves are implemented (i.e., 8-11, 13-14, 17-19, and 21) and corresponding patches in the population model. For example, fishing location alternative 10 is composed of patch 10, a non-reserve patch, and patch 25, a reserve patch. After marine reserve implementation, $h_{pjt} = 0$ for reserve patches and $h_{pjt} = 1$ in non-reserve patches for $j = p$; all fishing takes place in non-reserve patches. Before marine reserve implementation, we split trips to a fishing location alternative in a given month between corresponding reserve and non-reserve patches according to the proportion of the fishing location's habitat (kelp) area lying in each patch. This assumption allows us to estimate fishing mortality rates at the scale of Figure 3.6, although we only observe fishing decisions at the spatial scale of Figure 3.5.

Parameter values

Table 3.5 reports the biological model parameter values. We adopt growth, natural mortality, and allometric length-weight parameters that have previously been used to model the Southern California red urchin population (i.e., CNRA 2009, Rassweiler et al. 2012, Rassweiler et al. 2014) and the fecundity parameters reported in Campbell et al. (1999). We derive the larval dispersal matrix using numerical output from the Regional Ocean Circulation Modeling System (ROMS) and Lagrangian particle-tracking model described in Mitarai et al. (2009).[16] We estimate patch-specific Beverton-Holt settler-recruitment parameters using the methodology of Rassweiler et al. (2012), which allows for heterogeneity in maximum recruitment across patches based on the amount of available urchin habitat.[17]

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[16] ROMS models ocean circulation patterns from 1996-2003, and our larval connectivity matrix represents the average connectivity probabilites for these years. The model was calibrated assuming a December-March spawning season and a seven-week pelagic larval duration consistent with scientific knowledge of red urchin reproduction and early life history characteristics (Leet et al., 2001; Rogers-Bennett et al., 1995).

[17] Here habitat is measured as the area of hard substrate from 0-100m derived from CDFW Marine GIS Unit Predicted Substrate shapefiles, consistent with the urchin habitat and depth range used in the MLPA Analysis for the South Coast Region (CNRA 2009)

Initial population matrix

Simulating urchin population dynamics requires us to have an estimate of age-structured patch-level biomass at the beginning of our study. There is no stock assessment for this fishery, and thus underlying population levels are not known. We therefore construct an estimate of age-structured patch-level biomass in January 2000 using the methods described in Appendix B.1.2.

3.4.3 Calibrating the bioeconomic model

We calibrate the model by estimating q_t using maximum likelihood estimation. In particular, we assume q_t has the following structure

$$q_t = \begin{cases} q_0 & \text{if } t_{post} \leq 0 \\ q_1 + \phi t_{post} & \text{if } t_{post} > 0 \end{cases} \tag{3.15}$$

where q_0 is a pre-reserves catachability coefficient, q_1 is a post-reserves catachability coefficient, ϕ is a post-reserves catchability time trend, and t_{post} indexes the number of months since marine reserves were implemented (i.e., $t_{post} = 1$ in April 2003).

We assume q is constant before reserves are implemented. The rationale for this assumption is that the composition of divers participating in the fishery was relatively stable, and there are no major known changes in regulatory policy, fishing technology, or diver skill during this period. Two major policy changes occurred in Spring 2003, which lead to the assumption of a discrete jump in the value of q accompanied by a time trend. First, reserves were implemented in April 2003, which could lower q in the short term, if divers fished primarily in reserve areas and were forced to search for new productive fishing grounds. Second, the number of days open to urchin fishing under CDFW regulations increased by five weeks beginning in May 2003, which could increase q, if divers were better able to choose to fish during favorable conditions. The time trend ϕ allows for intertemporal fishing behavior adjustment in either scenario.[18]

Figure 3.7 presents simulated monthly yields from the calibrated model alongside observed yields. Figure 3.8 presents the corresponding residuals. The calibration is somewhat noisy, but generally tracks annual cyclical variation in yields. We estimate q_0 = 2.56E-4, q_1 = 3.89E-4, and ϕ= 1.85E-5. Since $q_1 > q_0$, we assume the change in q is driven by the change in number of open fishing days, although the relative effects of each policy change merit further investigation.

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[18] We also considered a quadratic time trend, but model fit was diminished.

3.4.4 Counterfactual simulation results

We evaluate biological and economic outcomes under the marine reserve policy by simulating a status quo counterfactual scenario in which marine reserves are not implemented in April 2003, and all areas remain open to fishing. To simulate the counterfactual scenario, we use observed trips in the pre-reserve period (January 2000 - March 2003) and predicted trips in the post-reserve period (April 2003 - June 2011), estimated using the amount of kelp area available in each fishing location had reserves not been implemented. Since the estimate of q_1 identified in the calibration for post reserve implementation period appears to be driven by the change in the number of open fishing days, not the reserve policy, we use the same q parameter estimates in the counterfactual simulation exercise.

We find that the marine reserve policy decreases fishery yields and revenues relative to the simulated status quo, although these losses are generally modest. Figure 3.9 presents observed annual fishery yields and simulated annual yields in the counterfactual no-reserve scenario beginning in 2003. In 2010, eight years after marine reserve implementation, observed fishery yield is 0.78 million pounds or 11.3% lower than the simulated counterfactual yield. Yields are on average 10.7% lower during the entire 2003-2010 post-reserve period relative to the counterfactual. Figure 3.10 presents observed annual fishery revenues and simulated annual revenues in the counterfactual no-reserve scenario beginning in 2003. We calculate simulated annual revenues by multiplying simulated monthly yields by the average monthly price and aggregating by year. In 2010, observed fishery revenues are \$0.47 million or 10.9% lower than the simulated counterfactual revenues. Revenues are on average 10.7% lower during the entire 2003-2010 post-reserve period relative to the counterfactual.

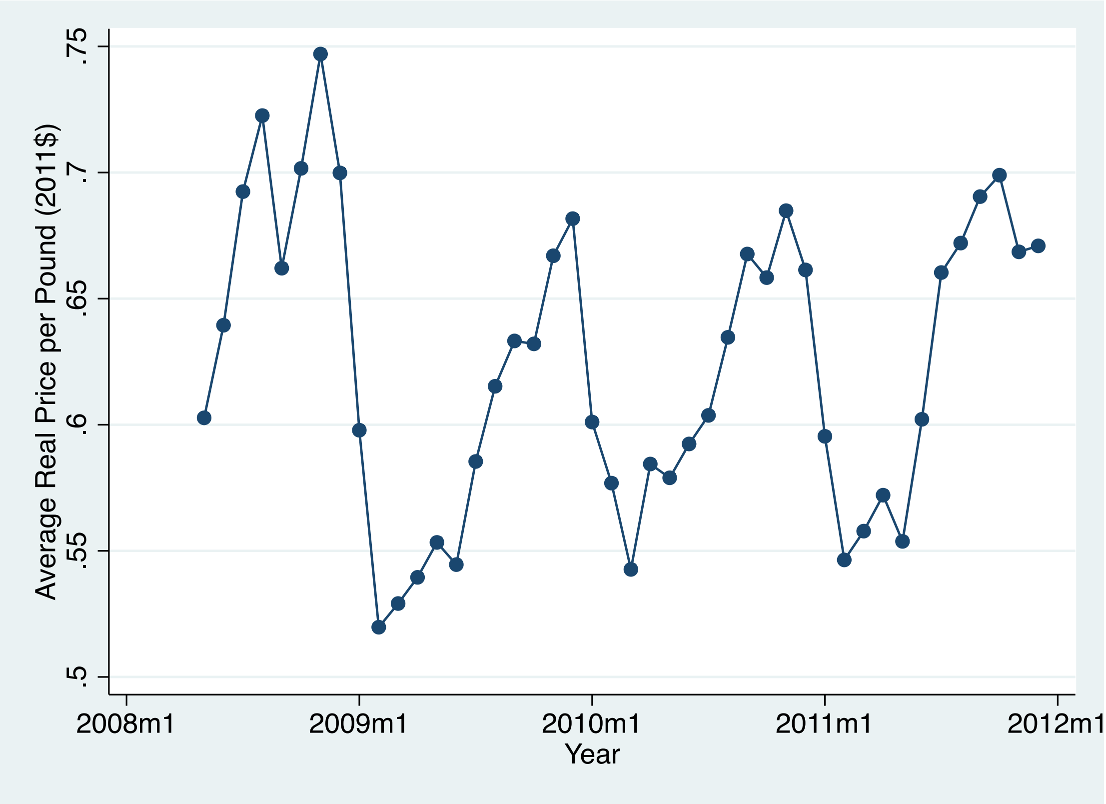
Our yield and revenue results are consistent with Smith and Wilen (2003) who find that hypothetical marine reserves implemented in the northern California urchin fishery never confer fishery benefits, regardless of location. Our findings are also consistent with other recent *ex post* empirical work that finds no evidence of economic benefits of marine reserve policies (e.g., Smith et al. 2006, 2007).

We also estimate biomass outcomes. We never directly observe biomass, so we must simulate it. We simulate biomass resulting from the marine reserve policy using observed trips, and we simulate biomass in the counterfactual scenario using the time series of observed trips during the pre-reserve period and predicted trips in the no-reserve scenario during the post-reserve period. Figure 3.11 reports our biomass results. Simulated biomass under the marine reserve policy is substantially higher than simulated biomass in the counterfactual scenario. In 2010, simulated mean annual biomass is 20.0 million pounds or 53.8% higher than the simulated counterfactual biomass. Biomass is on average 24.3% lower during the entire 2003-2010 post-reserve period relative to the counterfactual. Our biomass findings are consistent with the ecological literature, which generally finds that spatial closures increase population biomass (Halpern 2003, Lester et al. 2009, Halpern et al. 2010).

3.5 Conclusion

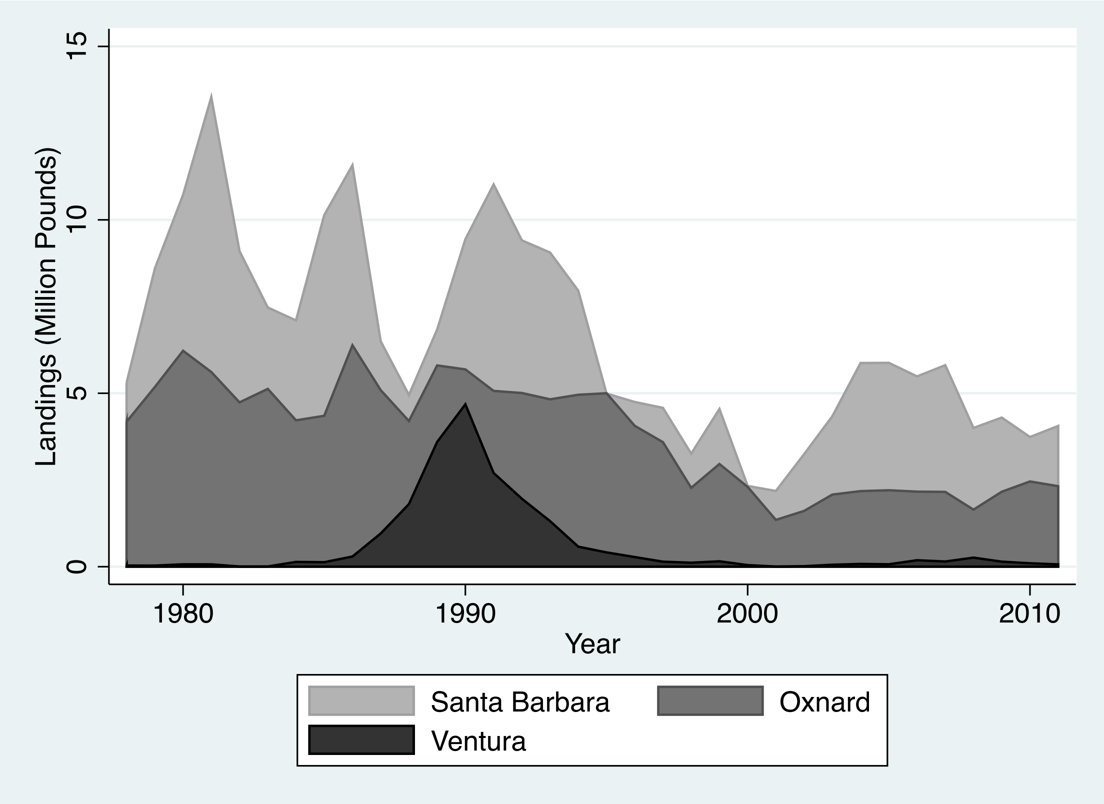
Using a bioeconomic model that maps individual fisherman behavior to aggregate outcomes, we evaluate the implementation of a network of marine reserves in April 2003 on biological and economic outcomes in the Santa Barbara red sea urchin fishery. Preliminary results indicate that eight years after reserve implementation fishery biomass is higher, but yields and revenues are lower, compared to a simulated non-reserve counterfactual. These results support *ex ante* empirical findings from the northern California urchin fishery and *ex post* empirical findings in the Gulf of Mexico. In future work, we intend to refine our economic analysis and calibration methods to improve the precision of our model estimates.

Figure 3.1: Montly Average Price Trends



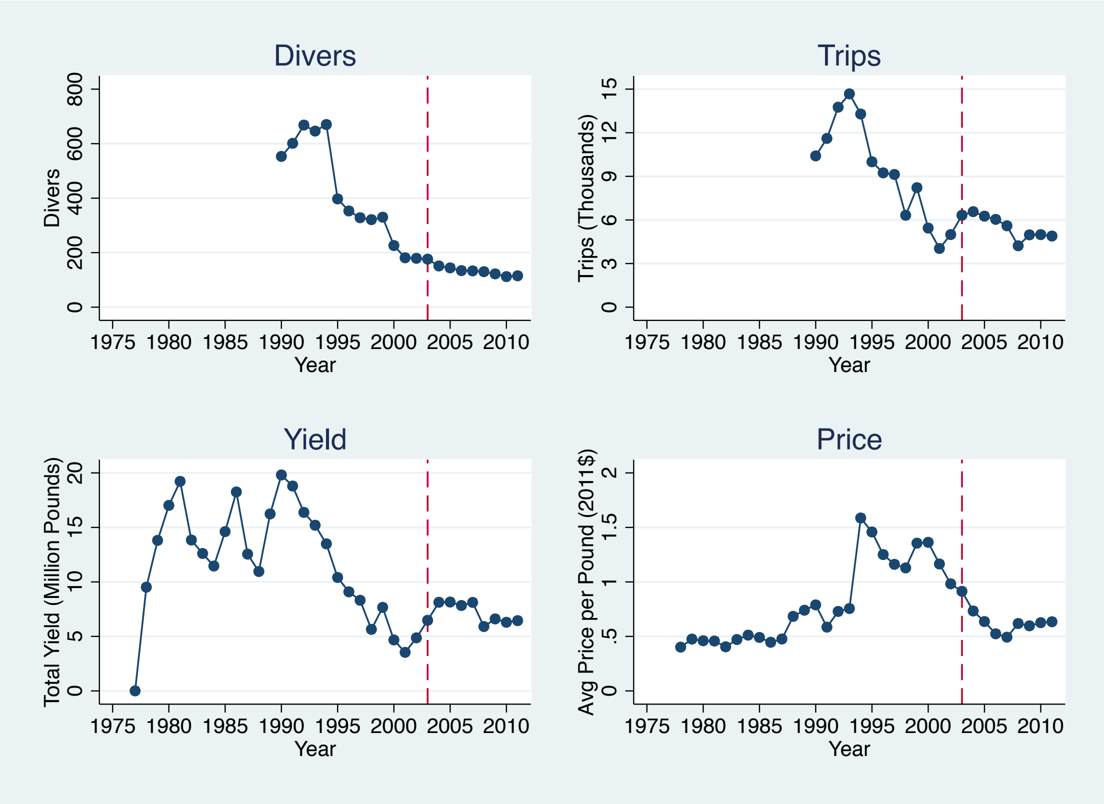
Notes: This figure presents average prices for whole urchin landed in Santa Barbara, Oxnard, and Ventura ports from June 2008 to December 2011. Prices generally peak between September and December, reflecting high gonad mass and quality during these months.

Figure 3.2: Landings by Port



Notes: This figure presents annual yields from 1978-2011 reported in Santa Barbra, Oxnard, and Ventura. The data series are not stacked.

Figure 3.3: Fishery Trends



Notes: This figure presents annual fishery trends including number of divers, number of trips, yields, and average unit price. The red dashed line indicates the year of marine reserve implementation.

Figure 3.4: Nested Fishing Decision

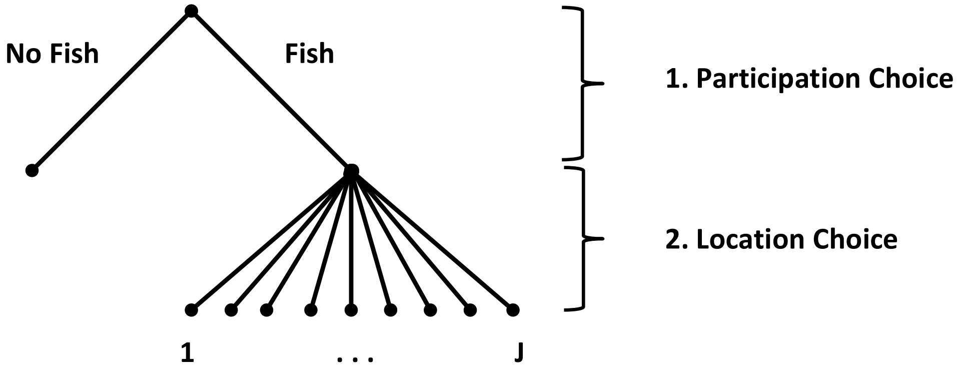


Figure 3.5: Santa Barbara Area Urchin Fishery

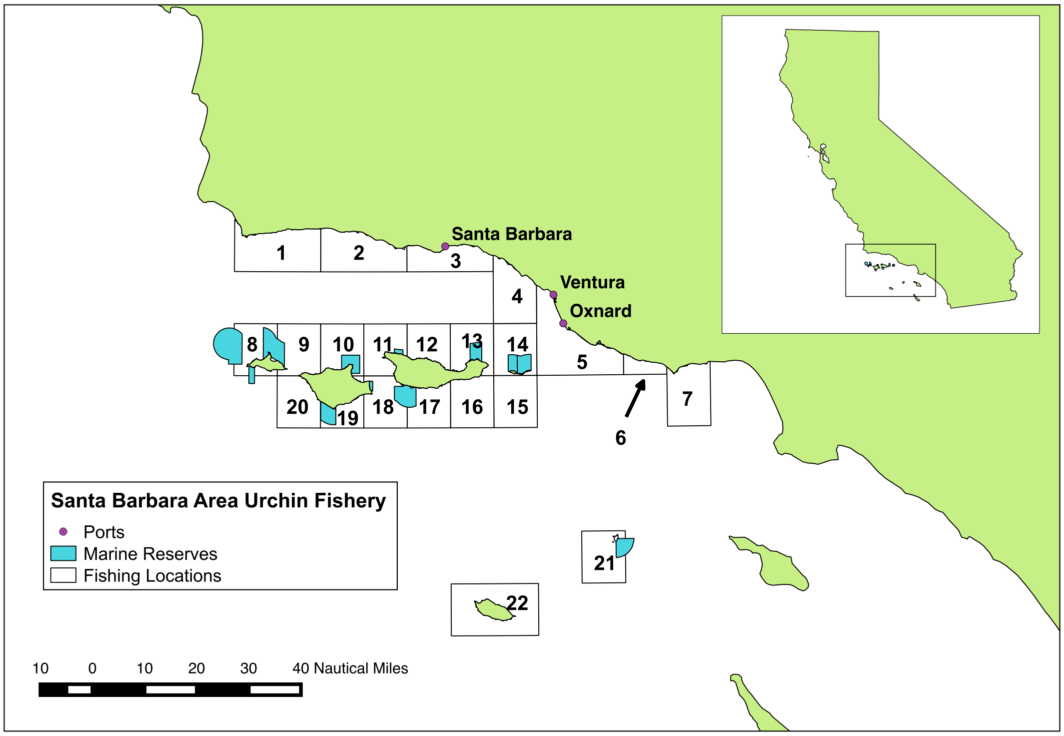


Figure 3.6: Biological Model Patches

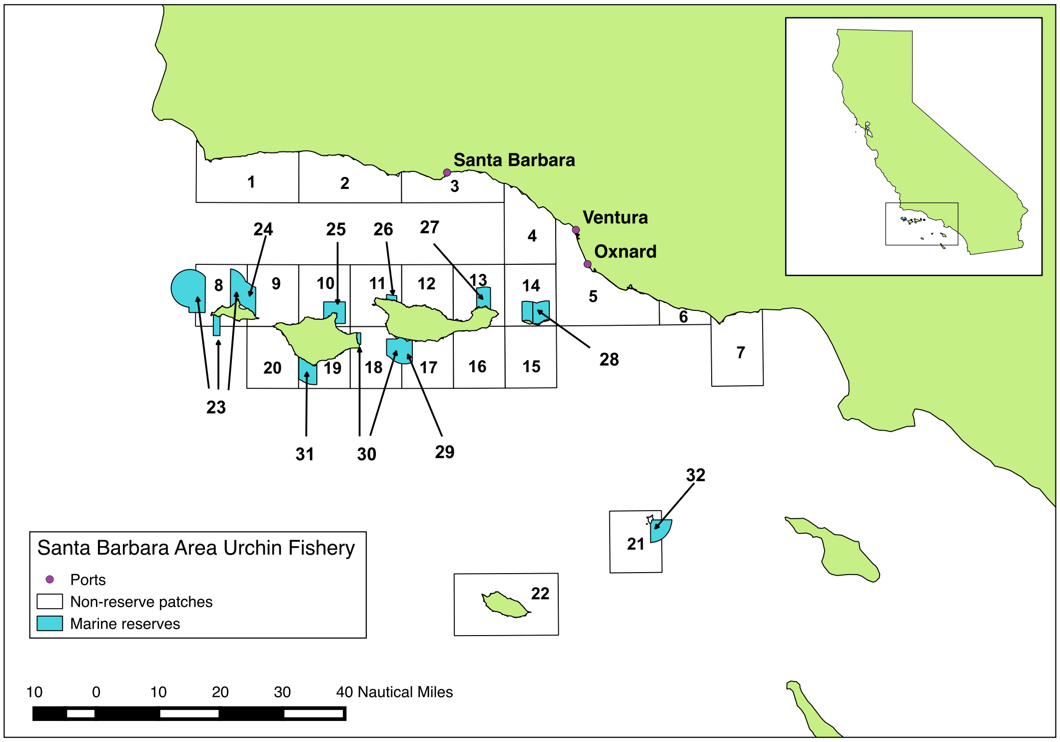
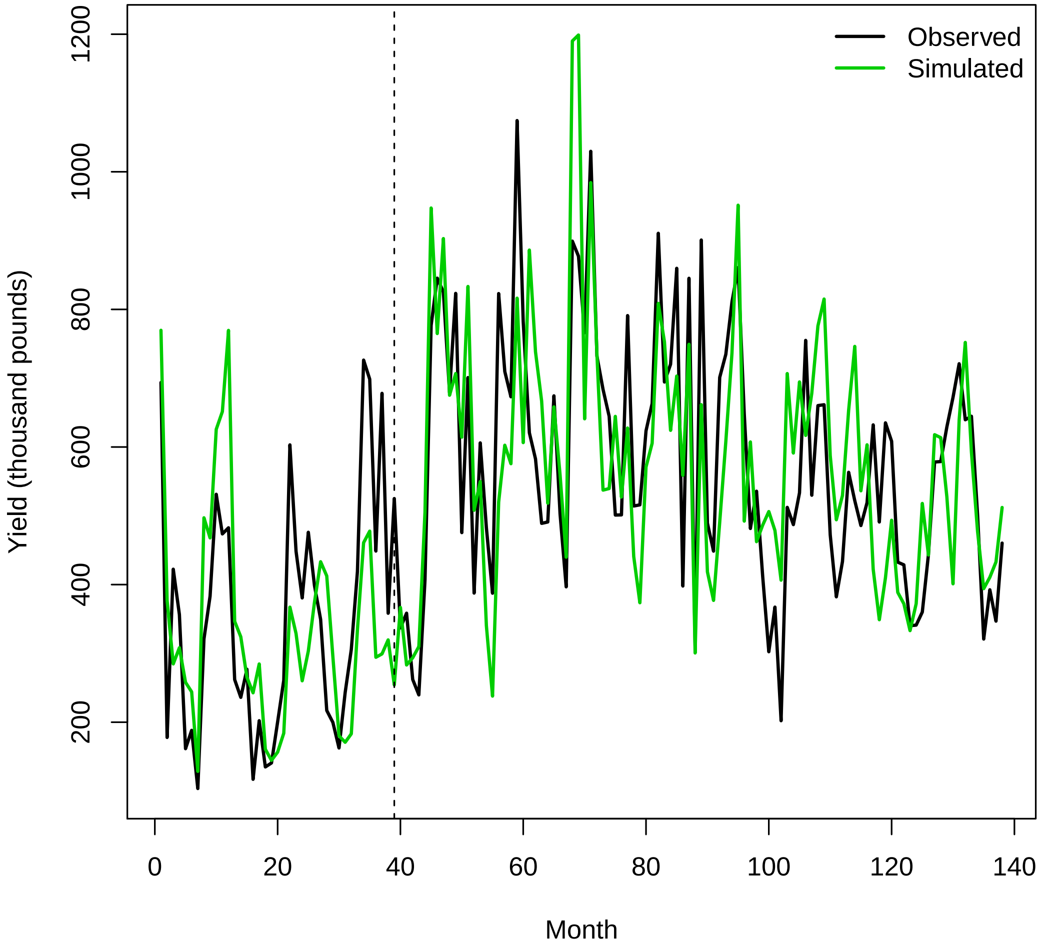
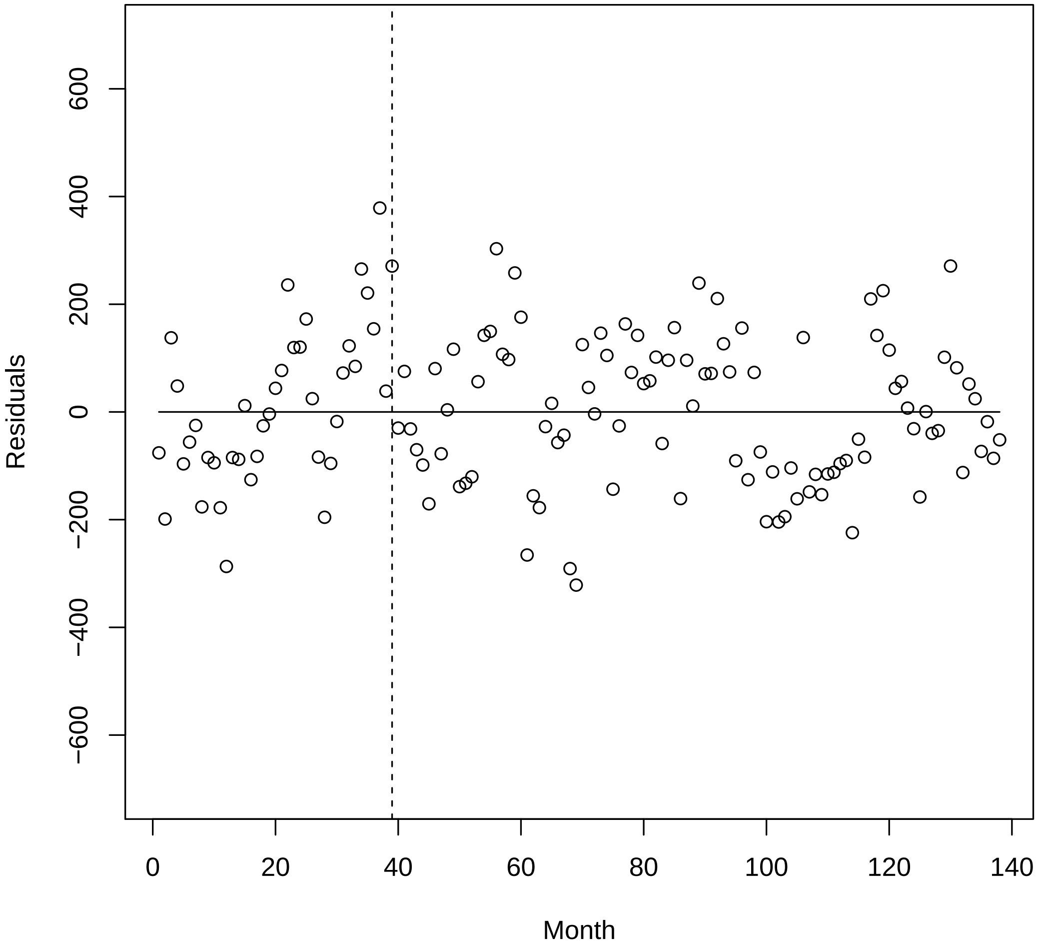


Figure 3.7: Observed vs. Simulated Yield



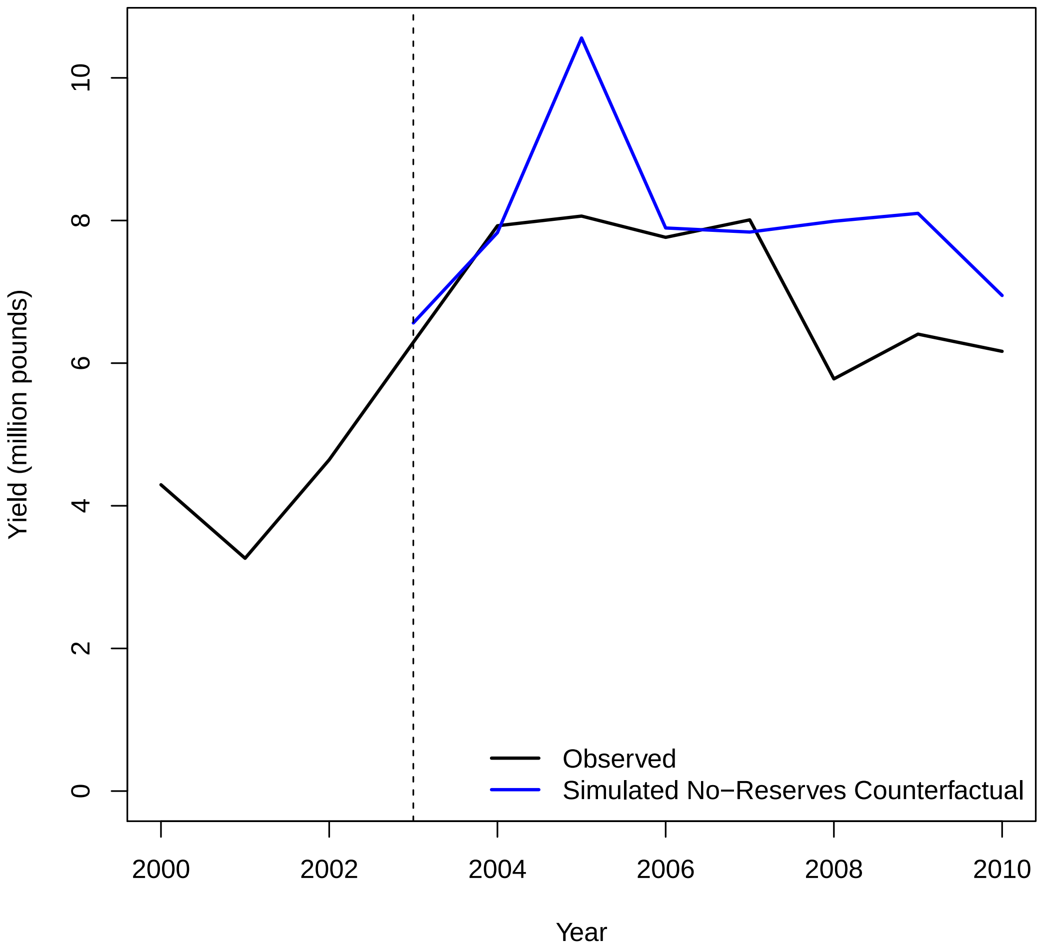
Notes: This figure presents observed monthly yields (black) and simulated monthly yields using predicted trips from the fishing behavior model (green). The data run from January 2000 (month 1) to June 2011 (month 138). Marine reserves are implemented in April 2003 (month 40).

Figure 3.8: Yield Residuals



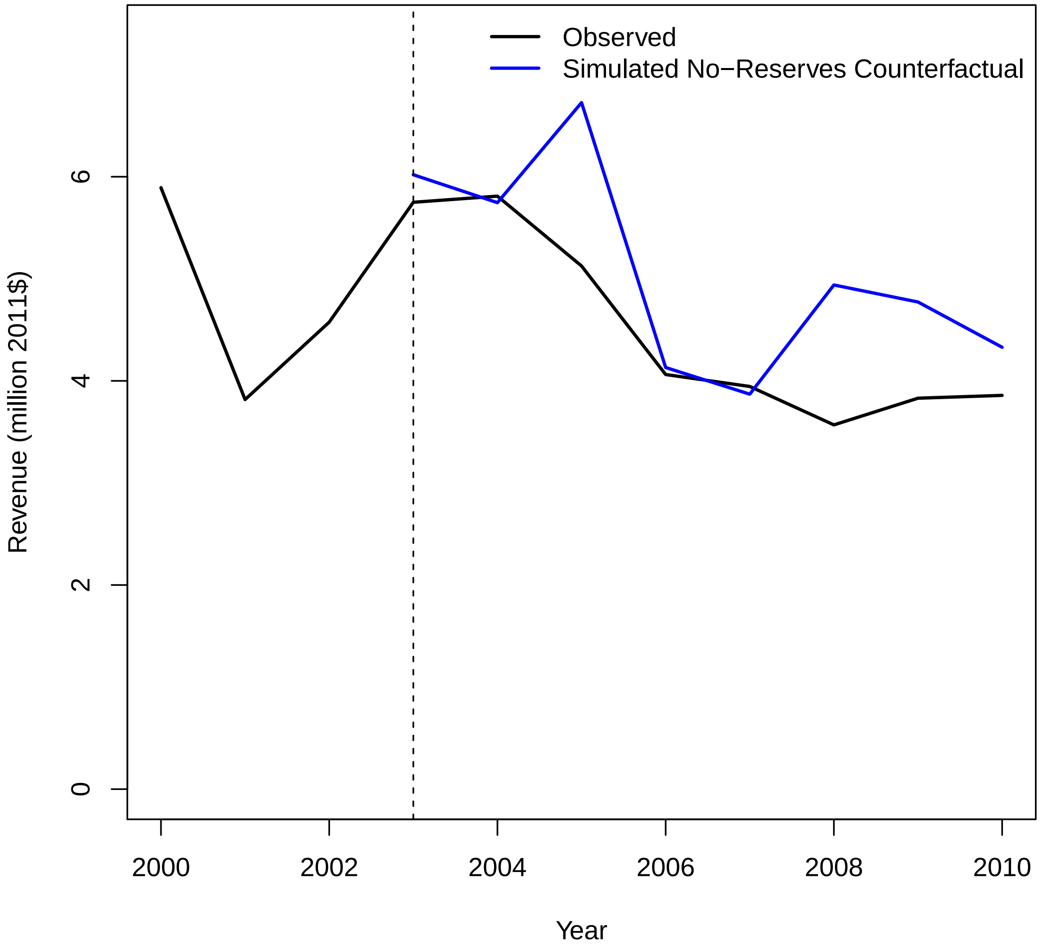
Notes: This figure presents monthly yield residuals from January 2000-June 2011. Marine reserves are implemented in April 2003 (month 40).

Figure 3.9: Yield Results



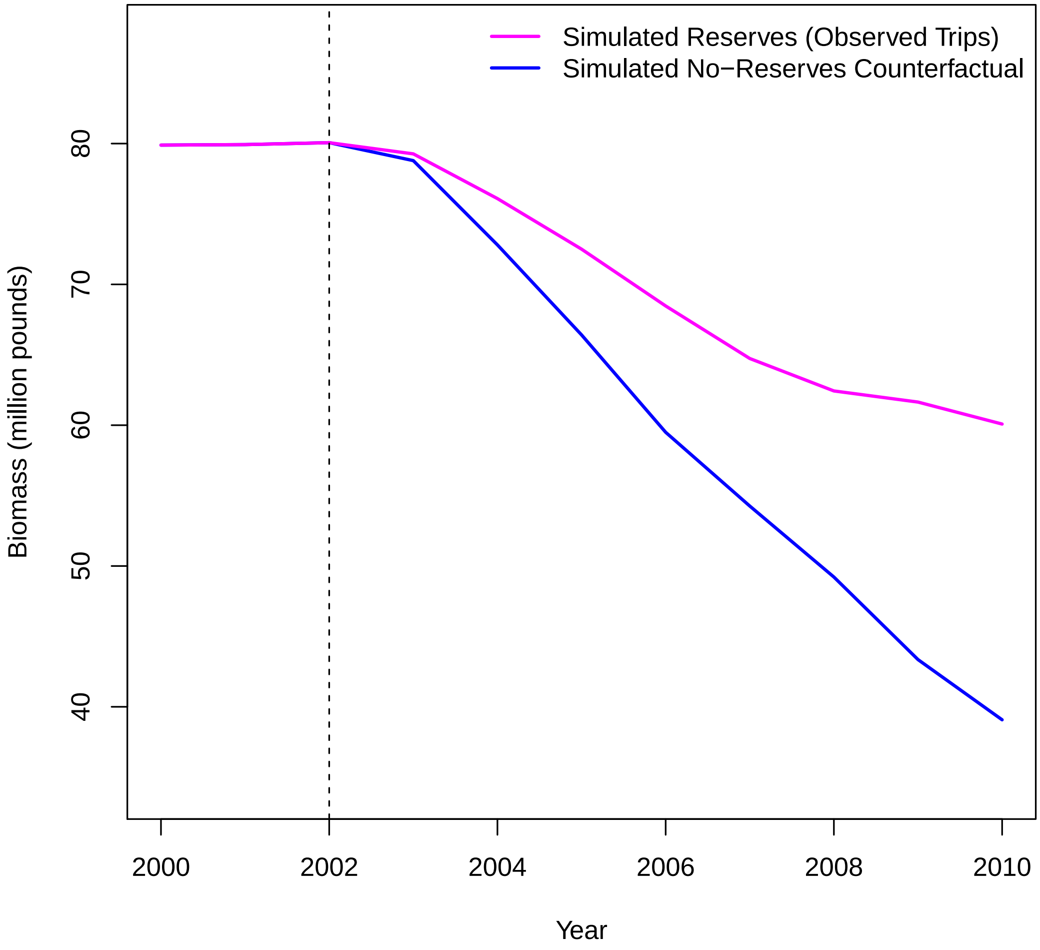
Notes: This figure presents observed annual yields (black) and simulated annual yields in the counterfactual no-reserve scenario (blue).

Figure 3.10: Revenue Results



Notes: This figure presents observed annual revenues (black) and simulated annual revenues in the counterfactual no-reserve scenario (blue).

Figure 3.11: Biomass Results



Notes: This figure presents simulated mean annual biomass with observed trips (magenta) and in the counterfactual no-reserve scenario (blue).

Table 3.1: Santa Barbara Area Red Sea Urchin Fishery Management Timeline

Year	Event
1971	Commercial fishery begins.
1984	State Legislature authorizes a required permit for fishing in the urchin fishery but does not limit entry.
1987	The Fish and Game Commission places a moratorium on the issuance of new permits.
1988	76 mm minimum size limit established. Temporal closures established. Fishery is closed the entire first full week of the month from May to September.
1989	Restricted access program begins. Temporal closures revised so that fishery is closed the entire second full week of the month from May to September.
1990	Effort reduction program introduced that currently requires 10 permits to be retired for each new entrant to reach a goal of 300 divers statewide. Temporal closures revised so that the fishery is closed Friday-Sunday and the entire second full week of the month from May to September.
1992	Temporal closures revised so that the fishery is closed Friday-Sunday in April, May, September, and October; Thursday-Sunday in June and August; Wednesday-Sunday in July; and the entire second week in May-September. Minimum size limit increased to 83mm. Annual landing requirement (20 landings of 300 pounds each in one of the two immediately preceding permit years) for renewal of the permit (repealed in 2004).
2003	Temporal closures revised to eliminate the closure during second full week from May-September. Channel Islands Marine Protected Areas Network established.
2008	Temporal closures revised so that the fishery is closed Friday, Saturday, and Sunday from June through October.

Table 3.2: Diver Trends

		Patches Fished				Trips			
Year	Divers	Mean	SD	Min	Max	Mean	SD	Min	Max
2000	209	5.02	3.06	1	15	24.57	21.93	1	95
2001	167	4.74	2.75	1	12	22.62	20.33	1	84
2002	171	5.04	2.62	1	11	28.20	24.43	1	118
2003	167	5.73	2.97	1	12	36.99	30.71	1	138
2004	146	5.73	2.77	1	14	43.95	31.40	1	135
2005	144	5.46	2.89	1	14	43.03	30.72	1	117
2006	133	5.91	2.87	1	13	45.04	30.08	1	124
2007	132	5.48	2.59	1	13	41.87	30.82	1	121
2008	129	5.22	2.87	1	12	32.26	27.38	1	128
2009	119	5.56	2.75	1	13	40.89	33.27	1	154
2010	110	5.86	3.45	1	16	44.65	34.54	1	142
2011	108	6.16	3.02	1	16	44.48	34.17	1	145

Table 3.3: Trips by Day of the Week

Day	Trips	%
Monday	10,990	18.29
Tuesday	16,258	27.06
Wednesday	12,708	21.15
Thursday	11,588	19.29
Friday	3,615	6.02
Saturday	2,255	3.75
Sunday	2,668	4.44
Total	60,082	

Table 3.4: Descriptive Statistics

Variable	N	Mean	SD	Min	Max
Wind speed (m/sec)	3,032	7.38	3.53	0.74	17.19
Wave height (m)	3,032	2.18	0.83	0.44	7.68
Wave period (sec)	3,032	11.90	3.04	3.69	22.08
Distance (km)	66	61.09	29.24	3.65	127.09
Expected revenue (2011$)	66,704	691.99	390.54	0	3,266.54
Kelp area (ha)	3,036	115.06	207.36	0	2,083.77

Table 3.5: Model Parameters

Symbol	**Description**	**Value**	**Source**
P	Number of patches	32	This study
a_{max}	Maximum age (months)	600	CNRA 2009
L_∞	Terminal size (cm)	11	Rassweiler et al. 2012
k	Growth rate (month^{-1})	0.0183	Rassweiler et al. 2012
α	Length-weight coefficient (cm, lbs)	0.00149	Rassweiler et al. 2012
β	Length-weight exponent	2.68	Rassweiler et al. 2012
m	Natural mortality rate	0.0067	Rassweiler et al. 2012
L_{MSL}	Minimum size limit (cm)	8.3	CDFW regulations
L_{mat}	Length at maturity (cm)	6.0	Rassweiler et al. 2012
γ	Fecundity coefficient	16.82	Campbell et al. 1999
δ	Fecundity exponent	3.27	Campbell et al. 1999
$\mathbf{D}$	Larval dispersal matrix	On request	Based on Mitarai et al. 2009
η	1st Beverton-Holt parameter	3.37E-5	This study
ν_j	2nd Beverton-Holt parameter	1.90E-12 - 9.96E-10	This study
q_0	Pre-reserves catchability coefficient	2.56E-4	This study
q_1	Post-reserves catchability coefficient	3.89E-4	This study
ϕ	Post-reserves catchability time trend	1.85E-5	This study

Notes: Fecundity parameters are based on length in mm, which was accounted for in model calculations.

Table 3.6: Nested Logit Estimates

	(1)
Constant	0.230***
	(0.063)
<u>Choice-specific</u>	
Wind speed	-0.070***
	(0.002)
Wave height	-0.645***
	(0.007)
Wave period	-0.024***
	(0.002)
Weekend	-0.807***
	(0.013)
<u>Alternative-specific</u>	
Distance	-0.0012***
	(0.0004)
Expected revenue ('00s)	0.0094***
	(0.0030)
Kelp ('00s)	0.0073***
	(0.0023)
θ	0.047***
	(0.063)
Observations	9,995,570
Cases	434,590
Log-likelihood	-334,543

Notes: Robust standard errors in parentheses

* $p < 0.10$, ** $p < 0.05$, *** $p < 0.01$

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Appendices

Appendix A

Appendix to Chapter 1

Table A.1: Fisheries in the Analysis

Fishery Management Plan	Species/ Species Complex	Management Area	Catch Shares Adoption Year
BSAI Groundfish	Pollock	BS	1999
BSAI Groundfish	Pollock	AI	
BSAI Groundfish	Pollock	Bogoslof	
BSAI Groundfish	Pacific cod	BSAI	
BSAI Groundfish	Atka mackerel	Eastern AI/BS	2008
BSAI Groundfish	Atka mackerel	Central AI	2008
BSAI Groundfish	Atka mackerel	Western AI	2008
BSAI Groundfish	Yellowfin sole	BSAI	2008
BSAI Groundfish	Rock sole	BSAI	2008
BSAI Groundfish	Greenland turbot	BS	
BSAI Groundfish	Greenland turbot	AI	
BSAI Groundfish	Arrowtooth flounder	BSAI	
BSAI Groundfish	Flathead sole	BSAI	2008
BSAI Groundfish	Other flatfish	BSAI	
BSAI Groundfish	Pacific ocean perch	BS	
BSAI Groundfish	Pacific ocean perch	Eastern AI	2008
BSAI Groundfish	Pacific ocean perch	Central AI	2008
BSAI Groundfish	Pacific ocean perch	Western AI	2008
BSAI Groundfish	Other rockfish	BS	
BSAI Groundfish	Other rockfish	AI	
GOA Groundfish	Pollock	Shumagin	
GOA Groundfish	Pollock	Chirikof	
GOA Groundfish	Pollock	Kodiak	
GOA Groundfish	Pollock	Eastern GOA	
GOA Groundfish	Pacific cod	Western GOA	
GOA Groundfish	Pacific cod	Central GOA	
GOA Groundfish	Pacific cod	Eastern GOA	
GOA Groundfish	Shallow-water flatfish	Western GOA	
GOA Groundfish	Shallow-water flatfish	Central GOA	
GOA Groundfish	Shallow-water flatfish	Eastern GOA	
GOA Groundfish	Deep-water flatfish	Western GOA	
GOA Groundfish	Deep-water flatfish	Central GOA	
GOA Groundfish	Deep-water flatfish	Eastern GOA	
GOA Groundfish	Rex sole	Western GOA	
GOA Groundfish	Rex sole	Central GOA	

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Table A.1 – *Continued from previous page*

Fishery Management Plan	Species/ Species Complex	Management Area	Catch Shares Adoption Year
GOA Groundfish	Rex sole	Eastern GOA	
GOA Groundfish	Arrowtooth flounder	Western GOA	
GOA Groundfish	Arrowtooth flounder	Central GOA	
GOA Groundfish	Arrowtooth flounder	Eastern GOA	
GOA Groundfish	Flathead sole	Western GOA	
GOA Groundfish	Flathead sole	Central GOA	
GOA Groundfish	Flathead sole	Eastern GOA	
GOA Groundfish	Pacific ocean perch	Western GOA	
GOA Groundfish	Pacific ocean perch	Central GOA	2007
GOA Groundfish	Pacific ocean perch	Eastern GOA	
GOA Groundfish	Northern rockfish	Western GOA	
GOA Groundfish	Northern rockfish	Central GOA	2007
GOA Groundfish	Shortraker and rougheye rockfish	Western GOA	
GOA Groundfish	Shortraker and rougheye rockfish	Central GOA	
GOA Groundfish	Shortraker and rougheye rockfish	Eastern GOA	
GOA Groundfish	Pelagic-shelf rockfish	Western GOA	
GOA Groundfish	Pelagic-shelf rockfish	Central GOA	2007
GOA Groundfish	Pelagic-shelf rockfish	Eastern GOA	
GOA Groundfish	Demersal shelf rockfish	Southeast Outside	
GOA Groundfish	Thornyhead rockfish	Western GOA	
GOA Groundfish	Thornyhead rockfish	Central GOA	
GOA Groundfish	Thornyhead rockfish	Eastern GOA	
GOA Groundfish	Other rockfish	Western GOA	
GOA Groundfish	Other rockfish	Central GOA	
GOA Groundfish	Other rockfish	Eastern GOA	
GOA Groundfish	Atka mackerel	GOA	
BSAI Crab	Red king crab	Bristol Bay	2005
BSAI Crab	Snow crab	BS	2005
BSAI Crab	Tanner crab	BS	2005
BSAI Crab	Red and blue king crab	Pribolif Islands	2005
BSAI Crab	Blue king crab	St. Matthew Island	2005
BSAI Crab	Red king crab	Norton Sound	
BSAI Crab	Golden king crab	Pribolif Islands	
Total		**68**	**18**

Notes: In both groundfish FMPs, the sablefish and “other species” fisheries were excluded from the analysis. In the GOA Groundfish FMP, the skate fisheries were excluded from the analysis due to lack of available data. In the BSAI Groundfish FMP, northern, rougheye, and shortraker rockfish fisheries were excluded from the analysis because their management area designations have evolved in a way that makes them impossible to track in a consistent way across time. In the Crab FMP, the Western Aleutian Islands red king crab, Eastern Aleutian Islands golden king crab, and Western Aleutian Islands golden king crab fisheries were excluded from the analysis due to lack of available data. Fisheries are reported using 1996 as a base year. In cases where area designations have become more disaggregated over time, all variables have been aggregated to the baseline area delineations with the following exception. Thornyhead rockfish was managed as a single fishery in the Gulf of Alaska until 1998, and area-specific variables for 1996 and 1997 were approximated using weights established from 1998-2002 data.

Table A.2: Fisheries Group Variable Definition

Group ID	**Fisheries**	**Rationale**
1	Bristol Bay red king crab Bering Sea snow crab Bering Sea Tanner crab Pribilof Islands red and blue king crab St. Matthew Island blue king crab	Rationalized crab fisheries
2	Pribilof Islands golden king crab	Longline pot gear; low utilization
3	Norton Sound red king crab	Small vessel fleet by regulation
4	Bering Sea pollock	Rationalized pollock fishery
5	Aleutian Islands pollock	Catch allocated to Aleut Corporation
6	Bogoslof pollock	Incidental catch only
7	BSAI Pacific cod	Multiple sectors; main target of freezer longline and non-AFA trawl fleets
8	BSAI yellowfin sole BSAI rock sole BSAI flathead sole Eastern AI/Bering Sea Atka mackerel Central AI Atka mackerel Western AI Atka mackerel Eastern AI Pacific ocean perch Central AI Pacific ocean perch Western AI Pacific ocean perch	Amendment 80 fisheries
9	Bering Sea Greenland turbot AI Greenland turbot	Amendment 80 and freezer longline fleet target fisheries
10	Bering Sea Pacific ocean perch BSAI Arrowtooth Flounder BSAI Other flatfish Aleutian Islands Other Rockfish Bering Sea Other Rockfish	Non-rationalized fisheries caught by Amendment 80 fleet
11	Shumagin pollock Western GOA shallow-water flatfish Western GOA deep-water flatfish Western GOA rex sole Western GOA arrowtooth flounder Western GOA flathead sole Western GOA Pacific ocean perch Western GOA northern rockfish Western GOA shortraker/rougheye rockfish Western GOA pelagic-shelf rockfish Western GOA thornyhead rockfish Western GOA other rockfish	Western GOA trawl fleet fisheries
12	Chirikof Pollock Kodiak Pollock Central GOA shallow-water flatfish Central GOA deep-water flatfish Central GOA rex sole Central GOA arrowtooth flounder Central GOA flathead sole Central GOA shortraker/rougheye rockfish Central GOA thornyhead rockfish Central GOA other rockfish	Central GOA trawl fleet non-rationalized fisheries
	Eastern GOA pollock Eastern GOA shallow-water flatfish Eastern GOA deep-water flatfish Eastern GOA rex sole Eastern GOA arrowtooth flounder	

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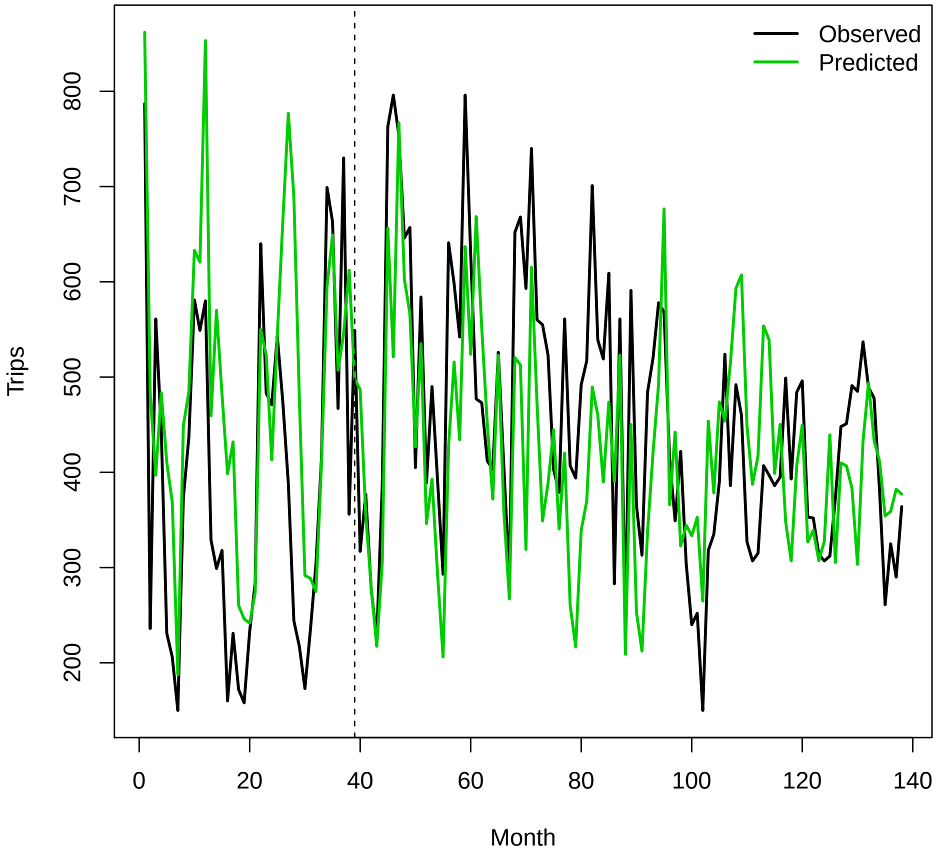
Table A.2 – *Continued from previous page*

Group ID	**Fisheries**	**Rationale**
13	Eastern GOA flathead sole Eastern GOA Pacific ocean perch Eastern GOA shortraker/rougheye rockfish Eastern GOA pelagic-shelf rockfish Eastern GOA thornyhead rockfish Eastern GOA other rockfish	Eastern GOA fisheries
14	Western GOA Pacific cod Central GOA Pacific cod Eastern GOA Pacific cod	Multiple sectors; main target of freezer longline and non-AFA trawl fleets
15	Southeast Outside GOA demersal shelf rockfish	Management delegated to State of Alaska
16	GOA Atka mackerel	Incidental catch only
17	Central GOA pelagic-shelf rockfish Central GOA Pacific ocean perch Central GOA northern rockfish	Central GOA rockfish cooperatives primary species

Appendix B

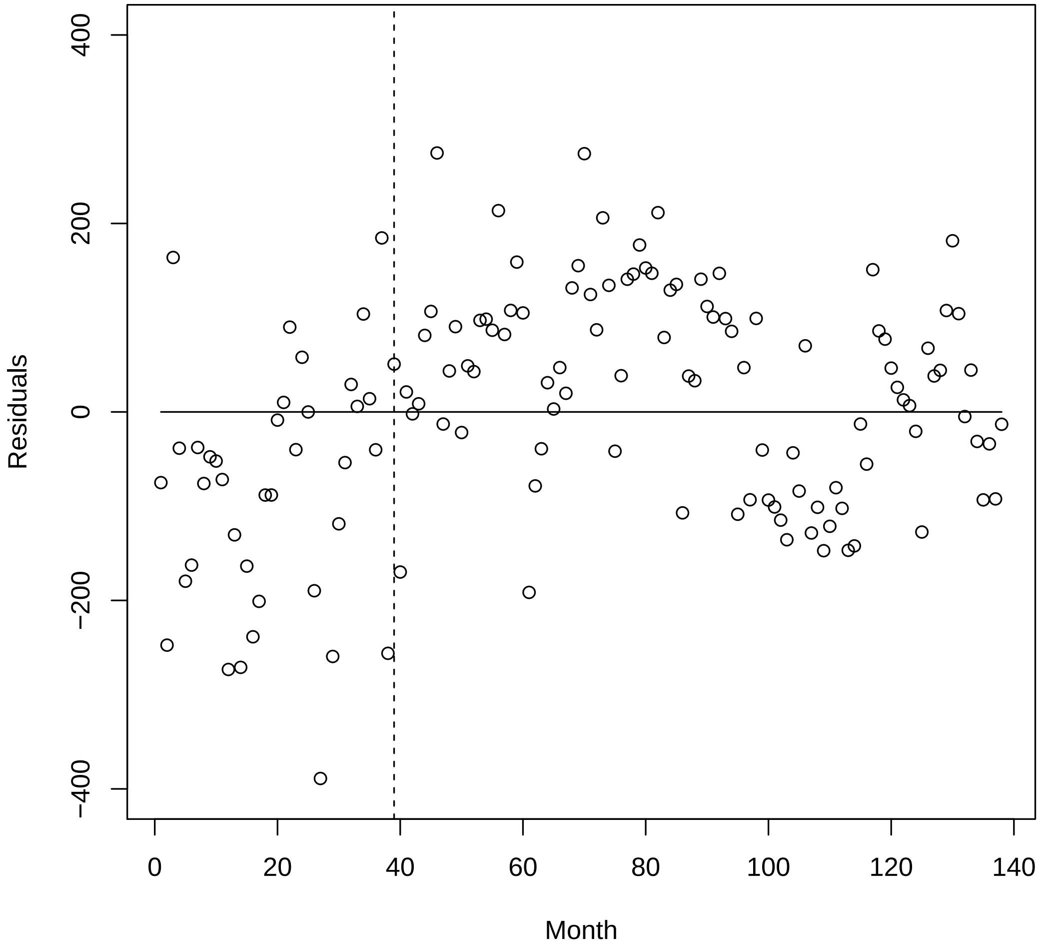
Appendix to Chapter 3

Figure B.1: Observed vs. Predicted Trips



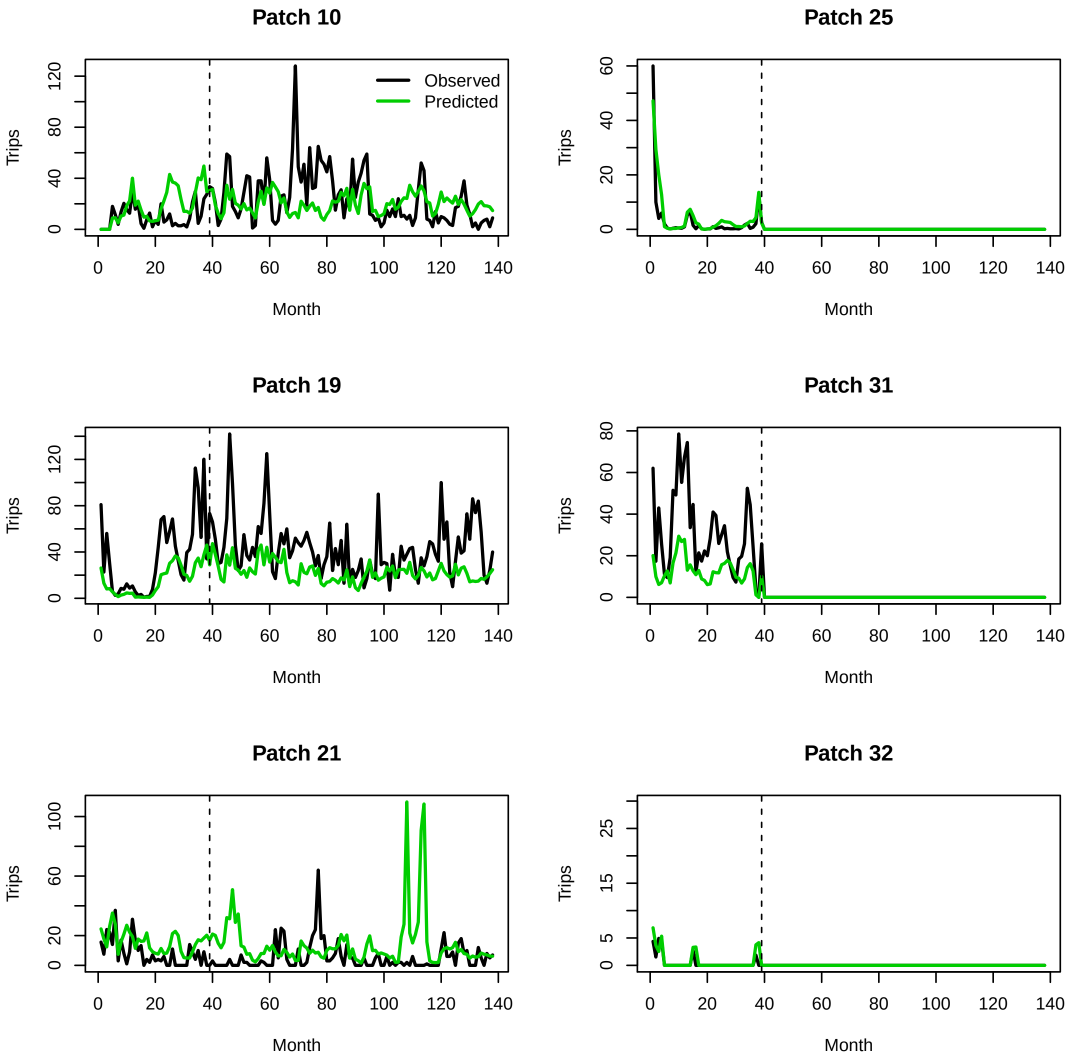
Notes: This figure presents observed trips (black) and predicted monthly trips from the fishing behavior model (green). The data run from January 2000 (month 1) to June 2011 (month 138). Marine reserves are implemented in April 2003 (month 40).

Figure B.2: Trip Residuals



Notes: This figure presents monthly trips residuals from January 2000-June 2011. Marine reserves are implemented in April 2003 (month 40).

Figure B.3: Patch-level Trips



Notes: This figure presents observed patch-level trips (black) and predicted patch-level trips from the fishing behavior model (green). Marine reserves are implemented in April 2003 (month 40). Patches 25, 31, and 32 are reserve patches.

B.1.1 Fishing Behavior Dataset

Fish Ticket Data

Fish ticket data come from the California Department of Fish and Wildlife Commercial Landings Receipt Database, which contains individual records of statewide urchin landings from 1977 to present. Fish ticket information includes date and port of landing, diver identification number, vessel identification number, buyer identification number, catch, unit price, and CDFW statistical reporting block where the fish were caught. To arrive at our 60,082 trip sample, we restrict the data to fish tickets recorded between January 2000 and June 2011 (the last month for which we have kelp data and six months prior to implementation of new mainland marine reserves under the California Marine Life Protection Act) at the ports of Santa Barbara, Oxnard, and Ventura. We drop observations for CDFW statistical reporting blocks outside our choice set (<1% of observations), observations missing vessel ID, diver ID, catch, or price data (<0.1% of total observations), observations recorded on days the fishery was closed to commercial fishing (<0.3% of total observations), and observations for divers that reported fewer than 10 fish tickets during the sample period (<0.5% of total observations).

Finally, we address the issue of multiple fish tickets recorded for the same diver on the same day (approximately 8% of remaining observations). For the 85% of multiple fish ticket cases that result from divers either selling catch to multiple buyers or from fishing in adjacent CDFW blocks aggregated to a single location alternative in our dataset, we aggregate the fish tickets by summing recorded catches and taking the catch-weighted average of prices. For the remaining 15% of multiple diver-day fish tickets that result from divers fishing in multiple fishing location alternatives, we retain the fish ticket with the highest catch. In many cases, this serves to eliminate observations of within-trip search behavior wherein a diver conducts a test dive at a particular location, finds subpar abundance or quality and decides to fish in a different location.

A remaining issue in the fish ticket data is that some price data may be inaccurately reported. Because urchin unit price is determined by the quality of the uni, which is not fully known until urchin have been processed, ex-vessel prices recorded on fish tickets represent an estimate of the actual unit price ultimately awarded to divers. There is some evidence that estimates of unit price are sometimes purposely under-reported. For example, fish tickets for catch sold to three large buyers consistently report a price of \$0.30, between 2004-2007 (compared to an average unit price of \$0.64 for catch sold to other buyers), affecting approximately one-third of fish tickets reported in those years. A limited number of observations (48) report price as zero. Future work will attempt to address and resolve these issues.

Participation and Port Assignment

To estimate the fishing behavior model, we must determine diver choice occasions and port location. For a given diver, we determine fishery participation at an annual level. We deem a diver to be active in the fishery in a given calendar year if he reports at least one fish ticket during the year. Conditional on fishery participation, we also observe some limited port-switching behavior among divers. We assign active divers to ports at the monthly level by assigning each diver to the port at which he recorded the majority of his fish tickets. If the diver does not report a fish ticket during the month, we assign the diver to the port at which he last reported a fish ticket.

Expected Revenue

We use the 60,082 observation fish ticket dataset to construct monthly patch-level expected revenue. We considered three expected revenue calculations. Our first approach is the product of a 30-day backward average of patch-level prices and catches. Our second approach is the product of a 60-day backward average of patch-level prices and catches. Our third approach is the product of a 90-day backward average of patch-level catches and a 30-day backward average of fishery-wide prices. In all three approaches, we faced a missing data problem resulting from lightly fished patches in which 30- to 90-day backward averages could not be constructed because no fish tickets were observed in the relevant window. In these cases, we imputed missing data by filling missing values with the first available previous value. Future work will develop a more robust method for determining expected revenue by either refining our current imputation methods or using a regression approach to determine expected revenue. Our preferred specification (determined from comparison of AIC and BIC values for the competing models) uses the third approach.

Weather Data

We collect hourly weather data on wind speed, significant wave height, dominant wave period, and mean wave direction from the National Data Buoy Center. Wind speed is measured in meters per second averaged over an eight-minute period. Significant wave height, measured in meters, is calculated as the average of the highest one-third of all of the wave heights during the 20-minute sampling period. Dominant wave period, measured in seconds, is the period with the maximum wave energy. Mean wave direction is the direction from which the waves at the dominant period are coming. The units are degrees from true north, increasing clockwise, with north as 0 degrees and East as 90 degrees. Our primary weather dataset comes from the West Santa Barbara buoy (Station 46054) located 38 nautical miles northwest of Santa Barbara (34.265 N 120.477 W). Missing data were imputed using data from the Harvest buoy (Station 46218; 34.458 N 120.782 W), the East Santa Barbara buoy (Station 46053; 34.262 N 119.879 W), and the Point Arguello Station (Station PTGC1; 34.577 N 120.648 W). Future work will investigate incorporating wave direction into the fishing behavior model.

B.1.2 Methods for Estimating Initial Age-Structured Biomass

Production Model

A Stock-Production Model Incorporating Covariates (ASPIC) (ver. 5) software was used (Prager 1992, 1994, 2011) to estimate parameters of a non-equilibrium, generalized production model for the Santa Barbara Red Sea Urchin stock using catch and CPUE data from 1978-2010. The generalized production model used by ASPIC is described as:

$$B_{t+1} = B_t + \frac{r}{p}B_t\left(1 - \frac{B_t^p}{K}\right) - C_t \tag{B.1}$$

where B is biomass at time t, K is the populationâĂŹs carrying capacity, r is the populationâĂŹs intrinsic growth rate, and C is the catch at time t, and p is a shape parameter that describes the relationship of B_{MSY}/K (Prager 1992).

Annual CPUE is assumed to be proportional to biomass:

$$CPUE_t = B_t q \tag{B.2}$$

where q is a catchability coefficient. To account for changes in CPUE that were not related to stock biomass (i.e., changes in minimum size regulations), three catchability coefficients were estimated for the model for the years 1978-1987, 1988-1991, and 1992-2010.

The stock status at the start of the model (1978) was not known, so we allowed the model to estimate this parameter. The best-fit model (based on AIC value) had an estimated carrying capacity $K = 1.30\text{x}10^8$ and an estimated starting stock biomass B_{1978} that was equivalent to 70.1% of carrying capacity. We thought the estimate of B_{1978} seemed reasonable as the fishery began in 1971, and when data collection began in 1978, the fishery was likely lightly exploited. To determine how sensitive our estimate of K was to this assumption we ran the model with a range of initial stock status parameter values and found our starting biomass parameter had no significant effect on our estimate of K.

The model was fit to annual CPUE data from 1978-2010. A sum of squares objective function was used to provide maximum likelihood estimates and a lognormal distribution of residuals was assumed and verified. Bias-corrected 90% confidence intervals were calculated for each estimated parameter from 1,000 bootstrap runs.

Estimate of Age-Structured Biomass

We assume that virgin stock biomass is equal to the stock's carrying capacity ($B_0 = K$). We run the age-structured model described in Section 3.3.2 with a biomass of age 1 individuals that is equal to virgin recruitment (R_0) and a fishing mortality of zero until equilibrium is reached at the stock's estimated K.[1] The age-structured population in February (month of recruitment) at K is assumed to equal the age-structured virgin biomass of the stock prior to the fishery (1970).

We solve for a constant instantaneous fishing mortality rate F that when applied to all vulnerable age classes, brings the stock from K to equilibrium at the starting stock biomass estimated by the ASPIC model ($B_{1978} = 0.71K$). The age-structure of the stock at equilibrium with this estimated constant F is the assumed age-structure of our stock in 1978.

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[1] Virgin recruitment R_0 is equal to the ratio of B_0, which is estimated by the production model, to biomass per recruit BPR, which is calculated using parameter estimates from the metapopulation model.

Monthly spatial catch data is used to estimate the F in every patch for each month from 1978-2010 (Haddon 2001):

$$h_{p,t} = \frac{C_{p,t}}{B_{p,t}} \tag{B.3}$$

Where C is catch in patch p at time t and h is the harvest rate in patch p at time t. Fishing mortality (F) in each patch p, at time t is then calculated as:

$$F_{p,t} = -\ln\text{âĄą}(1 - h_{p,t}) \tag{B.4}$$

We apply F in every patch for each month from 1978-1999 in our age-structured model to arrive at our assumed age-structured starting biomass for 2000.