Property rights, regulatory capture, and exploitation of natural resources*

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Abstract

The literature concerning the impact of property rights on resource exploitation has studied individual harvester's incentives, but in many instances it is a regulator, not the individual firms, who determines the aggregate exploitation rate. We study how the strength of property rights to individual firms affects a regulator's choice over aggregate exploitation rates for a natural resource when the regulator is captured by the industry. The regulator is modeled as an intermediary between current and future resource harvesters, rather than between producers and consumers, as in the traditional regulatory capture paradigm. When incumbent resource users have weak property rights, they have an incentive to pressure the regulator to extract resources at an inefficiently rapid rate. In contrast, when property rights are strong, this incentive is minimized or eliminated. We build a theoretical model in which different property right institutions can be compared for their incentives for exerting influence on the regulator. The main theoretical prediction - that stronger *individual* property rights will lead the *regulator* to mandate more economically efficient extraction paths - is tested empirically and robustly confirmed, with a novel panel data set from global fisheries. These changes in extraction paths are expected to lead to dramatically different environmental outcomes.

1 Introduction

Many natural resources are governed not by the well-studied extremes of open access or sole ownership, but instead by an intermediate case in which a regulator sets extraction caps and a limited number of firms are granted the rights to extract. We focus on two features of these

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systems: First, because the regulator herself sets the extraction rate, she may be susceptible to the influence of the extraction sector, i.e. to regulatory capture. Second, the strength of property rights may affect firms' incentives to exert pressure on the regulator. Thus we study the incentives of firms to influence regulatory decisions over a natural resource. We then derive the equilibrium behavior of the regulator and show how extraction paths will depend on the strength of property rights granted to the harvest sector. We test our theoretical predictions with a novel data set of global fisheries. Consistent with our theory, stronger individual property rights lead the regulator to reduce aggregate rates of extraction. In our empirical study the reduction in extraction rates is expected to lead to dramatically different resource outcomes.

Property rights institutions play a central role in shaping incentives and determining economic outcomes, including investment, resource use and growth (e.g. Acemoglu and Johnson, 2005; Grainger and Costello, 2013; North, Weingast and Wallis, 2009). It is well-established that individual owners' behaviors depend on the security of the underlying property rights. Weak property rights can lead to incentives for individual overexploitation in fisheries (e.g. Costello and Kaffine, 2007), forests (Deacon, 1995, 1999)[1], and land tenure has been shown to affect land use decisions in several settings (e.g. Besley 1995; Alston, Libecap and Mueller, 2000). Though for several reasons it may be impossible or undesirable to grant sole-ownership to a single firm. Examples abound: large aquifers, highly migratory fish stocks, inflow rights to water in rivers or streams, and atmospheric pollution. Thus, these, and countless other resources, are often managed as follows: A regulator sets an extraction cap and a limited set of firms is granted rights over resource extraction.

[1]See Liscow (2012) for a recent counterexample regarding the impact on deforestation.

Despite the ubiquity of this arrangement, the literature on property rights and natural resources has focused on individual incentives and has ignored the critical role of the regulator in setting the overall extraction rules. In fisheries, for example, the aggregate harvest level is commonly determined by a regulator in fisheries worldwide, regardless of whether the fishery is managed by individual property rights.

We address two fundamental questions with respect to property rights institutions in a common pool resource: First, in a theoretical sense, can property right strength to individual firms somehow affect a regulator's choice over aggregate exploitation? And second, what is the empirical evidence of the effects of property right strength on aggregate exploitation? There is some recent theoretical and empirical work on the impacts of rights based management (e.g. Costello, Gaines and Lynham, 2008; Grafton, Squires and Fox, 2000; Grainger & Costello, 2013; Newell, Sanchirico and Kerr 2005), but this literature has been sharply criticized because for most natural resources, it is the regulator, not the individual firms, that determines exploitation rates. This critique calls into question the economist's intuition about the role of property rights on efficient resource use. To our knowledge we are the first to examine the impact of individual property rights on the regulator's choice of resource exploitation rates. We develop a general framework to determine when individual incentives align with the incentives of resource users, and we test the model's predictions using data from a wide range of large, commercial fisheries. This has broader implications for policy design and understanding economic outcomes as shaped by property rights institutions.

Following Karpoff (1984) and others, we take for granted that the industry will attempt to influence the regulatory decisions in their favor. Most of the literature on regulatory capture focuses on natural monopolies or industries in which rights are well-defined and perpetual (e.g. see Stigler 1971, Posner 1974, Peltzman 1976). However, in the natural resources sector rights are often not well-defined, are easily disputed, or are granted only temporarily. These differences in property rights will affect an industry's incentives, and ability, to engage in regulatory capture. Ultimately, then, the property right designation will affect economic and environmental outcomes.

Our results show that when property rights to the resource are strong, the regulator's choice (which is influenced by resource harvesters) coincides with the public interest. However, when property rights to the resource are weak, the regulator's choice leads to overexploitation. This suggests that the resulting extraction level is closer to the socially-optimal extraction level when rights to the resource are strong. Thus private and public interests are more closely aligned when property rights to the resource are strong.

We build an analytical model in which different property right institutions can be compared. The strength of property rights is modeled as the likelihood that the incumbent's rights will be revoked; this allows us to explore a wide spectrum of possible institutions, from open access to perfectly secure rights to a resource. We show how incentives to exert regulatory pressure depend critically on the property right institution. In turn, regulatory outcomes are expected to depend on property right institutions.

We begin with an analytical model that separates the regulator's problem from the extraction path that individuals would choose. We show that when property rights are more secure individuals have an incentive to lobby for a lower exploitation rate. We then provide an empirical test using a novel dataset of historical stock assessments from global fisheries. We estimate policy functions, where the exploitation rate is a function of the property rights institution underlying the resource, the state of the resource (i.e. biomass levels) and fishery-specific and year-specific characteristics. Our estimates are consistent with the analytical model's predictions. We then conclude by discussing the equilibrium biomass implications of our empirical estimates.

2 Resource Dynamics and Regulatory Capture

In this section we develop a simple discrete-time representation of the micro problem facing an incumbent harvesting sector and the "macro" problem facing the regulator. We begin by observing that property rights, and the manner in which they are strengthened or weakened, are strongly context-dependent. A property right has several characteristics, including exclusivity, security, duration, and transferability. Considered in this light, a the strength of a property right may be infringed in any one of several dimensions. For example, the duration or tenure of a property right often differs across institutions (see Costello and Kaffine, 2008, for a theoretical analysis of incentives under limited tenure), as with forestry, wildlife management areas, and some fisheries. But often, the duration is not made explicit; rather, harvesters must cope with uncertainty about the longevity of their rights. An extreme case is open access, in which one's ownership horizon can be regarded as infinitesimally short. Less extreme examples include ownership in many developing countries which lack rule of law so owners face a perpetual likelihood of revocation (by government or indeed by rival owners). On the other hand, a prevalent type of management is limited entry, or "regulated open access" resources (e.g. Homans and Wilen, 1997), which involve rights that may be revoked or limited at any time.[2] Even when rights are explicit, there may be differences in the probability of revocation across jurisdictions.

We focus on the regulator's choice of the harvest rate, though in practice the regulator may influence the management type (i.e. property rights vs. limited entry vs. open access), who may access the resource, and who is granted new access (or how rights are reallocated if expropriated).

The regulator's choice of aggregate harvest rules is the focus of our empirical application. Holding management type constant, the regulator is charged with determining how much of the resource may be harvested during a specified length of time. In a fishery, the regulator typically sets the total allowable catch (the TAC), which places a quota on total harvest.

How the regulator sets the TAC varies across fisheries, but in the United States this involves several steps. Regional management councils, which are typically elected by fishery stakeholders, appoint scientific committees to study the health of the stock through so-called stock assessments. Stock assessments combine sampling and ecosystem models to estimate the biomass as well as a reference point such as the "maximum sustained yield" biomass level (i.e. the biomass level associated with the highest sustainable harvest rate for that fishery's biological characteristics). Given estimates of biomass, the scientific committee recommends an exploitation rate to the management council. Councils then typically have meetings, both public and private, to determine the exploitation rate for the next year (or sometimes several years). During this process, stakeholders provide comments to the regulator, and often the regulator's decision strays from the scientific committee's recommendation.

[2] In addition to revoking an individual's rights, the regulator could increase the set of individuals with access if rights are not well-defined.

We take as given that the industry will exert pressure on the regulator. But because the incumbent industry's rights may be insecure, the regulator also accounts for the effects of her decisions on the future owners' utilities. Thus, the regulator acts as an intermediary between incumbent and future harvesters. Thus, the regulator's problem may not coincide with the social planner's problem, as the regulator can be influenced by different stakeholders in the process. In this sense, the regulator's choice may differ from the socially-optimal harvest rate. That is, capture of the regulator could result in a policy that is optimal for private individuals (e.g. harvesters or processors) but not optimal for the public at large. Our model predicts that this difference will be largest when property rights to the resource are weak.

We abstract away from the precise manner in which property rights are "stronger" or "weaker" by representing the strength of property rights with a single parameter.[3] We assume that incumbent harvesters face the possibility that his resource ownership will be revoked in the future; the higher is this probability, the weaker is the property right. Given this undermining of property right strength, the incumbent will exert pressure on the regulator to achieve her own private objective, which we assume is to maximize expected profit over the duration of his tenure. The regulator, on the other hand, must simultaneously account for the welfare of current *and future* extractors. If any one group is excessively favored, the regulator faces her own retribution.

2.1 Resource user's objective

Let resource stock in period t be given by $b(t)$, and the harvest rate be given by $h(t)$. Current period profit may depend both on resource stock and on the extraction rate: $\pi(t) =$

[3] Another approach from resource economics is in Arnason (2012).

$\pi(b(t), h(t))$, where $\pi_b > 0$, $\pi_h > 0$, $\pi_{bb} < 0$ and $\pi_{hh} < 0$. With complete and secure property rights, the objective function of the resource owner is given by:

$$\max_{h(b)} \sum_{t=0}^{\infty} \pi(b(t), h(t))\phi^t \tag{1}$$

where ϕ is a discount factor. For a renewable resource, this is subject to the familiar equation of motion: $\dot{b} = g(b) - h$,[4] where g has an inverted U shape and $g''(b) < 0$. This is the familiar sole-owner's problem, which has as its solution an optimal feedback control rule, $h^*(b)$; if that rule is followed in perpetuity, the value of the resource will be maximized.

While this will turn out to be a limiting case of the problem we study, it is not the typical case. In our problem, the incumbent resource user faces an uncertain future. In particular, we assume that the incumbent harvester faces a perpetual chance that his rights will be revoked without compensation. Let this probability be given by $1 - \theta$, so θ is a measure of the durability of the property right. We will assume that the parameter θ is fixed and exogenous.[5] Under this setup, the objective function facing the incumbent is:

$$\max_{h(b)} \sum_{t=0}^{\infty} \pi(b, h)(\theta\phi)^t \tag{2}$$

When the incumbent's rights are revoked, they are granted to a subsequent user, who himself faces annual revocation probability $1 - \theta$. Thus, once an owner is granted the harvest

[4]We have removed time subscripts for expositional ease.

[5]It is common, for example, for the general property right structure governing natural resource use to be set at a higher political level than are the decisions over annual extraction rates. In the empirical example that follows (fisheries) this will be the case. In the United States the Department of Commerce sets overall policy on property rights and governance of fisheries, while the regional fishery management councils determine annual quotas, or exploitation rates.

right, he has the objective function given by Equation 2.

2.2 Regulator's objective

In many natural resources, exploitation rates are not directly chosen by the harvesting sector. Instead, a regulator is charged with setting the "cap", or the aggregate extraction rate. In our model the resource owner does not directly set harvest policy. Rather, each owner (current and subsequent) influences the regulator, and the regulator determines the policy. Here we derive the regulator's objective.

What motivates the regulator to choose a harvest policy? On what will that harvest policy depend? We will assume that the regulator is entirely motivated by the influence of his current and future constituents. We will allow for the possibility that *current* resource users might have a stronger influence on regulatory decisions than will *future* resource users. We assume that the regulator wishes to maximize a simple weighted objective function where the weights represent the relative influence of subsequent extractors. Denote by $\gamma \leq 1$ the weight placed by the regulator on the subsequent extractor's welfare. For example, if $\gamma = .5$ then the regulator will consider the welfare of the subsequent owner, but only gives it half the weight of the incumbent's welfare. As γ approaches one, the influence of subsequent owners approaches that of the current owner.

The problem can thus be summarized as follows. An incumbent harvester exerts pressure on the regulator to achieve the objective given by Equation 2. That harvester maintains the same objective, and presumably the same influence over the regulator, until the resource is granted to a new owner. All the while, the regulator must consider the fact that the subsequent resource owner (and indeed those that follow) will inherit a resource stock that is the product of her current regulation. If she allows too much extraction in the current period, thus favoring the incumbent, future resource owners are disadvantaged. Conversely, if she favors future owners too much, e.g. by reducing current harvest to build up resource stocks, then the current owner is disadvantaged. Wanting to balance the welfare of the incumbent with the welfare of the next owner (down-weighted by γ), and indeed the welfare of the owner after that (this time, down-weighted by γ^2), etc., the regulator must solve a complicated dynamic problem. The regulator wishes to solve:

$$\max \mathbb{E}\left[\sum_{t=0}^{\tau_1(\theta)} \pi(b,h)\phi^t + \gamma \sum_{t=\tau_1+1}^{\tau_2(\theta)} \pi(b,h)\phi^t + \gamma^2 \sum_{t=\tau_2+1}^{\tau_3(\theta)} \pi(b,h)\phi^t + ...\right] \tag{3}$$

where the expectation is taken over the uncertain revocation times τ_1 (for owner 1), τ_2 (for owner 2), etc., and will thus involve the annual probability θ.

This seemingly complicated expression turns out to have a simple representation because the problem has a recursive form: once an incumbent is granted harvest rights to the resource, the regulator faces exactly the same problem as she did under the previous owner. Thus the regulator's problem (Equation 3) can be rewritten as follows:

$$\max_{h(b)} \sum_{t=0}^{\infty} \pi(b,h)\Phi^t \tag{4}$$

where the new effective discount factor, Φ, is given by:

$$\Phi \equiv \phi(\theta + \gamma(1-\theta)) \tag{5}$$

which can be thought of as the "capture adjusted discount factor" reflecting: (1) the standard discount factor itself (ϕ, where $\frac{\partial \Phi}{\partial \phi} > 0$), (2) the strength of property right (θ, where $\frac{\partial \Phi}{\partial \theta} > 0$), and (3) the weight given by the regulator to future owners' welfare (γ, where $\frac{\partial \Phi}{\partial \gamma} > 0$). A few special cases are worth noting. First, if the regulator is equally influenced by incumbents and future harvesters (so $\gamma = 1$) the capture adjusted discount factor equals the social planner's discount factor. Second, if property rights are weak (in the limit, $\theta = 0$), the regulator is certain of revocation, so the capture adjusted discount factor equals the social planner's discount factor, down-weighted by γ. Finally, if property rights are strong (so $\theta = 1$), we again recover the social planner's discount factor.

Under this setup, the regulator's problem, which embeds the welfare of the incumbent property right holder and all future property right holders (who exert some influence over the current regulator), has exactly the same functional form as the problem facing the perpetual property right holder (i.e. Equation 4 has the same form as Equation 2). The only difference is the interpretation of the discount factor. For ease of exposition, we henceforth define by δ the regulator's *discount rate* associated with *discount factor* Φ: $\Phi = \exp(-\delta)$.

3 Property Rights and the Consequences of Regulatory Capture

Here we employ the dynamic model above to analyze the consequences of regulatory capture. In particular, we would like to examine the interplay between the strength of property right (represented in our model by θ) and the weight the regulator places on future resource owners' welfare (captured by γ), and how they jointly determine the captured regulator's extraction path.

The regulator's problem (given by Equation 4) has a solution of the form: $h^*(b; \delta)$, which is time-independent. In other words, the regulator will choose a single *policy function* that can be followed every period.[6] That optimized policy function maps the current level of the resource stock (b) into an exploitation rate (h); when that policy function is followed every period, the regulator's objective is optimized. In general, that optimal policy function will depend on the discount rate δ, and by Equation 5, on property right strength θ. The above analysis holds for any sized time step. To facilitate proofs and tractability, we interpret the regulators problem in continuous time. The regulator facing the problem in Equation 4 obtains the following Hamiltonian:

$$H^c(b, h, \mu) = \pi(b, h) + \mu(g(b) - h) \tag{6}$$

for shadow value, μ. Invoking the Maximum Principle yields two useful equations which define the optimal movement of b and h over time:

$$\dot{b} = g(b) - h \tag{7}$$

$$\dot{h} = \frac{\pi_h(\delta - g_b) - \pi_b - \pi_{hb}\dot{b}}{\pi_{hh}} \tag{8}$$

Equation 7 defines movement of the stock. Equation 8 is the *Euler Equation* for this problem, and defines the optimal movement of the harvest rate. In what follows, we assume that the profit expression is separable in h and b, so $\pi_{hb} = 0$, which simplifies the Euler equation to:

$$\dot{h} = \frac{\pi_h(\delta - g_b) - \pi_b}{\pi_{hh}} \tag{9}$$

Equations 7 and 9 can then be used to derive the properties of the steady state of this system, and, more importantly for our analysis, to derive the optimal policy function itself. In particular, we would like to determine what can be said about the optimal policy function's

[6]Formally, a *feedback control rule.*

dependence on the discount rate, δ. The first useful result is that for any δ, the policy function is upward-sloping:

Lemma 1. *The optimal feedback control rule, $h^*(b)$ is upward-sloping:* $\frac{dh^*(b)}{db} > 0$.

Proof. First we show that the $\dot{h} = 0$ isocline, given by the Euler Equation, is upward-sloping. Setting Equation 9 equal to 0, gives $\pi_h(\delta - g_b) - \pi_b = 0$. Totally differentiating with respect to b and h and using the Implicit Function Theorem, gives the derivative of the $\dot{h} = 0$ isocline in the neighborhood of the steady state:

$$\frac{dh}{db} = \frac{\pi_h g_{bb} + \pi_{bb}}{\pi_{hh}(\delta - g_b)} > 0 \tag{10}$$

In the isosector above $\dot{h} = 0$ and below $\dot{b} = 0$, we find that $\dot{b} > 0$ and $\dot{h} > 0$. In the isosector above $\dot{b} = 0$ and below $\dot{h} = 0$, we find that $\dot{b} < 0$ and $\dot{h} < 0$. Since $\frac{dh}{db} = \frac{\dot{h}}{\dot{b}}$, we find that the policy function is upward-sloping. □

Lemma 1 provides an intuitive result: higher levels of resource stock always lead to higher optimal extraction rates. While this result holds for any discount rate, δ, we are primarily interested in how δ influences the policy function itself. An interesting initial question is how the steady state depends on δ, which we summarize as follows:

Lemma 2. *A larger value of δ delivers a lower steady state stock level (b). Whether it delivers a lower, or higher, harvest rate (h) depends on whether the steady state is to the left or right of the maximum value of $g(b)$.*

This result is also intuitive: larger discount rates tend to favor present extraction over future extraction, thus we tend to expect a lower steady state stock. But whether increasing the discount rate increases the harvest rate along the entire optimal policy function is a more subtle question. In the framework presented above, we find an unambiguous answer, summarized as our main theoretical result:

Proposition 1. *A larger value of δ results in a higher optimal extraction rate for any level of stock:* $\frac{\partial h^*(b)}{\partial \delta} > 0$.

Proof. Lemma 2 shows that higher discount rates always move the steady state left on the $h = g(b)$ curve. Lemma 1 shows that the optimal policy function is upward-sloping through that steady state. We proceed by contradiction. Consider two discount rates, $\delta_L < \delta_H$, with corresponding optimal policy functions $h^*(b; \delta_L)$ and $h^*(b; \delta_H)$. Suppose for some stock, a higher value of δ *did not* result in a higher optimal extraction rate. Then it must be the case that the policy functions cross. Let the crossing occur at point $(h, b) = (\hat{h}, \hat{b})$, at which point: $h^*(\hat{b}; \delta_L) = h^*(\hat{b}; \delta_H)$. The crossing $(\hat{h}, \hat{b})$ can occur either to the right or the left of the δ_H steady state; we consider each case in turn.

If the crossing occurs to the left of the steady state, then $h^*(\hat{b}; \delta_H)$ must be steeper than $h^*(\hat{b}; \delta_L)$, i.e.

$$\frac{dh}{db}|_{\hat{h},\hat{b},\delta_H} = \frac{\dot{h}}{\dot{b}}|_{\hat{h},\hat{b},\delta_H} > \frac{dh}{db}|_{\hat{h},\hat{b},\delta_L} = \frac{\dot{h}}{\dot{b}}|_{\hat{h},\hat{b},\delta_L} \tag{11}$$

Invoking Equations 7 and 9, this inequality becomes:

$$\frac{\pi_h(\delta_H - g_b) - \pi_b}{\pi_{hh}(g - h)} > \frac{\pi_h(\delta_L - g_b) - \pi_b}{\pi_{hh}(g - h)} \tag{12}$$

Taking the difference between these expressions (which must be positive, if Equation 12 is to hold), we obtain: $\frac{\pi_h(\delta_H - \delta_L)}{\pi_{hh}(g-h)} < 0$, which delivers the contradiction.

If the crossing occurs to the right of the steady state, then $h^*(\hat{b}; \delta_H)$ must be shallower than $h^*(\hat{b};\delta_L)$. Following the procedure above, we obtain the inequality:

$$\frac{\pi_h(\delta_H - h_b) - \pi_b}{\pi_{hh}(g-h)} < \frac{\pi_h(\delta_L - h_b) - \pi_b}{\pi_{hh}(g-h)} \tag{13}$$

Taking the difference between these expressions (which must be negative, if Equation 13 is to hold), we obtain: $\frac{\pi_h(\delta_H - \delta_L)}{\pi_{hh}(g-h)} > 0$, which delivers the contradiction. □

Proposition 1 provides a powerful, and quite general result: for *any* stock level inherited by the regulator in period t, a higher discount rate results in a higher extraction rate. While this result certainly holds in steady state, Proposition 1 shows that this also holds out of steady state, given any biomass level., if she places more weight on the future (so δ is larger), she will optimally choose a lower extraction rate.

Recall that Equation 5 showed that weaker property rights to the incumbent industry translated into a high capture adjusted discount rate for the regulator. Viewed in that light, Proposition 1 gives rise to some testable hypotheses:

Result 1. *A stronger property right (i.e. higher θ) leads to a lower harvest rate holding resource stock constant.*

Proof. $\frac{d\Phi}{d\theta} = \phi(1-\gamma) > 0$. So an increase in θ causes an increase in the discount factor (Φ), a decrease in the discount rate (δ), and by Proposition 1, a decrease in the harvest rate. □

We also obtain:

Result 2. *The more influence the incumbent has relative to subsequent owners (i.e. lower γ), the stronger is Result 1.*

Proof. $\frac{d^2\Phi}{d\theta d\gamma} = -\phi < 0$, so an increase in gamma decreases the effect of θ on Φ and increases the effect of θ on δ, thus strengthening Result 1. □

3.1 Model Extensions

The analytical model makes several simplifying assumptions. Here we discuss two extensions to the baseline model, namely endogenous revocation probabilities and endogenous redistribution.

In addition to the choice of the aggregate harvest quota, it is conceivable that the regulator would also be charged with setting the probability of revocation itself. In our model it can easily be shown that, if future resource owners have less influence over the regulator than current owners (i.e. $\gamma < 1$), the regulator would choose strong property rights (i.e. zero probability of revocation). That is, the value of θ that maximizes the value of the resource will be zero.

To see this, simply note that by definition $\gamma < 1$ implies that the welfare of future resource owners is given less weight than the welfare of current owners. Furthermore, current owners are strictly better off by maintaining ownership of harvest rights. Therefore current owners would exert pressure on the regulator to choose strong property rights, given here by a zero probability of revocation.

The issue of *who* the next owner will be, should the rights of current owners be revoked, will depend on political considerations that ares outside of our model. If the regulator can choose, though, who the future owners are, we note that the regulator would choose to give rights back to the current owner under reasonable conditions.

To see this, suppose that the probability of revocation (θ) is fixed. If the regulator can choose how rights will be reallocated after expropriation, then her objective function would be maximized by giving rights back to current owners if $\gamma < 1$. If all owners, current and future, have the same influence over the regulator (i.e. $\gamma = 1$), then the regulator would be indifferent between giving the resource rights back to its current owners or reallocating among some new set of users.

4 An Application to Global Fisheries

The analytical model developed so far in this paper generates a clear set of predictions that can be tested empirically using data on exploitation rates and resource stocks. We use a novel data set covering nearly 200 of the world's largest commercial fisheries. The durability of the underlying property rights in fisheries varies dramatically. Some fish stocks are managed by some variant of regulated open access, where some constraints on entry may exist (e.g. restricting access to domestic vessels only), but where there are no individual property rights to the resource or its flow. On the opposite end of the spectrum are management regimes with secure, transferable property rights over harvest shares, with many fisheries falling somewhere in between. This variation in management over space and time will allow us to test whether property rights-based management leads to different exploitation rates than other management approaches.

We focus on catch share management such as the individual transferable quota (ITQ).[7] An ITQ program allocates a share of the overall harvest (the quota) in a fishery and time period to individuals. Rights are then defined as a share (i.e. a percentage) of the annual

[7] There are many types of so-called catch shares, and many variations for any given category. For the purposes of this paper, we follow the classifications laid out by Environmental Defense Fund (EDF) of catch share fisheries. For the most part, our categorizations aligned with those in Melnychuk, et al (2009). Details are available from the authors in an Appendix.

harvest quota for that management area and year. In practice the implementation of this type of management varies, with some catch share fisheries placing different restrictions on ownership and/or transferability,[8] but for purposes of this paper, we abstract away from those differences and categorize each fishery as being either a "catch share" or a non-catch share fishery, for simplicity referring to all types of rights-based management regimes as "ITQs". We then can test the main prediction of our analytical model, namely: do property rights (such as ITQs) lead to lower exploitation rates?

4.1 Data Overview

Our goal is to test whether individual property rights affect the aggregate exploitation rate in fisheries. We assume that under all types of management lobbying and political pressure will lead to the "capture" of the regulator. That is, we assume that the regulator's decision regarding exploitation reflects the will of the regulated fishermen. Given that the regulator is captured under both types of property rights regimes, our model predicts that exploitation rates should be lower when underlying property rights are strong.

We use a unique dataset that includes both exploitation rates and biomass estimates from historical fishery stock assessments. The stock assessment data come from the RAM II database, maintained at the University of Washington. The database includes large, commercially important fisheries from countries around the world. In the current analysis, there are 178 fisheries from 27 large marine ecosystems in the data.

The panel data contain annual fishery-level observations for historical exploitation and biomass. Both the exploitation rate and biomass are defined relative to those parameter

[8] See Grainger and Parker, 2013 for a discussion of the impacts of such restrictions.

values under "maximum sustained yield" (which come from surplus production models of the fishery). The two main variables of interest from the data include relative fishing mortality ($f_{it} = \frac{F_{it}}{F_i^{MSY}}$), or the overall exploitation rate for that fishery and year, and relative biomass ($b_{it} = \frac{B_{it}}{B_i^{MSY}}$). By normalizing by the fishery's maximum sustainable yield values, we can compare biomass and exploitation rates across fisheries and years. Note also that these variables are related to the extraction rate from our theoretical model by the following identity:

$$\frac{h_{it}}{MSY} \equiv \frac{F_{it}}{F_i^{MSY}} \frac{B_{it}}{B_i^{MSY}} \equiv f_{it} b_{it} \tag{14}$$

To test our theory regarding the impact of individual property rights on aggregate exploitation rates (through the captured regulator), we collected a history of management types for each fishery in the stock assessment database.[9] The main source was the EDF catch share database, which categorizes ITQ (catch share) management as well as other types of property rights. Our data are consistent with Melnychuk et al (2011). In addition, we collected ecological characteristics including species-specific traits, as well as socioeconomic characteristics of the fishery (which are limited given the scope of fisheries and time periods covered in the analysis).

We divide fisheries into the broad categories of non-ITQ fisheries and ITQ fisheries. This is largely due to data constraints, as it is difficult to fully characterize and quantify all of the combinations of regulations and property rights that a fishery may have at any point in time. For example, differences in gear restrictions, season lengths, licensing, and restrictions on the transferability of catch shares are not quantified. Instead, we adopt a broad definition

[9] The dataset is fully described in the Appendix.

of catch shares following EDF and Melnychuk et al (2011).[10] Any fisheries not managed by some form of "catch share" is treated the same, though we recognize that there could be important differences between purely open access fisheries and fisheries managed by limited entry (i.e. "regulated open access"). Again, given the scope of the fisheries and the long time period covered in this study, it is impractical (if not impossible) to fully characterize the regulations in place at any point in time.

One obvious empirical challenge is selection into treatment. Management is not exogenously determined, but in many cases the transition to property rights-based management occurs due to changes in policy at a higher political level. New Zealand's transition to ITQs in the early 1980s, for example, was largely due to liberalization reforms taking place throughout government (Rees, CITATION). And in the United States, federal policies surrounding ITQs influenced the starting date of catch share programs. Our empirical strategy involves exploiting the rollout of catch share programs over time to identify the impact of property rights on exploitation.

Of the 178 fisheries in the data, 78 have transitioned to some form of "catch share" rights-based management during the timeframe studied. Of those 78 fisheries that change management, 44 transition when relative biomass is low (i.e. $b_{it} = \frac{B_{it}}{B_i^{MSY}} < 1$), and 34 transition when relative biomass is high ($b_{it} = \frac{B_{it}}{B_i^{MSY}} > 1$). Importantly, the transition to ITQs does not appear to be systematically correlated with biomass levels or trends in exploitation leading up to the transition.

[10] We also include in our definition of "catch shares" any fisheries that are partially managed by catch shares as "treated." Any bias introduced in our definition of catch shares will make it less likely that we will find a significant effect.

To begin describing the data, we show the most recent cross-section of fisheries in (b,f) space, separating ITQ vs. non-ITQ management. A cross-sectional approach to the data reveals a striking result: ITQ fisheries tend to be managed significantly more conservatively than non-ITQ fisheries. Panel a) in Table 1 shows a simple scatter plot using the most recent year for the fisheries in the data. Relative exploitation rates (f) are plotted against relative biomass (b), with lines indicating $F = F_{MSY}$ and $B = B_{MSY}$. As shown in the figure, ITQ fisheries in a cross-section tend to have low relative exploitation rates and high relative biomass.

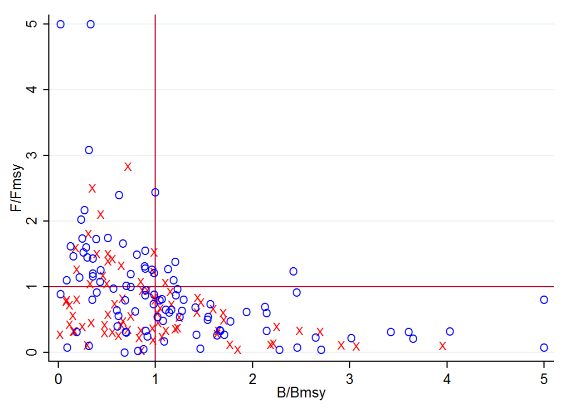
Using the most recent data in the sample we begin by estimating a simple regression of relative exploitation rates (f_i) on relative biomass (b_i), an indicator for ITQ management, and an interaction term between biomass and ITQ. The baseline specification is below:

$$f_i = \beta_0 + \beta_1 b_i + \beta_2 ITQ_i + \beta_3 ITQ * b_i + \epsilon_i \tag{15}$$

Results from this cross-sectional regression are shown in Table 1. This simple regression explains about 20% of the variation in f. The results suggest that, conditional on relative biomass levels, ITQ fisheries are managed more conservatively than other fisheries. The relative exploitation rate in ITQ fisheries is 0.33 units smaller than fisheries with comparable biomass levels. This impact is dampened for higher levels of biomass, suggesting that as biomass increases, the average ITQ fishery become less and less conservatively managed.

Moving beyond the recent cross-sectional differences, we next plot the exploitation rates for ITQ fisheries, "treated" fisheries that have not yet transitioned to ITQs, and other types of fisheries (that do not adopt ITQs in our sample period). Because exploitation rates are

Table 1: Cross-Sectional Differences in ITQ and Non-ITQ Fisheries



a) Scatter Plot by Management Type

Dependent Variable:	f
b	-0.397***
	(0.0861)
b*ITQ	0.192*
	(0.106)
ITQ	-0.330*
	(0.185)
intercept	1.300***
	(0.149)
N	178
R^2	0.1977

b) Cross-Sectional Regression

Notes: There are 178 fisheries included, using the most recent year for each fishery. In the left panel, a scatter plot of the data is shown; "x" denotes ITQ management and "o" denotes other management types. On the right, cross-sectional regression results are shown, using the most recent year available for each fishery in the dataset. Robust standard errors in parentheses. *, ** and *** correspond to significance at the 10%, 5%, and 1% levels, respectively.

expected to be a function of biomass, we break down the plots separately to show the median for "healthy" and "overfished" stocks.

Examining averages for ITQ, treated (pre-ITQ), and non-ITQ fisheries shows some general trends, where ITQ fisheries tend to have lower exploitation rates within any given year. However, even when examining the average exploitation rates for overfished and healthy stocks separately, it is difficult to disentangle the impact of ITQs on exploitation because of overall time trends and changes of which stocks are managed by ITQs over the sample period.

Ideally one would want to hold constant time-specific effects and fishery-specific differences, focusing only on differences in the timing of adoption of ITQs. Our approach in the following section will rely on the difference in timing of the implementation of rights-based management.

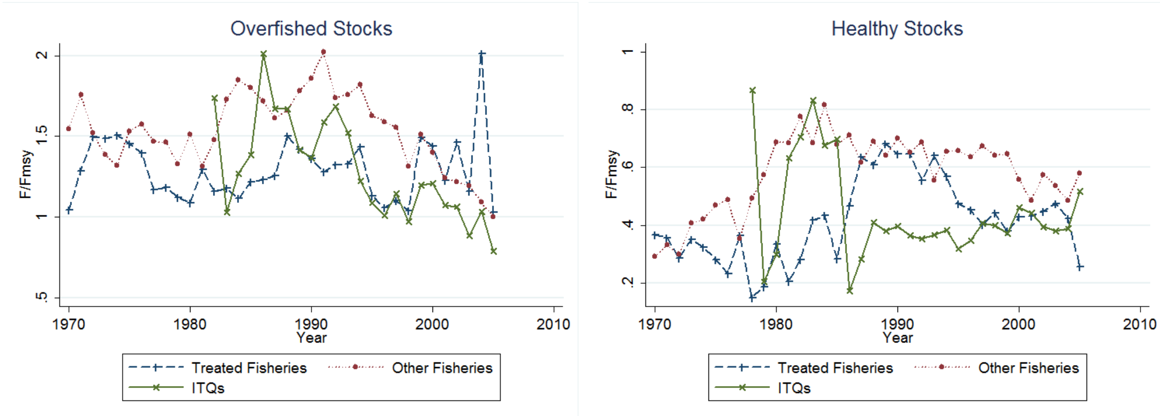


Figure 1: Notes: The figures above show the median exploitation rate for ITQ, "treated", and non-ITQ fisheries; the figures are separated into "overfished" vs. "healthy" stocks in that year.

4.2 Empirical Approach

We begin with a nonparametric approach to the data, looking for systematic deviations from the trend in exploitation rates when ITQs are introduced. It is useful to first examine the impact of ITQs on exploitation graphically. We begin by looking only at fisheries in our dataset that transition to ITQs at some point, exploiting the rollout in the timing of policy change. Because the effect of ITQs on exploitation rates may vary by fishery health, we separate the plots into "healthy" ($B > B^{MSY}$) and "overfished" ($B < B^{MSY}$) fisheries in the transition year. Because there are generally not more than one or two fisheries that change to ITQs in any given year, we divide the data into separate "cohorts" of ITQ adoption to examine the impacts of ITQs on both healthy and overfished stocks. To do so, we first create period dummy variables for years containing 5-year intervals. We then create indicator variables for ITQ adoption corresponding to each of these periods.

We restrict the sample to treated fisheries and regress f (i.e. F/F^{MSY}) on fishery dummy variables, period dummy variables, dummies for ITQ adoption in each period, and interactions for time period and the ITQ-period adoption dummy variables. We can then look at how exploitation rates change post-ITQ adoption, holding constant period- and fishery-specific characteristics.

The regression equation is given by

$$f_{ijt} = \theta_i + \lambda_t + 1(ITQ_{it}) * 1(i \in J)\tau_{ijt} + \epsilon_{ijt} \tag{16}$$

where J is a cohort of ITQ fisheries.

Because of the number of fixed effects estimated to produce these plots, we do not show the results of these estimations here. Instead, we plot the point estimates in the following two graphs separately for fisheries that "healthy" and overfished fisheries at the year of transition to ITQs. The average exploitation rate for each cohort is normalized such that the first year of ITQs is zero. An advantage of this approach is that it is nonparametric, and we can examine differences in exploitation rates under different property rights regimes over time with few assumptions. The plots show the group year averages, after controlling for year and fishery-specific fixed effects.

The first plot in Figure 2 shows the exploitation trends for overfished fisheries before and after the implementation of ITQs. Pre-ITQ, some cohorts show an increase in exploitation rates over time, while for others the exploitation rate is more-or-less constant pre-ITQ. After property rights are introduced (marked by t=0 on the horizontal axis), the increasing trend in exploitation rates tends to stop, or in a couple cases completely reverses. This is consistent with the predictions from the analytical model, which suggests that the introduction of property rights would lead to a lower aggregate harvest quota (through the regulator's choice

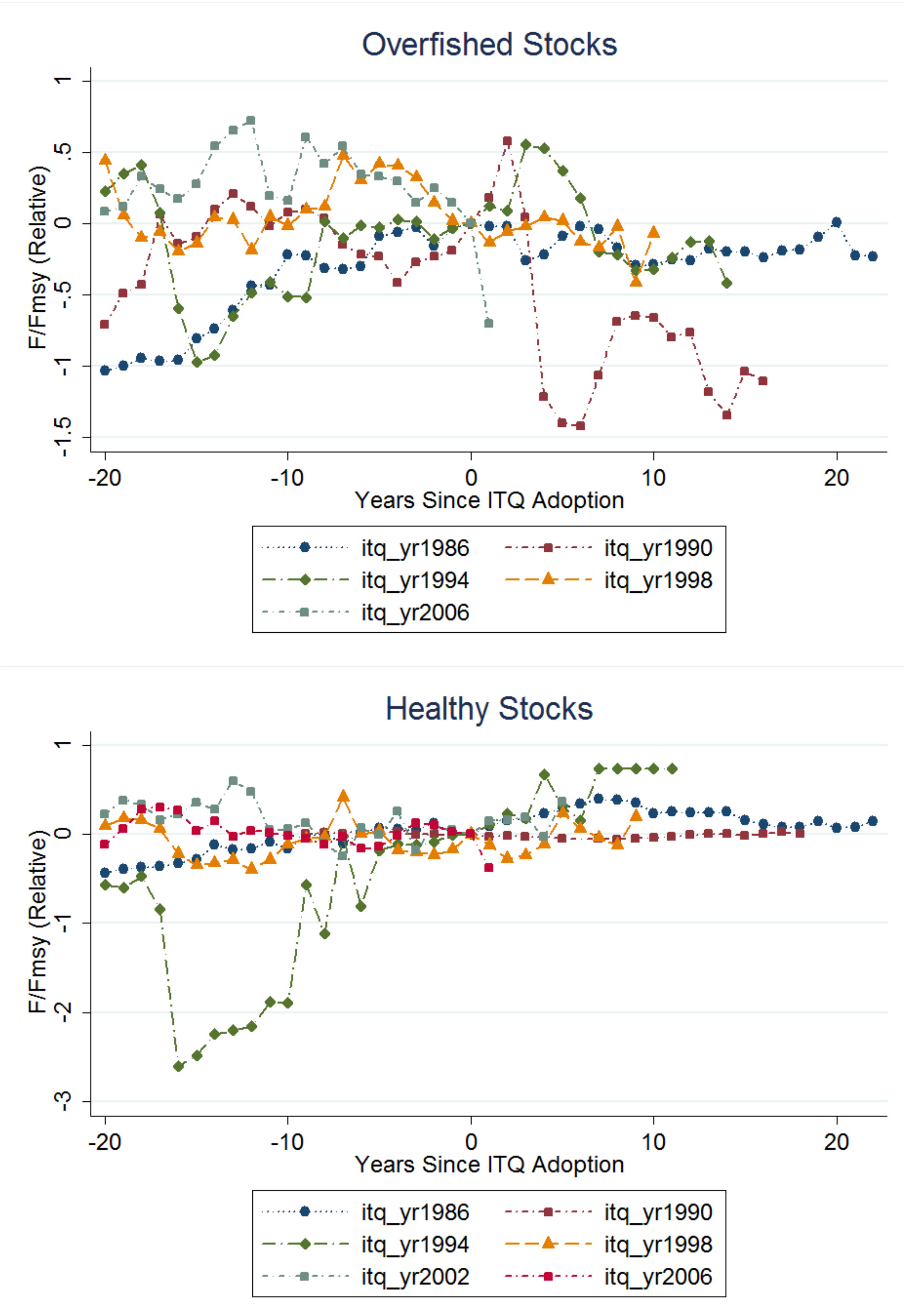


Figure 2: Notes: The figures above show the average effect of ITQs on exploitation for ITQs adopted in different year cohorts, shown separately for "overfished" vs. "healthy" stocks at the time of policy change. The variable "years since ITQ adoption" represents the number of years for that specific ITQ cohort. The sample is restricted to include only fisheries that adopt ITQs at some time in the data.

of the TAC).

The second plot in Figure 2 shows the exploitation trends for fisheries that are "healthy" ($b > 1$) at the time of adoption. This graph looks strikingly different from the overfished case. Pre-ITQ, exploitation rates are relatively constant (with the exection of the cohort of ITQ fisheries that transitioned in the early 2000s). Post-ITQ, exploitation rates remain constant, with the mean for some cohorts even increasing slightly (which is a movement toward the economically-efficient level of exploitation).

An advantage to this nonparametric approach is that it allows us to uncover trends in exploitation for different cohorts of ITQ fisheries, exploiting the timing in adoption. In the next section we estimate policy functions, which we need in order to test some of our propositions from the analytical model.

4.3 Fixed Effects Estimates

Since observations from the stock assessment data are at the fishery-year level, we can exploit the panel nature of the data (especially the rollout of ITQs) and estimate the impact of ITQs on exploitation, controlling for fishery-specific characteristics.[11]

We assume that the policy (f_{it}) is a function of the state (b_{it}) and a set of relevant ecological and socioeconomic characteristics. Such a function will determine the transition dynamics for the fishery i policy, f_{it} in any year. Fundamentally, this approach will allow us to estimate the implied policy function pursued by the fishery manager, and to determine

[11]Another method would be propensity score matching to attempt to control for any selection bias in ITQ adoption. Most of the variation in ITQ adoption can be explained by regional fixed effects, but predicting the *timing* of ITQ adoption would be a difficult task.

whether the presence of ITQs affects that policy function. The most basic specification is to regress f on b, an indicator for ITQs, an indicator variable for $b > 1$, and interaction terms, as the relative exploitation rate may respond differently for underexploited (i.e. "healthy") vs overexploited stocks. The exploitation rate is allowed to vary with the biomass level in the fishery. Fishery i's biomass in period t is given by b_{it}. This allows (but does not require) a manager's policy to be state-dependent. [12]

Fishery and year fixed effects are included to allow for unobserved heterogeneity in regions and/or time periods. As a result, we do not include time-invariant fishery-specific characteristics such as species traits. The functional form of the policy function is assumed to be piecewise linear in relative biomass, allowing for discontinuities and/or slope changes in some specifications at $B = B_{MSY}$. The baseline fixed-effects specification is given by

$$f_{it} = \alpha * ITQ_{it} + \beta * b_{it} + \gamma * ITQ_{it} * b_{it} + \phi_i + \theta_t + \epsilon_{it} \tag{17}$$

where f_{it} is the relative exploitation rate, F/F_{MSY}, for fishery i in year t; ITQ_{it} is an indicator for ITQ management in fishery i in year t; and b_{it} is the relative biomass in that fishery and year.

In the first table, specifications in columns (1) and (2) allow for discrete "jumps" in the policy function at $B = B_{MSY}$.[13] The key coefficient on ITQ suggests that the transition to ITQs is marked by a large, significant decrease in the exploitation rate. We also estimate the policy functions under alternative assumptions. First, we impose linearity and continuity

[12] In addition, there may be some "policy momentum", where this year's exploitation rate is a function of previous exploitation rates. In a subsequent specification we use a linear dynamic panel model for this case.

[13] In a subsequent specification we treat the panel as dynamic and allow for lagged dependent variables.

over the entire range of b_t for both ITQ and non-ITQ fisheries. Second, we allow for the policy to change at MSY biomass (b_t=1), but we impose the restriction that the policy cannot discretely jump at that point. These results are shown in specifications (3)-(6) of Table 2.[14]

The estimates in columns (1) and (2) suggest that individual property rights lead to a decrease in aggregate exploitation, and the impact is offset if the fishery is in good health. Estimates in (3)-(6) are similar in magnitude. ITQs lead to more conservative management, though this impact is not significant when year fixed effects are included.[15]. Examining the estimated year fixed effects shows a decrease in exploitation rates that coincides with the increased adoption of ITQs. Moreover, the interaction term $ITQ * Healthy$ is significant and positive, indicating that management becomes less conservative as biomass increases. In all specifications, the point estimates are large and negative, though it is not statistically significant when clustering standard errors at the fishery level. Across specifications, point estimates suggest that increases in biomass offset (partially) the impact of ITQs, though the interaction of ITQs and biomass is not significant in every specification.

In terms of magnitude, the estimates suggest that the implementation of ITQs leads to between a 0.13 and a 0.3 unit decrease in exploitation rates. This is partially (and in columns (1) and (2), fully) offset when the fishery is relatively healthy ($b > 1$) at the time

[14] Lemma 1 generates the hypothesis: $\frac{dh}{db} > 0$. Using the panel specification in Equation 17, and recognizing that h is proportional to fb, the sign of $\frac{dh}{db}$ equals the sign of $b(\beta + \gamma ITQ) + f$. Employing specification (3), for example this is clearly positive (because b and f are near 1), which confirms the theoretical prediction.

[15] Clustering on fisheries generates the most conservative standard errors, which are presented here. There are 27 large marine ecosystems represented in the data, and the number of fisheries in each LME varies, so clustering at the LME would require a different procedure such as a "wild bootstrap".

of transition to ITQs.

4.4 Dynamic Panel Estimator

In practice there could be constraints on how the regulator can change the exploitation rate over time.[16] This "policy momentum" suggests that the exploitation rate in any given year is likely to be a function of previous choices of f. As an alternative specification, we follow the approach of Arellano and Bond (1991) to estimate a linear dynamic panel model. Their estimator provides consistent GMM estimates for the regression parameters when the unobserved fixed effects are correlated with the lagged dependent variable, which is plausibly the case in our setting.

In Table 3, we show four sets of results. We allow for one and two lags of F/F^{msy}, respectively, as well as changes in the slope parameter on the ITQ indicator variable as B/B^{msy} changes. As the results show, ITQ is consistently negative and significant in these specifications, with magnitudes similar to our fixed effects specifications in table 2. On average, the exploitation rate (F/F^{msy}) is 0.20 units smaller in ITQ fisheries, conditional on fishery-specific and year-specific effects.[17]

[16]For example, the regulator may be required to (or simply prefer) to allow only gradual reductions or increases to aggregate exploitation from one year to the next.

[17]Moreover, consistent with the prediction in Lemma 1, the policy function is upward sloping in biomass.

Table 2: Fixed Effects Policy Function Estimates

	(1)	(2)	(3)	(4)	(5)	(6)
ITQ	-0.289	-0.180	-0.211	-0.115	-0.236	-0.133
	[0.0454]***	[0.0470]***	[0.0390]***	[0.0398]***	[0.0388]***	[0.0397]***
	(0.122)**	(0.129)	(0.112)*	(0.113)	(0.106)**	(0.110)
ITQ*Healthy	0.309	0.276				
	[0.0530]***	[0.0517]***				
	(0.121)**	(0.122)**				
Healthy	-0.352	-0.306				
	[0.0759]***	[0.0740]***				
	(0.146)***	(0.146)***				
B/Bmsy	-0.393	-0.373	-0.373	-0.362	-0.581	-0.548
	[0.0819]***	[0.0804] ***	[0.0182]***	[0.0178]***	[0.0662]***	[0.0647]***
	(0.140)***	(0.142)***	(0.0498)***	(0.0464)***	(0.131)***	(0.131)***
Healthy*(B/Bmsy-1)					0.250	0.221
					[0.0732]***	[0.0712]***
					(0.144)*	(0.145)
Healthy*(B/Bmsy)	0.109	0.0906				
	[0.146]	[0.150]				
	(0.0849)	(0.0830)				
ITQ*B/Bmsy			0.0648	0.0724	0.0766	0.0837
			[0.0133]***	[0.0138]***	[0.0130]***	[0.0136]***
			(0.0303)**	(0.0354)**	(0.0277)***	(0.0328)**
Fishery FE	Yes	Yes	Yes	Yes	Yes	Yes
Year FE	No	Yes	No	Yes	No	Yes
Observations	6,624	6,624	6,624	6,624	6,624	6,624
R-squared	0.618	0.638	0.614	0.636	0.615	0.637

Notes: The dependent variable is the relative exploitation rate (F/Fmsy). Observations are at the fishery-year level. ITQ is an indicator variable for catch share adoption in that year, which is defined to include other types of catch shares in addition to individual transferable quota. The variable "Healthy" is an indicator variable for B>Bmsy. Two sets of standard errors are included. In square brackets are heteroskedastic-robust standard errors; in parentheses the standard errors were clustered at the fishery level. ***,**, and * indicate significance at the 1%, 5%, and 10% levels, respectively.

Table 3: Arellano-Bond (Dynamic Panel) Policy Function Estimates

	(1)	(2)	(3)	(4)
ITQ	-0.207***	-0.187***	-0.208***	-0.187***
	(0.0528)	(0.0480)	(0.0591)	(0.0537)
ITQ*B/Bmsy			0.000837	0.000254
			(0.0328)	(0.0311)
B/Bmsy	0.134***	0.163***	0.134**	0.163***
	(0.0410)	(0.0418)	(0.0413)	(0.0421)
F/Fmsy (t-1)	0.726***	0.685***	0.726***	0.685***
	(0.0295)	(0.0336)	(0.0296)	(0.0336)
F/Fmsy (t-2)		0.0776**		0.0776**
		(0.0320)		(0.0320)
N	6,268	6,090	6,268	6,090

Notes: The dependent variable is the relative exploitation rate (F/Fmsy). Observations are at the fishery-year level. ITQ is an indicator variable for catch share adoption in that year, which is defined to include other types of catch shares in addition to individual transferable quota. Columns (1) and (3) include one lag of the dependent variable, whereas (2) and (4) include two lags.

5 Empirical Predictions and Implied Dynamics

So far we have focused on the effects of ITQs on fisheries exploitation through regulatory capture. By examining a panel of the world's largest stock-assessed fisheries, our empirical estimates show that fisheries managed with ITQs have a significantly lower exploitation rate than do non-ITQ fisheries. In this section we simulate how this reduction in fishing mortality leads to biomass changes for the resource. Coupling the empirical estimates from our econometric model of fisheries policy with the biological model dynamics allows us to how management affects resource dynamics over time in a phase plane.

Each econometric specification gives a different estimate of the policy function being pursued by the fishery manager, and in general the results show that ITQs lead to more conservative management, particularly when biomass levels are moderate or low. To simulate the bioeconomic system forward in time, we need to couple this policy function with a biological model. We use a surplus production (or logistic growth) model which takes the following form for fishery i:

$$B_{it+1} = B_{it} + r_i B_{it}\left(1 - \frac{B_{it}}{K_i}\right) - H_{it} \tag{18}$$

This describes the dynamics of biomass and exploitation rates as a function of the intrinsic growth rate, r, carrying capacity, K and harvests, H. We henceforth omit i noting that each fishery is assumed to be independent.[18] It is straightforward to show in this model that to achieve maximum sustainable yield, the following three conditions must hold:

$$B_{iMSY} = \frac{K}{2}, \tag{19}$$

$$F_{iMSY} = \frac{r}{2}, \tag{20}$$

$$H_{iMSY} = \frac{rK}{4} \tag{21}$$

Substituting these expressions into Equation 18, and coupling with our empirical model (Column 1 of Table 2)[19] above, gives the bioeconomic system that can easily be simulated.

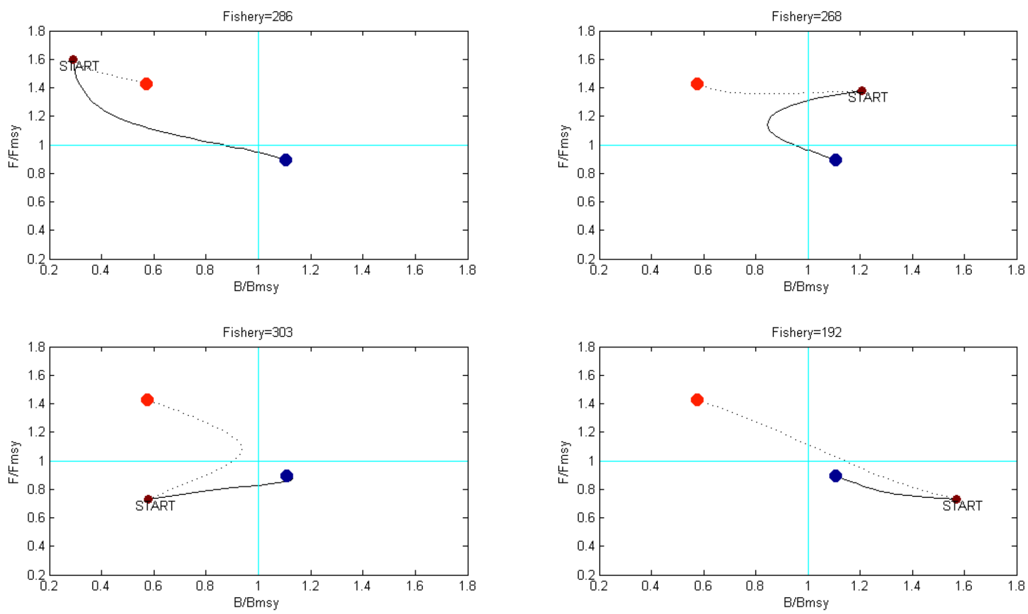
Simulating this system of equations with $ITQ_t = 0\ \forall t$ and for $ITQ_t = 1\ \forall t$ allows us to examine the likely future effects of ITQs on fishery dynamics. To display these dynamics in the phase plane, we will simulate the dynamics of four fisheries into the future. The four fisheries represent very different starting conditions. From each quadrant of the phase plane,

[18] This is obviously a simplification, as some stocks are highly interdependent, but a more complex analysis is far beyond the scope of the current paper.

[19] We could have used any of the specifications above. The results in column (1) of table 2 are particularly straightforward to interpret.

we have chosen a single fishery whose current level of (b, f) places it in that quadrant. For each of these four fisheries, we then run two forward simulations: one with ITQs ($C_t = 1$) and one without ITQs ($C_t = 0$). The results of these forward simulations are presented in Figure 3. Given our set of equations for simulating movement through (b,f) space, it is clear

Figure 3: Forecasts with ITQs (solid line) and non-ITQs (dotted line) for Four Fisheries.



Notes: Clockwise, starting in the upper left corner, are four fisheries. The top left is Herring ICES region 31, Bethnian Bay. The top right is Bigeye Tuna in the Western Pacific Ocean. Bottom right is Pacific Hake, and bottom left is ICES VIa Haddock (West of Scotland).

that the dynamics are independent of a fishery's carrying capacity (K), but do depend on its intrinsic rate of growth (r). The values of r for each fishery are calculated from data in the RAM II database. While the dynamics depend on r, the nature of the policy function is that each fishery's steady state will not depend on fishery-level biological traits.

Our forward simulations suggest that even if current biomass and current fishing mortality are similar across two fisheries, their ultimate steady states will differ substantially depending on whether they are managed with ITQs: on average our estimates suggest that ITQ-managed fisheries are predicted to have over twice the equilibrium biomass and roughly one-third lower steady state exploitation rates than are the non-ITQ fisheries.

This exercise is meant to be illustrative of the importance of differences in exploitation rates. Over time, lower exploitation rates arising from individual property rights can lead to large biomass differences over time. Moreover, this shows the importance of studying exploitation rates (such as F/Fmsy) rather than biomass levels, as changes in property rights would be expected to affect extraction immediately, but changes in biomass may not be achieved until the system has had time to respond. In particular, our results indicate that other studies (such as Melnychuk et al) may be premature in claiming that catch shares (such as ITQs) do not have a significant impact on biomass.

6 Conclusion

The literature has demonstrated a large, significant impact of property rights on economic outcomes, including natural resource extraction. Most of the existing empirical and theoretical evidence is about individual incentives: individuals have an incentive to overexploit a common pool resource when the underlying property rights are weak. However, the current literature has ignored the fact that extraction of natural resources in most cases is regulated by some entity. We introduce a regulator into a natural resource extraction model and demonstrate that under stronger property rights, individuals have an incentive to lobby for a lower extraction rate. The regulator, who balances the interests of current and (potential) future resource users, can be “captured” by resource users. When property rights are strong, current and future resource users are one-in-the-same, so the regulator’s choice is consistent with the maximization of the resource's value. When resource users have weak rights, however, there is an incentive for "capture" that leads to an inefficiently high rate of resource extraction.

The key prediction from our model is that individual property rights will cause the regulator to manage the resource more conservatively. We then test this prediction empirically using a novel dataset spanning nearly 200 large fisheries around the world. Roughly a quarter of the fisheries in the data have transitioned to some variant of an individual transferable quota system during the past thirty years. We exploit the rollout in management changes to test whether fisheries employing property rights-based management are managed more conservatively than other similar fisheries managed with weaker property rights. While we cannot differentiate between different types of catch shares (or different property rights strengths), future research could examine the impact of alternative property rights regimes on exploitation.

In our empirical test, we exploit the rollout of ITQs in different places over time. We find a significant, negative impact of property rights on overall exploitation rates in fisheries. That is, fisheries managed by some form of ITQ have significantly lower exploitation rates than otherwise similar fisheries, holding biomass and fishery-specific characteristics (such as biological characteristics and the economic value of the species) constant. Because all fisheries in the dataset have regulated harvest levels, we argue that differences in exploitation rates can be explained by different incentives for regulatory capture.

We then demonstrate that these differences in exploitation rates can have large impacts on steady state biomass levels by employing a simple dynamic surplus production function (i.e. a Gordon-Schaefer model). Relatively small differences in exploitation rates over a longer time horizon can lead to dramatically different extraction paths and steady state biomass levels (and hence ecosystem outcomes).

Our model and results demonstrate that, in the presence of individual property rights, the aggregate extraction rate can be influenced, even when determined by a regulator. It has been widely held that individual property rights will lead to more efficient harvest behavior by individuals, but our model and results suggest that the regulator's behavior also changes as a result of property rights.

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