# Can Environmental Markets Solve the Biodiversity Crisis?

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**Abstract**

We examine whether environmental markets – designed to achieve environmental goals in a flexible and economically-efficient manner – are effective at protecting biodiversity. We first develop a parsimonious model that uncovers how markets for quotas foster private economies of scale, which can however be associated with social diseconomies of scale. We test this central prediction in the context of commercial fishing, one of the main source of biodiversity depletion. We leverage stable national fishing quota shares in the Northern Atlantic Fishery combined with the staggered introduction of national markets for these quotas by three European countries as a source of exogenous variation in the exposure of fish populations to markets. We find that the adoption of environmental markets results in an overall decrease in both fish populations and annual fish landings. These effects are especially pronounced for species that are less commercially valuable, and more exposed to large scale extraction techniques. Environmental markets reshape the production function for the main natural resource by concentrating production among agents targeting this resource with large-scale assets, thereby exacerbating negative externalities on other parts of the ecosystem.

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*"If we save the sea, we save our world."*, Sir David Attenborough.

## 1 Introduction

Biodiversity is in decline around the world, both in terms of number of species and in overall abundance. Current extinction rates are thus estimated to be up to 1,000 times higher than their expected natural background rates (Pimm et al., 2014), while the World Wildlife Foundation has documented an average decline of almost 70% of monitored wild biomass across mammals, birds, amphibians, reptiles and fish. Unlike Earth's other mass extinctions, the primary force responsible for this destruction consists in externalities created by human economic activity (Barnosky et al., 2011).

For the last half century, environmental markets have been a canonical tool to address externalities in the natural world. Under the Coase Theorem, when property rights are well-defined and transaction costs are low, market mechanisms will lead agents to internalize externalities and achieve a social goal in an economically efficient way, regardless of the initial allocation of rights ((Coase, 1960) (Stigler, 1966)). Currently, environmental markets are used in a variety of contexts including carbon cap-and-trades, water rights, timber licenses, and air rights. Ayres et al. (2021) and Hernandez-Cortes and Meng (2023) provide evidence supporting the effectiveness of environmental markets in the context of water rights and carbon emissions. For biodiversity, environmental markets bear the promise to align private interests with the long-term sustainability of ecosystems (Heal, 2000) and to bring much needed private capital for ecosystem preservation (Karolyi and Tobin-de la Puente, 2023). However, biodiversity's high-dimensionality may cast a doubt on the effectiveness of this tool compared to contexts with a narrower outcome of interest. In practice, are environmental markets an effective tool to preserve biodiversity?

To answer this question, we first develop a basic dynamic model for renewable resource extraction in an ecological setting. In our model, we study how the introduction of cap-and-

trade for production rights reshapes firm incentives and, in turn, affects the biological steady states of the harvested species. In particular, if dirty producers are the lowest–marginal cost suppliers, then permit trading reallocates production toward them and environmental markets can decrease the long-term biomass of the ecosystem. Crucially, even with biologically sustainable species-level catch limits, permit trading can lower long-run biomass by shifting production toward dirtier firms and increasing non-harvest ecological damage.

We then test the central predictions of our model in the context of fisheries, one of the key settings where environmental markets have been implemented. This context is particularly important because marine ecosystems cover more than two-thirds of the planet and play a critical role in oxygen production, coastal protection, and climate regulation by absorbing and storing carbon.[1] Fisheries also provide approximately 20% of global animal protein consumption and support the livelihoods of 500 million people worldwide.[2] Moreover, they represent a sector where the direct exploitation of natural resources is a major driver of biodiversity loss (Jaureguiberry et al., 2022).

For our empirical analysis, we focus on the North Atlantic fisheries. In this region, commercial fishing is a first-order force as it removed an average of 21% of spawning fish biomass in 2022, with some stocks experiencing even steeper removal–for example, a Cod stock losing 66% and a Sprat stock losing 75% of their spawning biomass. This geography presents two additional key advantages for the purpose of answering our research question. First, it offers high-quality, homogeneous data covering fish populations, landings, and fishing fleets, with long and consistent time series. Second, the regulatory framework, which combines EU-wide rules with national-level decision-making, creates variation that support a causal

[1] IPCC (2019). Special Report on the Ocean and Cryosphere in a Changing Climate. Intergovernmental Panel on Climate Change. https://www.ipcc.ch/srocc/

[2] FAO (2022). The State of World Fisheries and Aquaculture 2022: Towards Blue Transformation. Food and Agriculture Organization of the United Nations.

identification strategy.

Our identification strategy builds on the concept of "Relative Stability" of EU Country quota shares. Since 1983 each European nation's share of the yearly fishing quota, a.k.a Total Allowable Catch (TAC), is fixed for a given sea zone and fish species. In response to the 1982 UN Convention on the Law of the Sea, European nations agreed to freeze fishing quotas to their 1970s level. While this policy was initially intended to be a stopgap measure, these shares are virtually unchanged today (Self, 2015). As a result, for the last forty years nations catch a relatively stable fraction of each fish species in a set geographic range. In addition, three countries adopted a national market for fishing quotas during our sample period: the Netherlands in 1994, the UK in 1999 and Denmark in 2007. Such adoption is unlikely to be correlated with the evolution of fish populations given that each country can only fish a fraction of any given fish population. We therefore construct a shift-share instrument that leverages these historical fishing shares with the national adoptions of a market for fishing quotas. We interpret this shift-share instrument as an exogenous source of variation in the exposure of fish populations to environmental markets, and measure the subsequent ecological impacts of this variation.

Our central result is to provide causal evidence that environmental markets can be detrimental to biodiversity. In our setting of the North Atlantic fishery, we find that the national adoption of transferable fishing quotas leads to a long-run decrease of fish species biomass, and consequently to fish landings. This fall occurs despite the European Common Fishery's policy of scientifically managing fishing quotas, which suggests that market allocation of quotas has a significant impact on the ecosystem even when controlling for production levels.

These effects present important heterogeneity by types of fish. The detrimental effects

are more pronounced for the fish which live primarily on the ocean floor, and for high-trophic fish. When splitting by the sample between species with high and low commercial value, we find that there is no depletion of the most commercially valuable fish. By contrast, fish with low commercial value gets particularly depleted when exposed to the market for fishing quota. We interpret these results as consistent with environmental markets exacerbating negative externalities to certain parts of the covered eco-system, and particularly the part with lower economic value as market participants are unlikely to internalize their depletion.

Having established the negative relationship between transferable fishing quotas and fish biomass, we investigate the mechanism underlying these effects. Using granular data from Denmark, we document that the nation's 2007 adoption of transferable quotas preceded an increase in the intensity of trawler fishing as quotas were almost entirely purchased by large trawler vessels. Groundfish, which include cod, haddock, and flounder, are especially vulnerable to the destruction caused by industrial bottom-trawling. This fishing technique catches fish by dragging a large net over the ocean floor and is a common source of seafloor damage and fish bycatch, where non-targeted species of fish are accidentally harvested. The downward pre-trend in trawler fishing in Denmark is fully reversed after the introduction of transferable quotas as the share of fishing went from 77% in 2007 to 93% in 2024.

We see similar trends in European landing data which is available starting in 2013. Nations that allocate their quotas through transferable markets show a statistically significant increase in landings from large trawlers, and a corresponding decrease in landings from both small vessels and vessels that use non-trawling fishing techniques.

These additional results are consistent with environmental markets fostering the consolidation of production to the most productive assets, which can also have the largest negative environmental externality. In this case, less profitable ships transfer their annual quotas

to a few large fish vessels which primarily utilize industrial trawling. This shift in fishing scale and technique to industrial trawling exacerbates by-catches, a negative environmental externality, despite production being capped.

Our paper contributes to several streams of the literature. First, we expand the growing literature about financial markets and biodiversity. This literature is interested in three main questions: How can financial structures be designed to preserve biodiversity? How will financial systems be affected by biodiversity depletion? How should we value biodiversity? On the question of mitigation, Flammer et al. (2025) argues that blended financing structures, which combine private and philanthropic or public capital, are an important tool for biodiversity preservation. On the adaptation question, biodiversity risk has been shown to effect equity pricing (see Giglio et al. (2023) and Garel et al. (2024)), but only pose a moderate risk to financial stability (Arlt et al., 2024). On the question of biodiversity value, Weitzman (1998) develops a ranking criterion to chose between preservation projects under a limited budget constraint. More recently, Frank and Sudarshan (2024) quantifies the social cost of the collapse of a keystone species using vultures in India and Giglio et al. (2025) provide an ecologically-founded model of biodiversity loss.

We build on this literature by documenting how different environmental markets, often viewed as a pre-requisite to financing biodiversity protection, can affect biodiversity themselves. Our results that asset transferability rights can be harmful to biodiversity raise important concerns about the role of traditional financial markets in environmental preservation.

Second, our setting of the fishery is tied closely to the economic literature on common-pool resource management. These papers study the governance structures around resources where exclusion is difficult, and consumption is rivalrous (e.g. (Warming, 1911), (Gordon, 1954),

(Ostrom, 2010), (Stavins, 2011)). Canonical examples include fisheries, forests, grazing land, and water rights. However, until recent years, a majority of the literature's empirical work has focused its attention to small settings with case study methodology (Ostrom, 1990). In line with a new generation of research (See Isaksen and Richter (2019)), we provide causally identified evidence to the subject of common-pool resource management. Closest to our work is Frank and Oremus (2023), who employ a difference-in-difference design between the US and the EU fisheries. They find that the adoption of scientific thresholds to rebuild fish population successfully increased fish population recovery in the US.

In contrast, our paper focuses on the effects of environmental markets for quotas under a uniform system of thresholds. Moreover, the unique setting of the European Common Fishery allows us to focus on an international setting, which poses novel challenges for resource management and to our knowledge is understudied in the empirical literature on biodiversity.

Third, our work is related to the long-running scholarship on corporate negative externalities that arise from contractual design (e.g. (Akey and Appel, 2021), (Kim and Xu, 2022), and (Bucourt, 2025)). There is relatively little work in this literature focused on how these negative externalities may be amplified through rights transferability.

Finally, our work contributes to a growing empirical literature on environmental markets. In carbon cap-and-trade markets, environmental markets have been shown to increase innovation in greener technology Calel (2020). In the setting of California, Ayres et al. (2021) finds that environmental markets for groundwater increased both land value and aquifer levels, while Hernandez-Cortes and Meng (2023) shows that the state's 2013 introduction of a carbon market narrowed pollution exposure gaps between disadvantaged and other communities. Aspelund and Russo (2024) finds that auctions in the US Conservation Reserve

Program, which aims to replace cropland with native vegetive cover, fails to account for the expected additionally of projects. Our work is the first to focus on the effectiveness of environmental markets for biodiversity. Biodiversity poses a unique challenge for environmental markets, as unlike carbon or groundwater, there is not an obvious single metric of resource health. We clarify this distinction and find that, even in an ideal setting, environmental markets may be detrimental to biodiversity.

The rest of the paper is organized as follows. Section 2 presents a basic theoretical framework for how the introduction of environmental markets may reshape natural resources standing. Section 3 introduces institutional background about the European Common Fishery, national markets for transferable quotas, and on fishing practices. Section 4 provides an overview of the data used in our analysis. Section 5 constructs our shift-share instrument and describes our empirical strategy. Section 6 shows our main results. Section 7 investigates the underlying economic mechanisms from our main results. Section 8 concludes.

## 2 Theoretical Framework

We first develop a basic dynamic model for renewable resource extraction in an ecological setting. The model highlights how environmental markets, in this case cap-and-trade for production rights, reshapes firm incentives and in turn affects the biological steady states of the harvested species.

### 2.1 Setup

Consider a discrete time, infinite horizon economy ($t = 0, 1, 2, \dots$) where $N$ firms extract biomass from two species $X$ and $Y$. $X$ is the main species of commercial value and $Y$ is a secondary species whose growth is interrelated to the main species $X$. In each period, firms

harvest biomass from population stocks $X_t$ and $Y_t$, where $X_t$ and $Y_t$ denote the biomass of species $X$ and $Y$, and sell each unit at constant prices $p_x$ and $p_y$. Following the ecological literature, the population dynamics for both of these species follow logistic growth models and so have growth proportional to their current population size. Additionally, we assume that both species are initially non-extinct and so satisfy the initial conditions $Y_0 > 0$ and $X_0 > 0$. The population for species $Y$ evolves according to the following law of motion:

$$Y_{t+1} = \underbrace{Y_t \; + \; r_y Y_t\Big(1 - \frac{Y_t}{K_y}\Big)}_{\text{Natural Net Growth}} \; - \; \underbrace{\Big(H_{y,t} \; + \; D_t\, Y_t\Big)}_{\text{Human Activity}} . \tag{1}$$

where $r_y > 0$ is the intrinsic net proportional growth rate of the population and the constant $K_y > 0$ is the environmental carrying capacity or saturation level of species $Y$. Human activity affects the population dynamic through the last two terms–$H_{y,t}$ is total harvest of species $Y$ in period $t$ and $D_t$ captures unintentional damage to the population from production externalities.

In the absence of human activity, when $H_{y,t} = 0$ and $D_t = 0$, the terms $K_y$ and $r_y$ have an intuitive ecological meaning. Under logistic growth, the dynamics admit a stable state at the carrying capacity level $Y_t = K_y$. Thus, for any initial population $Y_0 > 0$, the ecosystem will trend towards a unique biomass level $K_y$, which represents the natural carrying capacity for the species. The speed of this convergence is determined by the net proportional growth term $r_y$, which captures the difference between the population birth and mortality rate at a given biomass level. For example, a whale species with a long life cycle will have a lower proportional growth rate than a shrimp species with a rapid life cycle, and consequently will take longer to return to its carrying capacity.

The population of species $X$ evolves according to the following law of motion:

$$X_{t+1} = X_t + r_x X_t\Big(1 - \frac{X_t}{K_x + \kappa Y_t}\Big) \; - \; H_{x,t} \tag{2}$$

which has two major differences from species $Y$. First, the carrying capacity of the main species $X$ depends upon the secondary species so an increase in the biomass of species $Y$ increases the carrying capacity of species $X$. The constant $\kappa \geq 0$ captures this ecological relationship. Intuitively, species $Y$ can be thought of as an umbrella or keystone species, whose presence has a positive effect on the health of the habitat. Moreover, as $Y_t > 0$ we also have that $K_x + \kappa Y_t > 0$, so species $X$ always has a well defined carrying capacity.

Second, we assume there is no direct negative production externality on species $X$. Besides simplifying the model, this lack of negative externality can be motivated by the idea that the production technology was designed around the main species, but without concern for the secondary species. In fisheries, this production externality may represent bottom-trawling where the seafloor ecosystem of other species are damaged in order to catch the commercial target fish, which feeds on these secondary species.

The harvesting of both species is managed by a regulator who each period sets a production cap on the total allowable harvest. The aggregate harvest for each species is set by a simple rule based off the current population size:

$$H_{x,t} \equiv \theta_x X_t, \quad H_{y,t} \equiv \theta_y Y_t, \qquad \theta_x, \theta_y \in [0, 1]. \tag{3}$$

so a constant proportion of each species is harvested each period. Additionally, these rules are set so that the population absent any externalities will not be driven to extinction $\theta_x < r_x$ and $\theta_y < r_y$. In practice, natural resource management often resembles this kind of rule-

based policy, owing to ecological uncertainty and political-economy considerations that favor transparent and easily enforceable harvest rules. While aggregate production is set by the regulator, production choices are still made by firms each period to maximize their profits.

Each firm can be one of two types: $m$ firms use a dirty technology $c^d_{i,t}(q_{iX,t} + q_{iY,t})$ and the other $N - m$ firms use a clean production technology with cost function $c^c_{i,t}(q_{iX,t} + q_{iY,t})$. These costs, which are a function of the biomass of their joint-harvest of the two species, are twice continuously differentiable, increasing, and convex. Crucially, the marginal cost of these two types of firms may differ. For instance, when cleaner technology is cheaper than dirty technology:

$$c'^d_{i,t}(q_{iX,t} + q_{iY,t}) > c'^c_{i,t}(q_{iX,t} + q_{iY,t})$$

the model resembles a canonical cap-and-trade abatement case. However, another economic possibility is the setting where dirty technology is cheaper:

$$c'^d_{i,t}(q_{iX,t} + q_{iY,t}) < c'^c_{i,t}(q_{iX,t} + q_{iY,t})$$

While the relative private costs of clean and dirty production are ultimately an empirical question, the model clarifies that the environmental performance of cap-and-trade hinges on which technology has the lower marginal cost. In particular, if dirty producers are the lowest–marginal cost suppliers, permit trading reallocates production toward them and cap-and-trade can increase ecological damage. Moreover, unlike in many climate abatement contexts—where cleaner production is often explicitly incentivized and can coincide with lower long-run marginal costs—biodiversity protection frequently relies on costly avoidance and selectivity, making it more plausible that the low-cost producers are also the high-damage producers. To make matters even more challenging in biodiversity settings with

dozens of species, the ecological interaction term $\kappa$ is often unobserved, which makes targeted regulation of negative externalities infeasible. In the empirical section, we provide evidence that the introduction of tradable rights in the Northeast Atlantic Ocean led to an increase in dirty production and reduced the biomass of exposed fish.

The damage from the dirty technology is assumed to be a function of the quantity of biomass harvested using the dirty technology:

$$D_t = \delta \cdot \sum_{i \in \text{dirty}} (q_{ix,t} + q_{iy,t})$$

where $\delta > 0$ is a constant that scales the magnitude of the negative externality. As shown in Equation 1, this damage term directly reduces the growth of species $Y$.

### 2.2 Firm Choice

At the start of each period, the regulator allocates production shares $a_{ix,t}$ and $a_{iy,t}$ to each firm so that:

$$\sum_{i=1}^{N} a_{ix,t} = H_{x,t} = \theta_x X_t, \qquad \sum_{i=1}^{N} a_{iy,t} = H_{y,t} = \theta_y Y_t.$$

where the regulator sets total allowable harvest using the rules from Equation 3. These allocations ensure a firm with a constant proportion of the harvest over time:

$$a_{ix,t} = \alpha_{i,x} H_{x,t}, \qquad a_{iy,t} = \alpha_{i,y} H_{y,t}$$

and thus act as an asset that guarantees lifetime production rights of a fraction of the total harvest.

Furthermore, we restrict to the case where the regulator's production cap is economically

binding so that aggregate production is equal to the total allowable harvest:

$$H_{x,t} = \sum_{i=1}^{N} q_{ix,t}, \qquad H_{y,t} = \sum_{i=1}^{N} q_{iy,t} \tag{4}$$

Put differently, the prices are assumed to be high enough that all production rights are exercised in equilibrium. In this analysis, we will focus on two cases of allocating production rights and show each interacts with the long-run population dynamics of both species. First, we explore an autarkic benchmark case, where allocations are symmetric across all firms and rights are non-tradable and second we will explore a cap-and-trade setting where production rights are tradable in a frictionless market.

Under autarky, firm choice is trivial as they produce their allocated quota each period $q_{ix,t} = a_{ix,t}$ and $q_{iy,t} = a_{iy,t}$. Per period profit is then:

$$\pi^{A}_{i,t}(X(t), Y(t)) = p_x a_{ix,t} + p_y a_{iy,t} - c^{\tau_i}_{i,t}(a_{ix,t} + a_{iy,t})\,.$$

where $\tau_i \in d, c$ represents whether firm $i$ is clean or dirty. When markets are introduced firms are allowed to trade production rights each period, so they solve the static maximization problem:

$$\begin{aligned} \max_{q_{ix,t},\, q_{iy,t} \geq 0,\, b_{ix,t},\, b_{iy,t}} \quad & \pi^{T}_{i,t} = p_x q_{ix,t} + p_y q_{iy,t} - c^{\tau_i}_{i,t}(q_{ix,t} + q_{iy,t}) - s_{x,t} b_{ix,t} - s_{y,t} b_{iy,t} \\ \text{s.t.} \quad & q_{ix,t} = a_{ix,t} + b_{ix,t}, \qquad q_{iy,t} = a_{iy,t} + b_{iy,t} \end{aligned}$$

where $b_{ix,t}$ and $b_{iy,t}$ are bought production rights for a unit of biomass of their respective species, which are in zero-net supply. $s_{x,t}$ and $s_{y,t}$ are the market clearing price.

### 2.3 Equilibrium Characterization

To begin, we examine the steady states under both the symmetric autarkic benchmark and when there exists a market for production rights. Since firms are static and regulation is rule-based, long-run outcomes are pinned down by the steady states of the biological system. Because the harvest and damage term are proportional to the biomass, human activity can be interpreted as decreasing the natural rate of growth of the species:

$$\tilde{r}_y = \underbrace{r_y}_{\text{natural net growth}} - \underbrace{(\theta_y + D_y)}_{\text{human pressure}} = \underbrace{f_y}_{\text{fertility}} - \underbrace{m_y}_{\text{mortality}} - \underbrace{(\theta_y + D_y)}_{\text{human pressure}}.$$

so the law of motion may be rewritten as:

$$Y_{t+1} = Y_t + \tilde{r}_{y,t} Y_t - \frac{r_y}{K_y} Y_t^2,$$

This structure reveals a few important insights about the population dynamics. First if $\tilde{r}_y < 0$ this model trends towards an extinction steady-state as $\lim_{t\to\infty} Y_t = 0$. Thus if the production externality is large enough, $\theta_y + D_y > r_y$, the secondary species $Y$ can still be driven extinct even under this rule-based management. Second, as the net proportional growth rate $r_y$ varies across species the same level of human activity will have varying effects depending on these biological factors. Third, if $\tilde{r}_y > 0$ we have a globally asymptotically stable equilibrium where:

$$\bar{Y} = K_y \frac{\tilde{r}_y}{r_y}$$

where $K_y$ is the carrying capacity absent of human activity. Intuitively, more human activity reduces the steady-state population of the secondary species $Y$ and as $\kappa \geq 0$ there is indirect damage to the population of the main species. This intuition is formalized in the next lemma:

**Lemma 1** (Indirect ecological effect)**.** *In any interior steady state where the main species is viable* ($r_x > \theta_x$) *and damage reduces the secondary species* $\left(\frac{d\overline{Y}}{d\overline{D}} < 0\right)$*, the marginal effect of production damage on the main species is determined by:*

$$\operatorname{sign}\left(\frac{d\overline{X}}{d\overline{D}}\right) = -\operatorname{sign}(\kappa).$$

*Proof.* In steady state with $\bar{X} > 0$ and constant $\bar{Y}$, the law of motion

$$X_{t+1} = X_t + r_x X_t\Big(1 - \frac{X_t}{K_x + \kappa y_t}\Big) - H_{x,t}, \qquad H_{x,t} = \theta_x X_t,$$

implies

$$0 = r_x \bar{X}\Big(1 - \frac{\bar{X}}{K_x + \kappa \bar{Y}}\Big) - \theta_x \bar{X}.$$

Dividing by $\bar{X}$ and solving yields

$$\bar{X} = \Big(1 - \frac{\theta_x}{r_x}\Big)\big(K_x + \kappa \bar{Y}\big).$$

Differentiating with respect to $\bar{D}$ gives

$$\frac{d\bar{X}}{d\bar{D}} = \Big(1 - \frac{\theta_x}{r_x}\Big)\kappa \, \frac{d\bar{Y}}{d\bar{D}}.$$

Since $r_x > \theta_x$ and $\frac{d\bar{Y}}{d\bar{D}} < 0$, the sign of $\frac{d\bar{X}}{d\bar{D}}$ is $-\operatorname{sign}(\kappa)$, with equality to zero when $\kappa = 0$. □

One fact that follows immediately from the new steady state is that the biomass harvested per period should also change. When the steady-state biomass is lower than under the rule-based management then the steady-state harvest should also fall. Consequently, from an economic perspective this damage then reduces the sustainable flows from the resource. In

the next proposition, we show how the introduction of cap-and-trade can exacerbate or mitigate this production externality damage depending on the relative costs of technologies.

**Proposition 1** (Market Reallocation and Ecological Damage)**.** *Assume dirty firms have strictly lower marginal costs than clean firms:* $c_d'(q) < c_c'(q)$ *for* $\forall q$*. Comparing the tradable permit regime (T) to the autarkic benchmark (A):*

*(i) Aggregate damage is higher:* $D^T > D^A$*.*

*(ii) The steady-state secondary stock is lower:* $\overline{Y}^T < \overline{Y}^A$*.*

*(iii) If* $\kappa > 0$*, the steady-state primary stock is lower:* $\overline{X}^T < \overline{X}^A$*.*

*Proof.* Permit trading equalizes marginal costs by shifting production from high-cost (clean) firms to low-cost (dirty) firms. This increases the dirty share of production, $\phi^T > \phi^A$. Because damage $D$ is a linear function of production by dirty firms, total damage increases ($D^T > D^A$). Higher damage reduces the net growth of species $Y$, leading to a lower steady-state population $\overline{Y}^T$. From Lemma 1, when $\kappa > 0$, the decrease in $\overline{Y}$ leads to a corresponding decrease in $\overline{X}$. □

When the clean firms have a cost advantage the signs of the proposition switch. Intuitively, the introduction of cap-and-trade allocates production to the firms with the lowest private cost of production. When these low cost firms are green then cap-and-trade reduces the externality. However, when these low cost firms are dirty than the introduction of tradable assets leads to increased ecological damage. In the following proposition, we produce a condition for when this switch to cap-and-trade is beneficial from a resource extraction perspective:

**Proposition 2** (Economic Efficiency of Cap-and-Trade)**.** *If $C^A(\overline{H}^A)$ is the total steady-state cost under autarky and $C^T(\overline{H}^T)$ the total cost under Cap-and-trade then Cap-and-trade is long-run economically efficient if and only if the steady-state cost savings from permit trade exceed the value of the resulting biomass loss:*

$$C^A(\overline{H}^A) - C^T(\overline{H}^T) \geq p_x\theta_x(\overline{X}^A - \overline{X}^T) + p_y\theta_y(\overline{Y}^A - \overline{Y}^T)$$

*Proof.* Social welfare in the steady state is defined as total revenue from both species minus aggregate production costs. Trading is efficient if the net surplus under trade is greater than or equal to the surplus under autarky. Rearranging the welfare comparison to isolate costs and revenues yields the inequality. Applying the relation from Lemma 1, the right-hand side (the ecological loss) is equivalent to:

$$\left[p_y\theta_y + p_x\theta_x\left(1 - \frac{\theta_x}{r_x}\right)\kappa\right](\overline{Y}^A - \overline{Y}^T)$$

Thus, the market is efficient only if the gains from trade outweigh both the direct loss in species $Y$ and the indirect interaction-weighted loss in species $X$. □

When viewing both species as commodities to be extracted then the economic efficiency depends upon the direct economic loss from the species $Y$, the indirect effect loss on the main species $X$, and the relative cost saving from transferring production. This proposition also nests many common cases in resource management. For instance, as is often the case outside of fisheries, and the secondary species is of no commercial value, then we may set $p_y = 0$ and damage of the externality is only felt through its indirect effect on $X$. To recover a naive species by species model, we may set ecological link $\kappa = 0$ so then $\overline{X}^A - \overline{X}^T = 0$ and

the only relevant effect is the direct effect on the secondary species $Y$.

This equation also provides insight in the challenges of directly regulating the production technology. While the relative cost saving between the technologies may be observable, the ecological effects of the production technology are difficult to measure. This difficulty is compounded when there are more than 2 species each with their own ecological link. Thus while a Pigovian tax or a ban on the dirty production is a theoretically possible solution, it depends on accurately measuring the complex interactions of the ecosystem.

In the remaining of the paper, we bring our theoretical insights to the empirical setting of commercial fishing. We use the North Atlantic Fisheries, and the related regulation from European Union countries, as a research laboratory.

# 3 Institutional Background

In this section, we provide background information first on the institution of the European Common Fisheries Policy, followed by the national introduction of transferable quotas, and finally on fishing practices and technology.

## 3.1 European Common Fisheries Policy

When the Common Fisheries Policy was first passed in 1983, it was never intended to be the landmark structure it later became. Throughout the 1970s, member states in the Atlantic shared equal access to European fishing grounds. However, the 1982 UN's Convention on the Law of the Sea threatened to disrupt this open access by allowing member states to vastly expand their economic exclusive zones into traditionally shared waters. As an immediate response, the EU passed a temporary compromise to preserve the status quo by freezing European fishing at its historical levels from 1973-1978.

Crucially, starting in 1983, national catch shares became relatively fixed across different fish species and ocean geography. While the European Common Fishery is able to set the total allowable catch, the share of this catch, even today, is primarily determined by these historical fishing levels from the 1970s. Despite multiple subsequent reforms to the European Common Fishery, the 'Principle of Relative Stability' is still a foundational part of the Common Fisheries Policy even four decades after its passing.

An example of the stability can be seen in the catch share of cod in the North Sea. As shown in Panel A of Table 1, the Netherlands' 1982 catch share of total allowable catch is 11%. Consequently, in the years from 2001 to 2020, the Netherlands' annual catch share is remarkably stable, always between 9% and 12%.

INSERT TABLE 1

### 3.2 How Nations Allocate their Fishing Rights

Under the European Common Fishery, member states have close to full independence on how they allocate their own national catch shares. As a result, there is considerable heterogeneity in the method of allocation across the Atlantic. For example, Belgium uses a system of command and control, where a government ministry allocates its quotas each year. In contrast, the Netherlands deploys a market system of individual transferable quotas, where fishermen are free to sell or lease their fishing rights. Other nations still, have a hybrid system that allows some market transferability and some direct government control.

For the interest of this paper, three nations allocate their fishing rights primarily through environmental markets– Netherlands starting in 1994, United Kingdom in 1999, and Denmark in 2007. In each of these nations, fishing quotas are assets owned by individuals and are not directly tied to vessels or uncertain government permitting processes. The quotas in

these nations are property that can be easily transferred, leased, or sold.

### 3.3 Fishing Techniques

There are multiple technologies and techniques that are used in modern commercial fishing, each often designed to catch a specific type of fish. Moreover, there is heterogeneity in the effectiveness and externality of within technique based on technological factors such as net-mesh size or operational factors such as the selectivity of the fishermen. For the purpose of this paper, we will simplify and only focus on two fishing techniques-trawling and seiners.

Both of these techniques catch fish with the use of a large net. Seiners catch their targeted fish by deploying a large net around a school of their targeted fish. In ideal settings, the seiners net is also tailored to their targeted fish to avoid accidentally capturing either juvenile fish or other species of fish. In contrast, trawlers catch fish through driving dragging, or 'trawling,' a large net through the ocean. Because trawling involves dragging a net through water, this technique is much often more fuel intensive, but also more cost effective than fishing with seiners. In Europe, trawling along the ocean floor contributes close to a third of annual landings in total tonnage, while in seiners compose close to a quarter of landings.

One of the primary negative externalities for fishermen arises when other fish are accidentally caught. These accidental catches are referred to as bycatch and face a high mortality rate even when immediately discarded back into the ocean. The accidental damage to the non-targeted species of fish may have downstream effects on other populations of fish, especially when the non-targeted fish are juvenile. Moreover, because bycatch is not usually shored it is difficult to regulate unless an observer is directly monitoring vessels at sea.

Trawling typically has a much higher rate of bycatch than seiners. A trawler's large net, especially when dragged along the ocean floor, indiscriminately captures a large amount of

non-targeted fish.[3] In contrast, a seiner net targeting a school of fish will have much lower bycatch rate.

# 4 Data

We collect data from a combination of scientific and government sources. Specifically, we use data on fish populations from the intergovernmental science organization ICES, fishing production data released by the Common Fisheries Policy, property strength measurements released by the European Commission, and use fishing quota data as transcribed from official treaty documents.

## 4.1 Catch Shares

We create the historical catch shares data by transcribing the quotas from the Council Regulation (EEC) No 172/83 of 25 January 1983. This initial document includes quota data for seven nations and 19 different fish species (See Appendix B2 for a complete list of these fish). In Table 1 Panel A, we provide a digitized example of the catch share structure.

INSERT TABLE 1

In addition, we supplement these initial shares with quota data collected from 2001 via the NGO, The New Economic Foundation, who transcribed all European quotas from official documents for the years 2001-2020. Importantly, this dataset adds the sizable fishing quotas of Spain and Portugal to our analysis, as well a species of fish that were not commercially important enough to add formally into the original 1983 agreement. We find that the historical catch shares are indeed highly stable over the following four decades. Because of

[3]Incidentally, bottom-trawling also interferes with the carbon sink process and contributed to an extra 1.47 Pg of aqueous CO2 emissions between 2016 and 2019 (Hiddink et al., 2021)

this stability, we include these later shares into our historical catch share dataset. Results restricting our analysis to the core 1982 shares is highly similar, albeit underpowered.

In Appendix Table, we show summary statistics for both the 1982 catch shares and the 2001 shares C1. In Table 1 Panel B, we provide a summary statistics table of our preferred measurement of catch shares, which we use in our main results.

### 4.2 Data on Fish Population

For over a century, the International Council for the Exploration of the Sea (ICES) has produced annual estimates of fish population in the Northeast Atlantic. As the world's oldest intergovernmental science organization, ICES estimates of fish populations are considered as the gold standard in Europe. Indeed, their independent scientific assessments are used a formal benchmark when the European Fisheries Council sets the annual total allowable catch.

Instead of releasing population data on an individual sea zone level, ICES estimates are calculated on a fish population level. These populations correspond to the underlying ecology and migration patterns of the fish population. While ICES records the corresponding sea zones, these population estimates do not necessarily map one to one with either fishing quota ranges or preconceived geographies of the ocean. Moreover, these ICES ranges typically differ across fish species.

For each fish population, our main variable of interest is the annual average estimate for spawning stock biomass. More precisely, this variable is the estimated total tonnage of fish capable of reproduction. This measurement is both economically and ecologically important. For fishermen, the spawning biomass is the total stock of fish that can be extracted each year. Moreover, it the main determinant of future extractions as spawning biomass directly shapes

future population growth. Ecologically, this is the measurement of choice for biologists as it is highly informative of ecosystem health.

As an alternative measurement of fish health, we also use ICES estimates of annual landings. This metric has obvious importance to the fishing industry and allows us to capture ecological effects that are diminished by quota design. For instance, if trawling is especially destructive in one population, then the Fishery Council may purposefully lower the annual allowed catch in that zone to achieve its target. In this case, two populations may have identical stock size, but systematically different landings.

In Panel A of 2, we provide summary statistics for both landing and spawning stock biomass from 1980 to 2024. We also include basic statistics on the composition of our fish populations.

INSERT TABLE 2

### 4.3 Data on Fishing Production Function

To explore the effects of ITQ adoption, we employ comprehensive production data from Denmark from 2001 to 2024. This data was initially provided by the University of Copenhagen and later provided by the Danish government. Using a survey of vessels that covers approximately 80% of annual landings, the government releases production statistics across vessel lengths and gear type. Crucially, these statistics allow us to observe the evolution of fish landings across different vessels and, starting in 2009, to track transfers of annual quotas.

When turning to evidence across the EU, we use the EU Fleet register, which is the official database of all registered fishing vessels in Europe. This database provides basic information on the full universe of European vessels starting in 1989. Specifically, it includes

physical details about vessels such as their length and tonnage, as well as economic details such as gear type and age. Importantly for our purposes, we can track the entrance and exit of fishing vessels over time as these events are explicitly monitored by the EU. In our subsequent analysis, we investigate how the adoption of ITQ effects the retirement of fishing vessels. Panel B Table 2 presents summary statistics on the national retirements from 1990 to 2024.

To analyze how the production function of fishing across the European Common Fishery, we use landing data from the "Scientific, Technical and Economic support to the Common Fisheries Policy (STECF), which available starting in 2013. For each country and vessel group, this quarterly data has the official record of fish landed in each ICES sea zone.

One major limitation of this data is that for vessel groups with less than three vessels, the EU censors their summary statistics in order to preserve privacy. However, because the STECF also publishes uncensored aggregate statistics on the EU level, we can observe the magnitude of missing data for each combination of nation, fish species, sea zone, and vessel category. For our main analysis, we exclude observations which are missing more than 25% of their landings. In Appendix B, we provide robustness checks to this decision. In Panel C Table 2, we present summary statistics for the cross section of this data.

## 5 Empirical Setting and Strategy

In this section, we first describe the construction of our main explanatory variable, which captures a fish population's average exposure to an individual transferable-quota market system, and then introduce our primary empirical strategy.

### 5.1 Shift-Share Instrument

We define our main explanatory variable for fish population $f$ in year $t$ as:

$$\text{Transferable Quota Exposure}_{f,t} = \sum_{k} \text{Catch Shares}_{f,k} \times \text{ITQ Adoption}_{k,t},$$

where $\text{Catch Shares}_{f,k}$ is the fraction fished by country $k$ in 1982 and $\text{ITQ Adoption}_{k,t}$ is an dummy variable that indicates whether country $k$ has adopted an individual transferable quota system. This shift-share instrument therefore measures the share of a fish population's harvest coming from nations that allocate quotas via a fully transferable market system.

In Figure 1, we provide an example of $\text{Catch Shares}_{f,k}$ for the United Kingdom and France across populations of cod. This figure suggests that even within a single species of fish there is significant heterogeneity in national catch shares. In Appendix A, we provide more formal analysis explaining the source of this share variation following the example of (Goldsmith-Pinkham et al., 2020).

INSERT FIGURE 1

Figure 2 presents the evolution of Transferable Quota Exposure as the Netherlands, United Kingdom, and Denmark introduced ITQ. As the Transferable Quota Exposure is defined on a fish population level, we present the average exposure across sea zones in this figure. These panels show that average exposure varies significantly over time and geography.

INSERT FIGURE 2

### 5.2 Main Specification and Identification

Our empirical strategy exploits the historical variation in catch shares in order to estimate the relationship between market introduction and biodiversity. For our main specification,

we employ the following reduced form regression:

$$Y_{f,t} = \beta \, Transferable \ Quota \ Exposure + \delta_t + \mu_f + \epsilon_{f,t}, \tag{5}$$

where $Y_{f,t}$ is an outcome variable measuring the health or commercial value of a fish population $f$ in year $t$. In our preferred specification, we use population's biomass as this is the preferred measurement of ecologists to estimate a species health and abundance. In other specifications, we use annual landing tonnage as our outcome variable as the total allowable catch is endogenous to fish stock size.[4] Additionally, to estimate the relationship with economic outcomes we use the commercial counterparts to these outcome variables with fish stock value and annual landing value. $\delta_t$ and $\mu_f$ are year and fish population fixed effects, respectively. We always include these controls in every version of our main specification to absorb the composition effects on fish time-invariant characteristics, as well as general trends in the North Atlantic ecosystem.

As an alternative specification, we deploy the following event study:

$$Y_{f,t} = \sum_{k=-G}^{M} \beta_k \, \text{High Exposure}_{f,t+k} + \delta_t + \mu_f + \epsilon_{f,t} \, . \tag{6}$$

where $\text{High Exposure}_{f,t+k}$ is a dummy variable that indicates when a fish population's Transferable Quota Exposure is above the median of its 2010 levels. In practice, $\text{High Exposure}_{f,t+k}$ is equal to 1 when more than 38% of its catch share is from a country using transferable quotas. This treatment variable is robust to alternative choices of cutoffs.

The main identifying assumption in equation 5 and 6 is that, in the absence of ITQ

[4]The total allowable catch uses scientific benchmarks around fish health to maximize long-term sustainable yields. Whether this advice is strictly adhered to is a subject of controversy. However, in theory, the Common Fishery could adjust the annual total allowable catch to offset any gains or losses to the fish health.

adoption, our outcome variables for fish populations exposed to high versus low shares would have trended the same. In other words, our shift-share estimate can be interpreted causally as long as, conditional on observables, national catch shares predict changes in fish populations only through our described channel (Borusyak et al., 2025).

Although it is impossible to directly test this assumption, we provide several checks to verify it. First, because the Common Fishery froze catch shares at their average level from 1973 to 1978, there is a large time lag between their determination and the first major adoption of modern ITQ in 1994. This twenty-year lag between our treatment and initial shares complicates both reverse causality and the possibility of joint determination. Second, we show that our results are robust, albeit weakened, when using shares instruments directly by providing unpooled analysis of our main specification for individual countries. Third, we show that our shift-share estimate is not overly dependent on researcher choice by providing our main specification across every pairwise combination of shares. Lastly, we provide additional robustness checks for the United Kingdom, which is the most influential country in our analysis and show that the results are not biased by events such as Brexit. In Appendix A, we provide formal analysis investigating the validity of our identifying assumption.

## 6 Main Results

In this section, we provide our main estimates from equation 5 and 6 and establish a negative relationship between the introduction of a market of transferable quotas and biodiversity.

### 6.1 Effects on Fish Population and Landings

In Panel A Figure 3 estimates equation 6 using fish biomass as its outcome variable. This event study shows that in the years following the adoption of ITQ, highly exposed fish popu-

lations tend to trend downward. This relationship becomes clearer and statistical significant in Panel B when the sample is restricted to groundfish, which are particular vulnerable to trawling.

INSERT FIGURE 3

In Figure 4, we provide the same specification, but using annual tonnage of landings as the outcome variable. Once again, we observe a gradual downward trend that is especially pronounced when restricting the sample to groundfish.

INSERT FIGURE 4

### 6.2 Heterogeneity of Effects

#### 6.2.1 Heterogeneity By Ecological Characteristics

In Table 3, we report our baseline estimation of $\beta$ from equation 5 using varying sample splits and measurements of biodiversity. In Panel A displays our estimates across the entire sample and for groundfish and non-groundfish. As previously shown in the event studies, the introduction of transferable quotas causes a decrease in both fish biomass and annual landings. This negative effect appears to be primarily driven by the significant decrease in groundfish. When removing groundfish, as in Column 3 and Column 6, the relationship loses its statistical significance and the coefficients move towards zero. However, when restricting the sample to only groundfish, the implied effects are ecologically meaningful with a coefficient of -0.47 for biomass and -0.72 for annual landings. In other words, if a groundfish population was fully governed under a market of transferable quotas, we should expect the biomass to be close to half of its size under a a system with no transferable quotas.

Panel B of Table 3 shows the same specification, but restricting to fish relatively higher or lower in the food chain. Using a fish's average trophic level as reported by Fishbase, we can observe the distance a fish is from the start of the food chain. To exploit this heterogeneity, we divide our sample into high trophic and low trophic by using a simple cutoff at the median trophic level in our sample. An example of a relatively low trophic fish would be anchovies, which is 3.1 steps removed from the base of the food chain, and an example of relatively higher trophic fish is cod, which is on average 4.4 steps removed. In this panel, we observed that the implied negative effects are far stronger in fish that are higher on the food chain.

INSERT TABLE 3

These results establish that the introduction of markets has an overall negative effect on ocean biodiversity. However, the negative effects are especially pronounced for groundfish that are particularly vulnerable to the damage from industrial trawling and for fish that are relatively higher up on the food chain.

#### 6.2.2 Heterogeneity By Commercial Value

In Figure 5, we turn our attention to the economic impacts of this biodiversity decline. This figure estimates equation 6 using the value of annual landings as priced by the historical fish prices in the United Kingdom. As before, we observe a gradual downward trend in landing value after the adoption of transferable quotas. These results are statistically significant across the entire sample of fish, and when restricted to only groundfish.

INSERT FIGURE 5

Panel A of Table 4, reports baseline estimation of $\beta$ from equation 5 using economic outcome variables. The outcome variable in Columns 1-3 is the yearly market price of the

fish multiplied by the population's total stock. This variable can be interpreted as a static measurement of the value of the population if it was to be fully consumed in that year. In Column 4-6 the outcome variable is the value of annual landings. Once again, we see a pronounced decline the value of groundfish after the introduction of transferable quota markets. Interestingly, we see a positive increase in the value of non-groundfish in both their stock and annual landings.

Panel B presents our baseline estimation using non-economic outcome variables, but with sample split by the fish commercial importance. This panel shows that the negative relationship between transferability and fish population are heterogeneous–fish with the more commercial value show a statistical insignificant positive increase in their population, while all other fish show a statistical significant decrease in their population. In Column 1, we restrict our sample to the top five most commercially important fish as measured by the total value of landings in 2015. In Column 2, we restrict to the top ten most commercially important fish. In Column 3, we restrict to the set of fish that were regulated in the original 1983 treaty. In Columns 4-6, we show the compliments of these sub-sample. While commercial important fish show statistically insignificant increase their biomass, this relationship monotonically decreases as the sample includes a larger sample of fish. Likewise, the negative effect for non-commercial important fish monotonically decreases as the sample excludes commercial important fish.

INSERT TABLE 4

## 7 Environmental Markets and Production Functions

In this section, we provide evidence that the adoption of a market for quotas shifts the fishing production function towards large industrial trawling. While the introduction of

quota markets achieves its intended objective of reducing fleet over capacity, the remaining fishing activity is increasingly concentrated to highly productive trawlers, which purchase the vast majority of transferred quotas. We first provide evidence from Denmark, which provides extensive production data in the years preceding and following its adoption of ITQ in 2007. From there, we explore evidence from the EU landing data over the last decade, and conclude with data from the EU Fleet Register, which tracks all active fishing vessels in the EU starting in the early 1990s.

### 7.1 Evidence from Denmark

Each year since 2001, the University of Copenhagen and the Danish Ministry of Food, Agriculture and Fisheries, provide comprehensive fishing data based on a sample of Danish fishing vessels. This sample captures around 80% of total Danish fishing by landings and is representative of vessels across different sizes and gear types. Using this dataset, we find evidence that the Danish quota market increases the share of trawling on the extensive margin and increases the share of the largest trawlers on the intensive margin.

Figure 6 presents evidence on how the Danish adoption of transferable quotas effected their trawling on the extensive margin. Panel A presents the annual value of transferred quotas for vessels that use trawling and for vessels that do not trawl. Although, the system of transferable quotas was fully implemented in 2007, comparable data on these transfers is only available starting in 2009. For these initial missing two year, we plot the transfers as zeros. Despite this limitation, it is clear that the vast majority of annual quotas transfers are being purchased by trawlers. In Panel B, there is a corresponding increase in the share of fish landings from trawlers. Indeed, the introduction of a quota market seems to fully reverse the downward pre-trend in trawling as the share of trawling increases approximately

15% points over the following decade.

INSERT FIGURE 6

Figure 7 provides evidence of a similar effect on the intensive margin as larger trawlers increase their market share. Panel A shows that trawlers over 40 meters purchase the bulk of transferable quotas and Panel B displays a corresponding increase in landings by tonnage. As before, a downward pre-trend in landings is fully reversed for trawlers over 40 meters, while other trawler landings continue to decrease.

INSERT FIGURE 7

### 7.2 Evidence from EU Landing Data

Starting in 2013, the European Common Fishery begins to produce comprehensive data on all European fishing activity. Crucially, the EU publicly provide annual fish landing data grouped by fish species, sea zone, nationality, vessel size, and gear type. Although, in cases when there are less than three vessels within a group, the EU censors data from this group to avoid releasing fishing data on individual vessels. To avoid bias from this data limitation, we restrict our analysis to sea zones where we observe at least 80% of the total production.

In Panel A of Table 5, we provide evidence that the amount of fish landed by large trawlers has increased from 2013 to 2021 in nations with a fully functioning quota market. In Column 1, we regress the log difference in fish landed by large vessels on an indicator variable of whether a nation uses transferable quotas. We include fixed effects sea zone and fish family fixed effects to absorb time-invariant geographic or biological trends. In Column 2, we repeat the same analysis using an alternative measurement of property strength from Marine Research Assessment Group, which provides a continuous measurement of the ease of

transferability in EU fisheries. In Column 3 and 4, we conduct the same analysis for landings by trawlers and in Columns 5 and 6 we do the same for large trawlers. Nations using an transferable quotas s system see an approximately a 26% increase in trawler landings and 38% increase in trawler landings from large vessels between 2013 and 2021.

Panel B of Table 5 provides evidence that shift to large trawlers has come at the expense of smaller and non-trawling vessels. In Columns 1 and 2, we employ the same analysis for small vessels, in Columns 3 and 4 for non-trawling vessels, and in Columns 5 and 6 for small non-trawling vessels. We find that nations with an transferable quotas system see an approximately 70% decrease in landings from small vessels and a 78% decrease in landings from small vessels that employ non-trawling fishing gear between 2013 and 2021.

INSERT TABLE 5

### 7.3 Evidence from EU Vessel Register

Using the EU fleet register, we track basic data on every European vessel since the 1987. This data includes the nationality, length, age, and gear type used by each fishing vessel. Most importantly, vessel retirement, scrapping, exports, and inactivity, are fully captured by this dataset. Although, these vessel exits are an imperfect proxy for real production, they provide a useful measurement of fleet reduction.

In Panel A Table 6 presents a Cox Proportional Hazard model of how national adoption of ITQ effects the likelihood of vessel retirement. Column 1 shows that the adoption of transferable quotas is predictive of vessel retirement, but that this relationship is not statistically significant. However, ITQ becomes becomes highly predictive of retirement after the 2013 EU reforms to the common fishery. Column 3 indicates an even stronger relationship when restricting to the universe of trawlers with the adoption of ITQ making trawler retire-

ment over 60 times more likely. Column 4 shows this relationship is also strengthened in the years following the 2013 reforms. Column 5 and 6 also show adoption of transferable quotas increases the likelihood of large vessel retirement.

In Panel B, a similar analysis is employed using ordinary least square regression. In this panel, the outcome variable is the national annual number of vessel retirements, which is regressed on an indicator variable that is equal to 1 if a nation uses transferable quotas in that year. We also include fixed effects for each country and year. As in our previous model, the adoption of ITQ is predictive of vessel retirement. While Column 1 shows no predictive relationship between transferable quotas and retirements across all vessels, Column 2 suggests that nations have a 15% increase in the number of vessels that are exported. Column 3 shows a statistically significant 263% increase in the number of trawlers retired and Column 4 shows a non-statistically significant increase 175% in trawler exports. Column 5 shows an 137% in the number of large vessels retired and Column 6 shows a statistically significant increase of 127% of large vessels that are exported after the adoption of transferable quotas.

INSERT TABLE 6

These retirement results are consistent with the consolidation of trawler production seen in the Danish data. As vessels are able to transfer their fishing quota, they chose to sell their annual fishing rights to the vessels with the lowest production cost. This is in alinement with the retirement effects being driven by inefficient trawlers and large vessels transferring their quotas to more efficient large trawlers. Moreover, this fits with anecdotal evidence that suggests smaller fisherman, predominately own smaller and non-trawling vessels, are less willing to retire their vessels as they view the profession as a 'way of life'.

# 8 Conclusion

This paper finds that environmental markets are often ineffective at protecting biodiversity, as they can exacerbate the negative externality. We establish this central fact both theoretically and empirically. To test our model's predictions, we develop a shift-share instrument that exploits plausible exogenous variation from the historical fishing shares in conjunction with the staggered introduction of national environmental markets.

In our reduced form evidence, we document that fish populations that are differentially exposed to market mechanisms show a decrease in their biomass and annual landings. These effects are heterogenous across fish ecological and commercial characteristics with particular negative effects for ground fish and fish higher up on the food chain. We argue that this result is caused by environmental markets concentrating production into technology that requires large-scale assets and produces a negative externality on other parts of the ecosystem. In the case of fishing, this technology is exemplified by industrial bottom-trawling, while in agriculture this technology may be monoculture cropping or concentrated animal-feeding operations. Likewise, in the environmental market for carbon credits this externality may take the form of intensive monoculture forestry.

In summary, these findings caution that even well-regulated environmental markets may exacerbate negative externalities in ecological settings.

# References


Akey, Pat, and Ian Appel, 2021, The limits of limited liability: Evidence from industrial pollution, *Journal of Finance* 76, 5–55.

Arlt, Sophia, Tobias Berg, Xander Hut, and Daniel Streitz, 2024, A biodiversity stress test of the financial system, *Working Paper* .

Aspelund, Karl M., and Anna Russo, 2024, Additionality and asymmetric information in environmental markets: Evidence from conservation auctions, *Working Paper* .

Ayres, Andrew B., Kyle C. Meng, and Andrew J. Plantinga, 2021, Do environmental markets improve on open access? evidence from california groundwater rights, *Journal of Political Economy* 129.

Barnosky, Anthony D., Nicholas Matzke, Susumu Tomiya, Guinevere O. U. Wogan, Brian Swartz, Tiago B. Quental, Charles Marshall, Jenny L. McGuire, Emily L. Lindsey, Kaitlin C. Maguire, Ben Mersey, and Elizabeth A. Ferrer, 2011, Has the earth's sixth mass extinction already arrived?, *Nature* 471, 51–57.

Borusyak, Kirill, Peter Hull, and Xavier Jaravel, 2025, A practical guide to shift-share instruments, *Journal of Economic Perspectives* 39, 181–204.

Bucourt, Noémie, 2025, Real effects of personal liability: Evidence from industrial pollution, *Working Paper* .

Calel, Raphael, 2020, Adopt or innovate: Understanding technological responses to cap-and-trade, *American Economic Journal: Economic Policy* 12, 170–201.

Coase, R. H., 1960, The problem of social cost, *The Journal of Law and Economics* 56.

Flammer, Caroline, Thomas Giroux, and Geoffrey M. Heal, 2025, Biodiversity finance, *Journal of Financial Economics* 164.

Frank, Eyal, and Kimberly Oremus, 2023, Regulating biological resources: Lessons from marine fisheries in the united states, *Becker Friedman Institute for Economics Working Paper* .

Frank, Eyal, and Anant Sudarshan, 2024, The social costs of keystone species collapse: Evidence from the decline of vultures in india, *American Economic Review* 114, 3007–3040.

Garel, Alexandre, Arthur Romec, Zacharias Sautner, and Alexander F. Wagner, 2024, Do investors care about biodiversity?, *Review of Finance* 28, 1151–1186.

Giglio, Stefano, Theresa Kuchler, Johannes Stroebel, and Olivier Wang, 2025, The economics of biodiversity loss, *Working Paper* .

Giglio, Stefano, Theresa Kuchler, Johannes Stroebel, and Xuran Zeng, 2023, Biodiversity risk, *Working Paper* .

Goldsmith-Pinkham, Paul, Issac Sorkin, and Henry Swift, 2020, Bartik instruments: What, when, why, and how, *American Economic Review* 110, 2586–2624.

Gordon, H. Scott, 1954, The economic theory of a common-property resource: The fishery, *Journal of Political Economy* 62(2), 124–42.

Heal, Geoffrey M., 2000, *Nature and the Marketplace: Capturing The Value Of Ecosystem Services* (Island Press).

Hernandez-Cortes, Danae, and Kyle C. Meng, 2023, Do environmental markets cause environmental injustice? evidence from california's carbon market, *Journal of Political Economy* 217.

Hiddink, Jan Geert, Sebastiaan J. van de Velde, Robert A. McConnaughey, Emil De Borger, Justin Tiano, Michel J. Kaiser, Andrew K. Sweetman, and Marija Sciberras, 2021, Quantifying the carbon benefits of ending bottom trawling, *Nature* 617.

Isaksen, Elisabeth Thuestad, and Andries Richter, 2019, Tragedy, property rights, and the commons: Investigating the causal relationship from institutions to ecosystem collapse, *Journal of the Association of Environmental and Resource Economists* 6.

Jaureguiberry, Pedro, Nicolas Titeux, Martin Wiemers, Diana E. Bowler, Luca Coscieme, Abigail S. Golden, Carlos A. Guerra, Ute Jacob, Yasuo Takahashi, Josef Settele, Sandra Díaz, Zsolt Molnár, and Andy Purvis, 2022, The direct drivers of recent global anthropogenic biodiversity loss, *Science Advances* 8.

Karolyi, George Andrew, and John Tobin-de la Puente, 2023, Biodiversity finance: A call for research into financing nature, *Finance Management* 52, 231–251.

Kim, Taehyun, and Qiping Xu, 2022, Financial constraints and corporate environmental policies, *The Review of Financial Studies* 35, 576–635.

Ostrom, Elinor, 1990, *Governing the Commons: The Evolution of Institutions for Collective Action.* (Cambridge University Press, Cambridge, UK).

Ostrom, Elinor, 2010, Beyond markets and states: Polycentric governance of complex economic systems, *American Economic Review* 100, 641–672.

Pimm, S. L., C. N. Jenkins, R. Abell, T. M. Brooks, J. L. Gittleman, L. N. Joppa, P. H. Raven, C. M. Roberts, and J. O. Sexton, 2014, The biodiversity of species and their rates of extinction, distribution, and protection, *Science* 344.

Self, Emily, 2015, Who speaks for the fish? the tragedy of europe's common fisheries policy, *Vanderbilt Journal of Transnational Law* 48.

Stavins, Robert N., 2011, The problem of the commons: Still unsettled after 100 years, *American Economic Review* 101(1), 81–108.

Stigler, George, 1966, *The Theory of Price* (Macmillan).

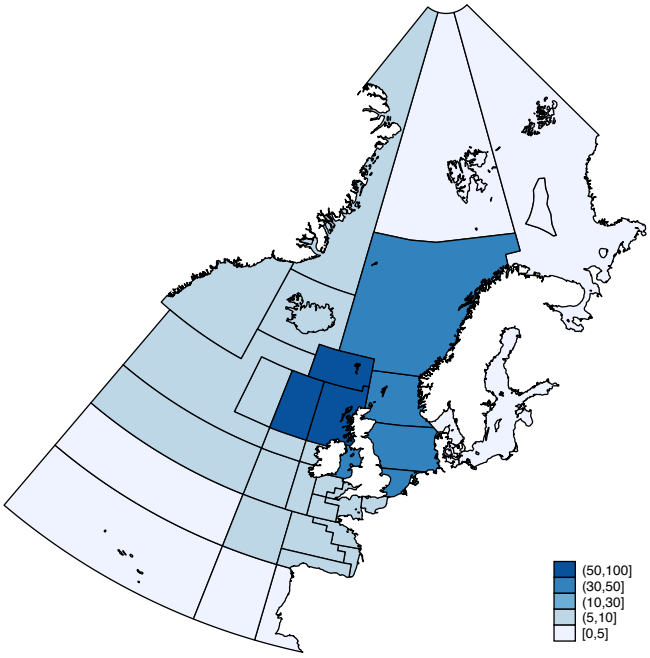
Warming, Jens, 1911, Om 'grundrente' af fiskegrunde [on the rent of fishing grounds]., *Nationaløkonomisk Tidsskrift* 49, 499–505.

Weitzman, Martin, 1998, The noah's ark problem, *Econometrica* 66, 1279–1298.

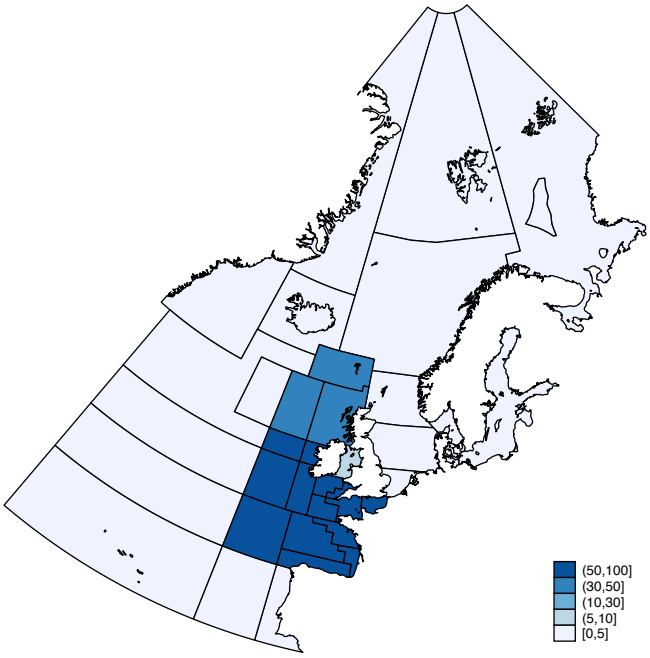
# 9 Figures

Figure 1: **Example of Quota Catch Shares**

**Panel A: British Catch Share of Cod**

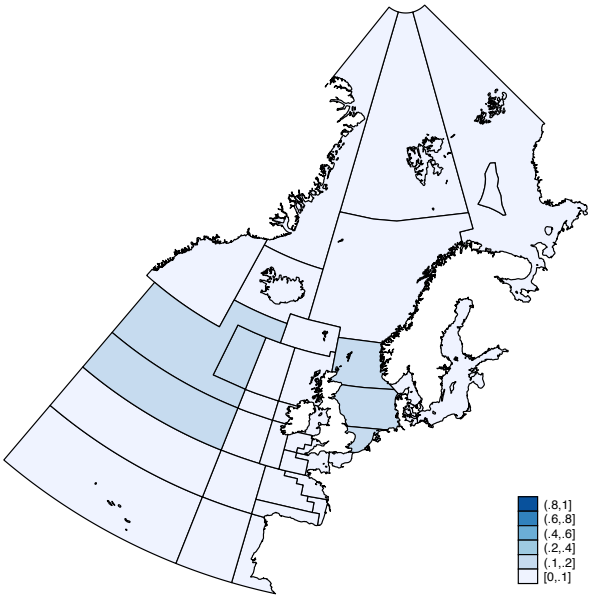



**Panel B: French Catch Share of Cod**

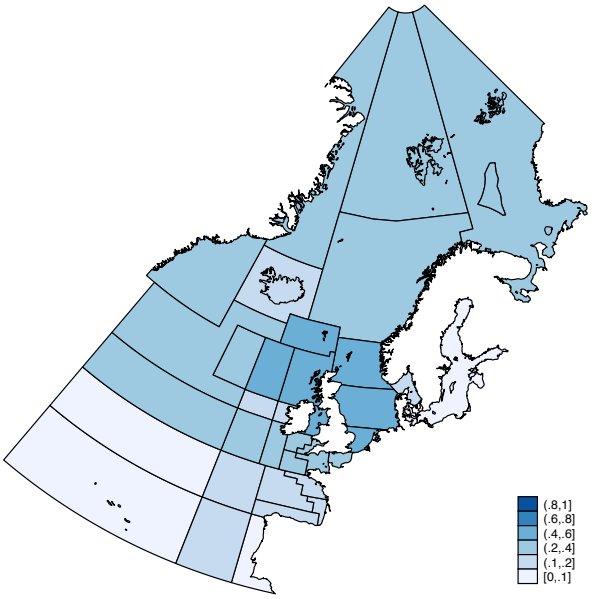



**Notes:** This figure presents examples of catch shares from 1983. Panel A presents the United Kingdom's catch share of cod and Panel B presents France's share of cod.

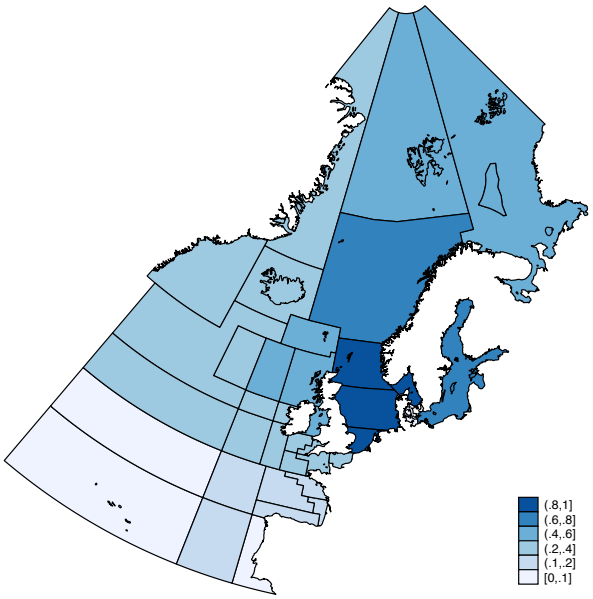
Figure 2: **Evolution of Average Transferability Exposure**




Netherlands 1994




United Kingdom 1999

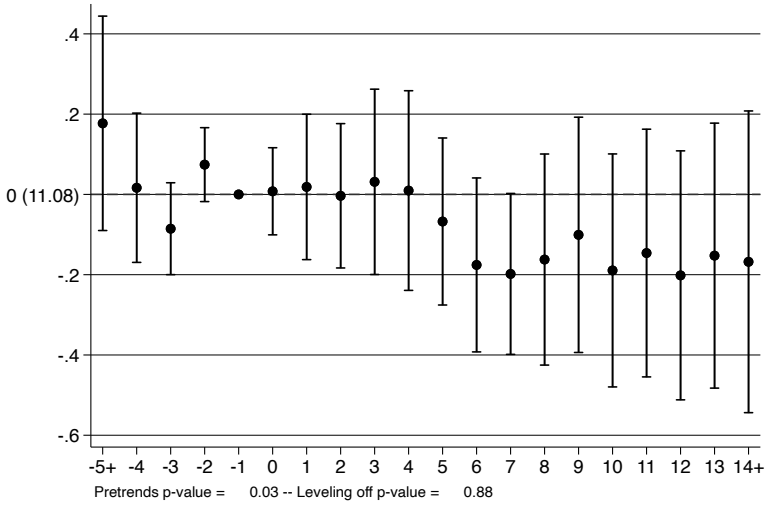



Denmark 2007

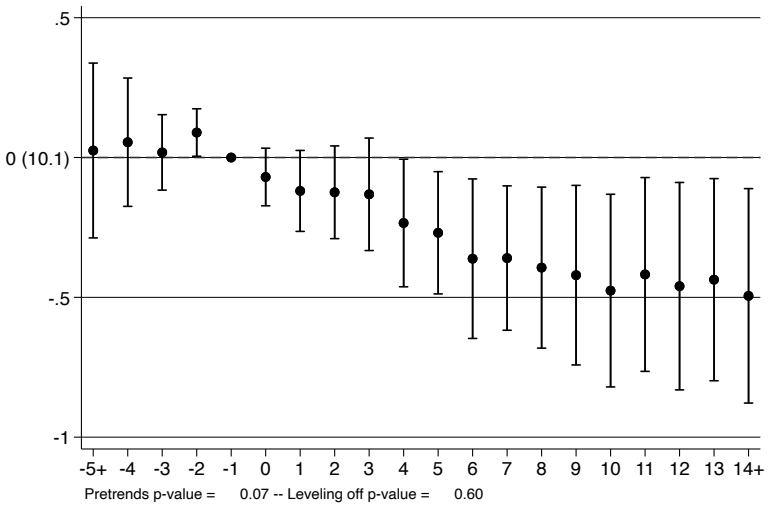
**Notes:** This figure shows the evolution of our main explanatory variable averaged across sea zones. We take a weighted average of our shift-share instrument, which is defined on a fish population level, for each ICES sea zone and year. Each panel presents this average after the adoption of an ITQ system– Panel A is in 1994 when the Netherlands adopted ITQ, Panel B 1999 for the United Kingdom, and 2007 for Denmark.

Figure 3: **Effects of Quota Markets on Fish Population**

**Panel A: All Fish**



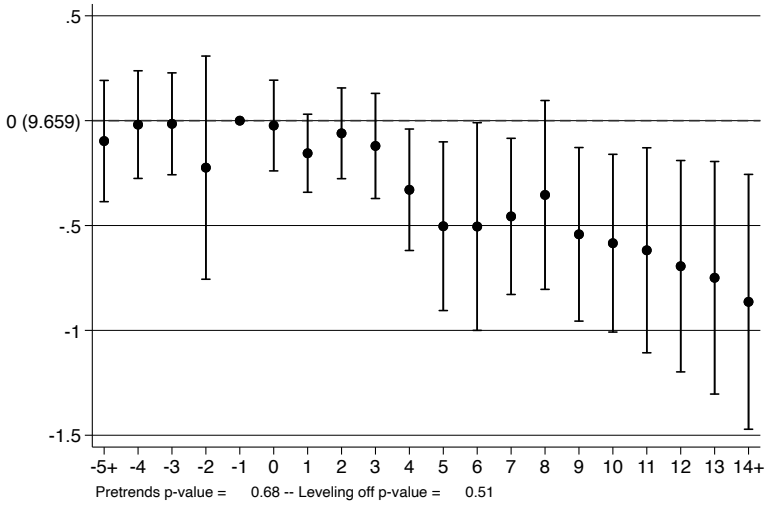

**Panel B: Groundfish**



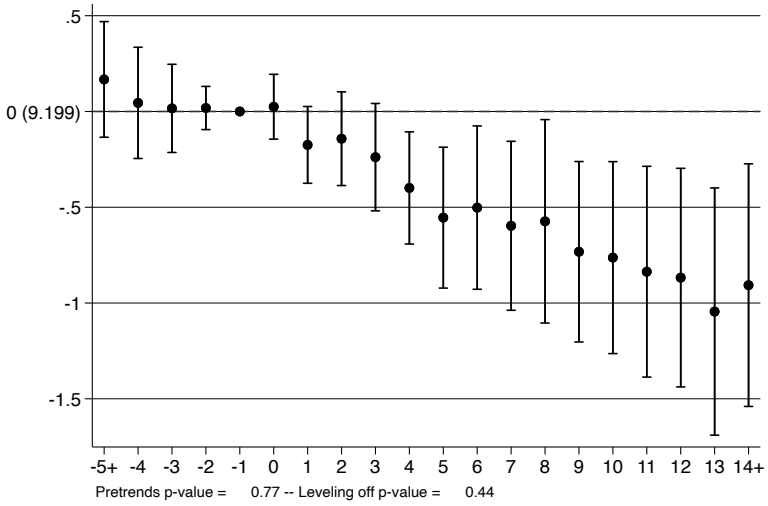

**Notes:** This figure plots the OLS coefficients and their corresponding 95% intervals from an event study on our treatment variable on fish population. Panel A provides an event study figure regressing log(biomass) of a fish population on an indicator variable that is 1 when the treatment is above median of the maximum of property strength. This corresponds to a fish population being treated when approximately 40% of its fishing are landed by nations with using an ITQ. Panel B provides identical analysis, but restricted to the sample of fish that are groundfish. This model employs fish population and year fixed effects and standard errors are clustered at the fish population level.

Figure 4: **Effects of Quota Markets on Annual Landings**

**Panel A: All Fish**



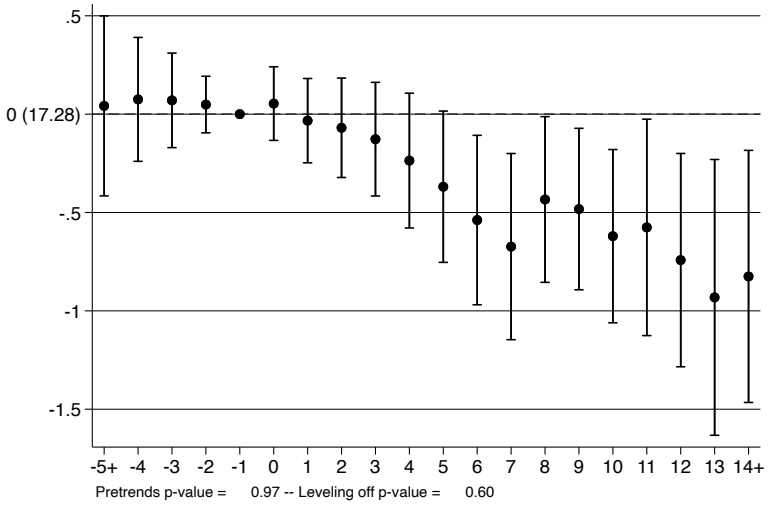

**Panel B: Groundfish**



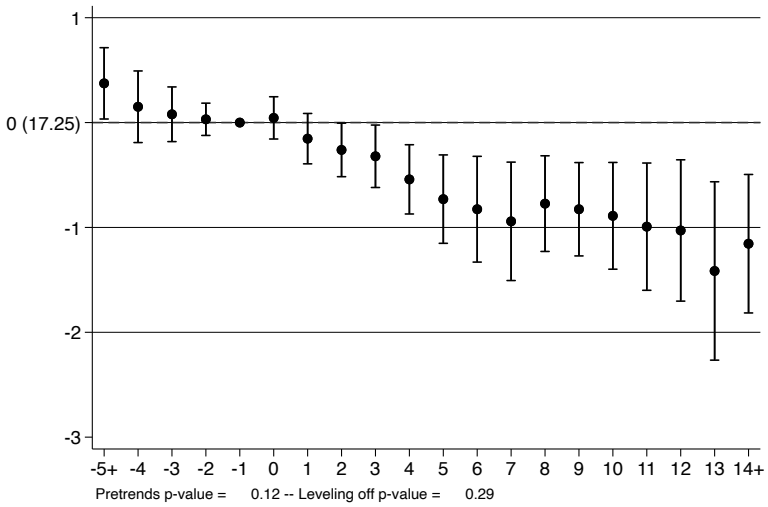

**Notes:** This figure plots the OLS coefficients and their corresponding 95% intervals from an event study on our treatment variable on fish landings. Panel A provides an event study figure regressing log(landings) of a fish population on an indicator variable that is 1 when the treatment is above median of the maximum of property strength. This corresponds to a fish population being treated when approximately 40% of its fishing are landed by nations with using an ITQ. Panel B provides identical analysis, but restricted to the sample of fish that are groundfish. This model employs fish population and year fixed effects and standard errors are clustered at the fish population level.

Figure 5: **Effects of Quota Markets on Annual Landing Value**

**Panel A: All Fish**



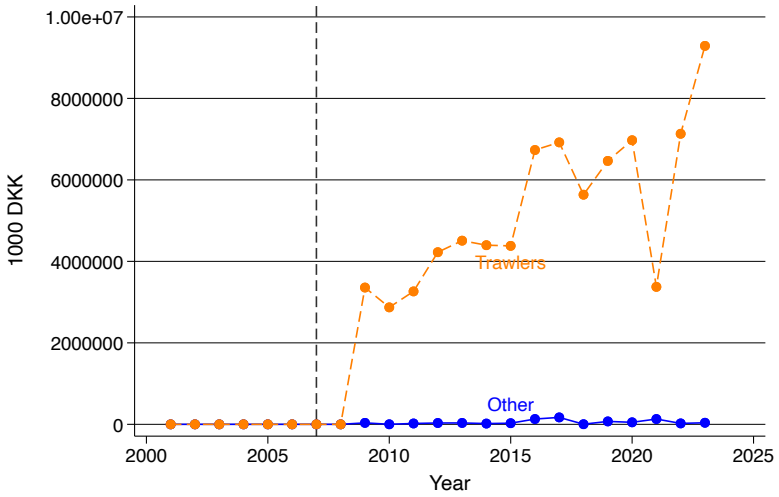

**Panel B: Groundfish**



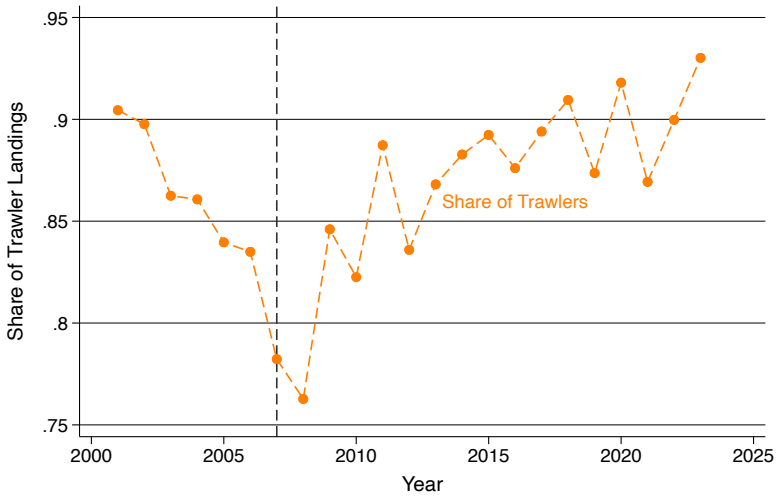

**Notes:** This figure plots the OLS coefficients and their corresponding 95% intervals from an event study on our treatment variable on fish landings value. Panel A provides an event study figure regressing log(landings) of a fish population on an indicator variable that is 1 when the treatment is above median of the maximum of property strength. This corresponds to a fish population being treated when approximately 40% of its fishing are landed by nations with using an ITQ. Panel B provides identical analysis, but restricted to the sample of fish that are groundfish. This model employs fish population and year fixed effects and standard errors are clustered at the fish population level.

Figure 6: **Denmark's Production by Share of Trawler**

**Panel A: Annual Vessel Quota Transfers**



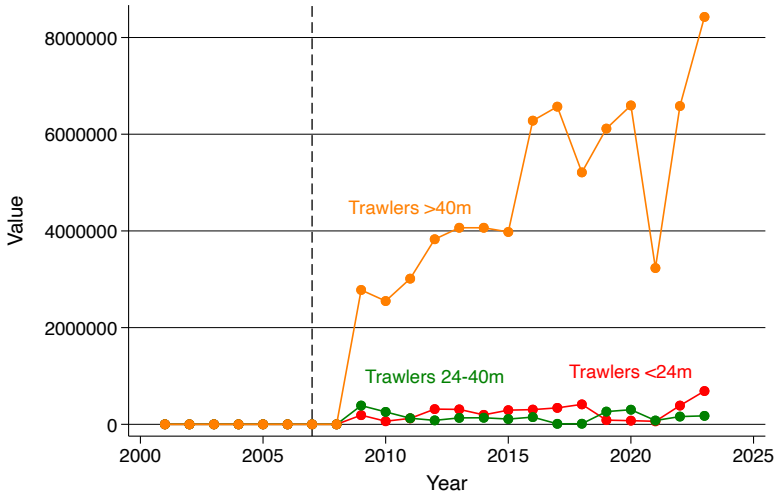

**Panel B: Share of Total Tonnage of Fish Landings**



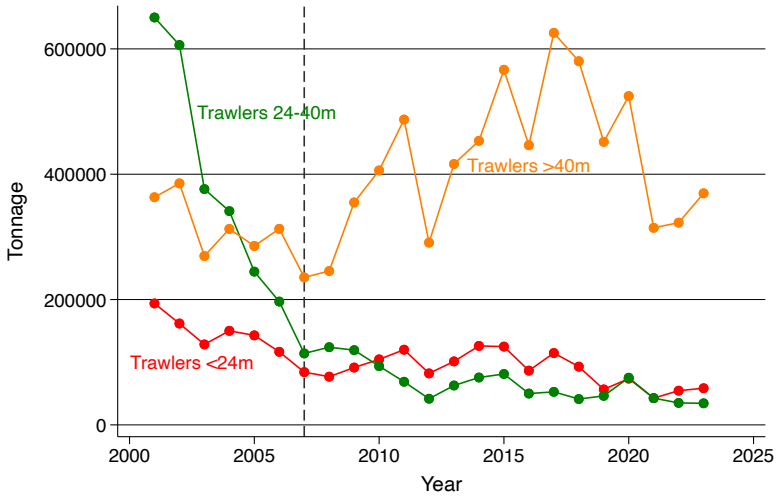

**Notes:** Panel A plots the total value of transferred annual quotas in thousands Danish Krones for trawlers and non-trawlers. For years between 2007 and 2009, there is a lack of comprehensive data on ITQ trading and so these years are marked as zero. Panel B plots the share of annual landings that were caught using trawling. A dashed line in 2007 marks the year the ITQ was rolled out for the Danish fishing fleet. Data is provided by University of Copenhagen and the Danish government.

Figure 7: **Denmark's Trawler Production by Size**

**Panel A: Vessel Quota Transfers**




**Panel B: Landings by Tonnage**




**Notes:** Panel A plots the total value of transferred annual quotas in thousands Danish Krones for trawlers across different sizes. For years between 2007 and 2009, there is a lack of comprehensive data on ITQ trading and so these years are marked as zero. Panel B plots the total tonnage of landings across different vessel sizes of trawlers. A dashed line in 2007 marks the year the ITQ was rolled out for the Danish fishing fleet. Data is provided by University of Copenhagen and the Danish government.

# 10 Tables

Table 1: **Catch Shares**

**Panel A: Example of Catch Share in 1982**

| **Stock** | | |
|---|---|---|
| Species | Cod | |
| Geographical regions | North Sea and Norwegian Sea | |
| ICES Sea Zones | 27.2.A, 27.4.A, 27.4.B, 27.4.C | |
| **Member State** | **1982 Quota (tonnes)** | **Catch Share %** |
| Belgium | 8,700 | 3.47 |
| Denmark | 57,000 | 22.68 |
| West Germany | 31,700 | 12.63 |
| France | 10,750 | 4.28 |
| Ireland | 0 | 0 |
| Netherlands | 28,250 | 11.25 |
| United Kingdom | 114,700 | 45.67 |
| **Total** | **251,100** | **100** |

**Panel B: Summary Statistics of Catch Shares**

| Country | $n$ | Quota (1/0) | Conditional ave. | SD | 99th pct. |
|---|---|---|---|---|---|
| Belgium | 62 | 0.55 | 0.07 | 0.11 | 0.63 |
| Germany | 62 | 0.56 | 0.17 | 0.18 | 0.94 |
| Denmark | 62 | 0.53 | 0.50 | 0.36 | 0.99 |
| Spain | 62 | 0.16 | 0.36 | 0.19 | 0.92 |
| France | 62 | 0.76 | 0.24 | 0.23 | 0.76 |
| United Kingdom | 62 | 0.79 | 0.28 | 0.22 | 0.78 |
| Ireland | 62 | 0.50 | 0.26 | 0.21 | 0.83 |
| Netherlands | 62 | 0.50 | 0.11 | 0.12 | 0.75 |
| Portugal | 62 | 0.13 | 0.24 | 0.14 | 1.00 |

**Notes:** Panel A presents an excerpt from the Council Regulation (EEC) No 172/83 of 25 January 1983 for cod in the North Sea. Panel B presents a summary table of the national catch share variable for the 62 observed fish populations. Column 1 presents an indicator variable that is equal to 1 when the nation has a fishing presence in that population. Column 2 presents a nation's average fishing share conditional on having a fishing presence. Column 3 presents the standard deviation of a nation's fishing shares across the 40 populations. Column 4 presents the 99th percentile of nation's fishing shares. This data is collected both from Council Regulation (EEC) No 172/83 and from quota data as reported from the New Economic Foundation.

Table 2: **Summary Statistics**

**Panel A: Fish Population × Year (1983–2024)**

| | N | Mean | Median | SD | p10 | p90 |
|---|---|---|---|---|---|---|
| Transferability Exposure | 2595 | 0.23 | 0.10 | 0.28 | 0.00 | 0.74 |
| Log(Stock) | 2091 | 10.50 | 10.47 | 1.92 | 8.11 | 12.86 |
| Log(Landings) | 2595 | 8.95 | 8.95 | 2.19 | 6.04 | 11.78 |
| Groundfish (1/0) | 2595 | 0.80 | 1.00 | 0.40 | 0.00 | 1.00 |
| Top Five (1/0) | 2595 | 0.19 | 0.00 | 0.39 | 0.00 | 1.00 |
| Top Ten (1/0) | 2595 | 0.31 | 0.00 | 0.46 | 0.00 | 1.00 |
| Core Fish (1/0) | 2595 | 0.67 | 1.00 | 0.47 | 0.00 | 1.00 |

**Panel B: Nation × Fish × Sea Zone**

| | N | Mean | Median | SD | p10 | p90 |
|---|---|---|---|---|---|---|
| ITQ Adoption | 3222 | 0.10 | 0.00 | 0.30 | 0.00 | 1.00 |
| Transferability (MRAG) | 3222 | 0.44 | 0.25 | 0.25 | 0.25 | 0.88 |
| $\Delta_{13-21}$ All | 3222 | -0.15 | -0.13 | 1.76 | -2.18 | 1.90 |
| $\Delta_{13-21}$ Large | 2163 | -0.15 | -0.12 | 1.81 | -2.24 | 1.88 |
| $\Delta_{13-21}$ Trawler | 2563 | -0.23 | -0.20 | 1.77 | -2.33 | 1.79 |
| $\Delta_{13-21}$ Large Trawler | 1972 | -0.25 | -0.16 | 1.88 | -2.47 | 1.85 |
| $\Delta_{13-21}$ Small | 1358 | -0.25 | -0.11 | 1.79 | -2.46 | 1.84 |
| $\Delta_{13-21}$ Non-Trawler | 1765 | -0.08 | -0.03 | 1.78 | -2.29 | 2.09 |
| $\Delta_{13-21}$ Small Non-Trawler | 1259 | -0.17 | -0.07 | 1.73 | -2.38 | 1.81 |

**Panel C: Nation × Year (1990–2024)**

| | N | Mean | Median | SD | p10 | p90 |
|---|---|---|---|---|---|---|
| ITQ Adoption | 288 | 0.27 | 0.00 | 0.45 | 0.00 | 1.00 |
| Log(1 + Total Vessel Exits) | 288 | 5.73 | 5.69 | 0.95 | 4.66 | 7.15 |
| Log (1 + Total Vessel Exports) | 288 | 2.15 | 2.20 | 0.99 | 0.69 | 3.40 |
| Log(1 + Trawler Vessel Exits) | 288 | 3.97 | 4.13 | 1.24 | 2.40 | 5.19 |
| Log(1 + Trawler Vessel Exports) | 288 | 1.64 | 1.87 | 1.03 | 0.00 | 2.94 |
| Log(1 + Large Vessel Exits) | 288 | 3.05 | 3.18 | 0.90 | 1.95 | 4.01 |
| Log(1 + Large Vessel Exports) | 288 | 1.48 | 1.39 | 0.92 | 0.00 | 2.64 |

**Notes:** This table provides summary statistics for the samples used in our main analysis. Panel A presents the sample of fish populations from 1983 to 2024 with biomass and landing data from ICES. Panel B presents cross sectional data on landings reported by the Common Fishery STECF. Panel C presents annual data from the European Vessel Register from 1990 to 2024.

Table 3: **Effects of Quota Markets on Fish Population and Harvest**

**Panel A: Baseline Effects**

| | Log(Biomass) | | | Log(Landing) | | |
|---|---|---|---|---|---|---|
| | All Fish | Groundfish | Other | All Fish | Groundfish | Other |
| | (1) | (2) | (3) | (4) | (5) | (6) |
| Transferability Exposure | -0.351* | -0.472** | -0.051 | -0.492** | -0.720** | -0.294 |
| | (0.182) | (0.217) | (0.344) | (0.224) | (0.285) | (0.319) |
| Year FE | Yes | Yes | Yes | Yes | Yes | Yes |
| Fish Population FE | Yes | Yes | Yes | Yes | Yes | Yes |
| Observations | 2,591 | 1,857 | 734 | 2,595 | 2,071 | 524 |
| $R^2$ | 0.945 | 0.922 | 0.936 | 0.911 | 0.894 | 0.950 |

**Panel B: Effects by Fish Position in Food Chain**

| | Log(Biomass) | | Log(Landing) | |
|---|---|---|---|---|
| | High Trophic | Low Trophic | High Trophic | Low Trophic |
| | (1) | (2) | (3) | (4) |
| Transferability Exposure | -0.587* | -0.286 | -1.390*** | -0.072 |
| | (0.338) | (0.224) | (0.489) | (0.203) |
| Year FE | Yes | Yes | Yes | Yes |
| Fish Population FE | Yes | Yes | Yes | Yes |
| Observations | 1,400 | 1,191 | 1,547 | 1,048 |
| $R^2$ | 0.935 | 0.957 | 0.895 | 0.942 |

**Notes:** This table reports the coefficients of the fish-population-level OLS regressions using our shift-share instrument. Panel A presents the effects of transferability exposure on the logarithm of fish biomass and annual landings. Column 1 reports the coefficient from the regression of transferability exposure on log(biomass) for all observed species of fish. Column 2 repeats this analysis, but restricts the sample to groundfish. Column 3 restricts the analysis to non-groundfish. Columns 4–6 repeat this analysis, but using log(tonnage of landings) as its dependent variable. Panel B runs the same specification, but splits the sample between high trophic and low trophic fish species (high trophic = above the median trophic level). All data on trophic levels is from FishBase. Standard errors are clustered at the fish-population level and are in parentheses. *, **, and *** denote 10%, 5%, and 1% significance.

Table 4: **Commercial Effects of Quota Markets on Fish**

**Panel A: Effects on Commercial Value**

| | Log(Value of Stock) | | | Log(Value of Landing) | | |
|---|---|---|---|---|---|---|
| | All Fish | Groundfish | Other | All Fish | Groundfish | Other |
| | (1) | (2) | (3) | (4) | (5) | (6) |
| Transferability Exposure | 0.568 | -0.523* | 1.165** | -0.587 | -0.936** | 0.339 |
| | (0.363) | (0.299) | (0.439) | (0.363) | (0.367) | (0.893) |
| Year FE | Yes | Yes | Yes | Yes | Yes | Yes |
| Fish Population FE | Yes | Yes | Yes | Yes | Yes | Yes |
| Observations | 1,710 | 1,161 | 549 | 1,513 | 1,212 | 301 |
| $R^2$ | 0.881 | 0.911 | 0.894 | 0.876 | 0.919 | 0.838 |

**Panel B: Effects on Commercially Important Fish Populations**

| | Log(Biomass) | | | | | |
|---|---|---|---|---|---|---|
| | Top Five | Top Ten | Core | Non-Five | Non-Ten | Non-Core |
| | (1) | (2) | (3) | (4) | (5) | (6) |
| Transferability Exposure | 0.262 | 0.116 | -0.221 | -0.485** | -0.538** | -0.699* |
| | (0.478) | (0.294) | (0.201) | (0.191) | (0.217) | (0.382) |
| Year FE | Yes | Yes | Yes | Yes | Yes | Yes |
| Fish Population FE | Yes | Yes | Yes | Yes | Yes | Yes |
| Observations | 708 | 985 | 1,841 | 1,883 | 1,606 | 750 |
| $R^2$ | 0.953 | 0.966 | 0.945 | 0.941 | 0.933 | 0.940 |

**Notes:** This table reports the coefficients of the fish-population-level OLS regressions using our shift-share instrument. Panel A presents the effects of transferability exposure on the logarithm of fish stock value and annual landings value. Column 1 reports the coefficient from the regression of transferability exposure on log(value of stock) for all observed species of fish. Column 2 repeats this analysis, but restricts the sample to groundfish. Column 3 restricts the analysis to non-groundfish. Columns 4–6 repeat this analysis, but using log(value of landings) as its dependent variable. Panel B runs the same specification, but splits the sample using the commercial importance of each fish species. The column for Top-Five represents the five fish with the highest Euros landed in 2015. Top-Ten represents the ten fish with the highest Euros landed in 2015. Core fish represents the fish that were regulated in the original 1983 treaty. Standard errors are clustered at the fish-population level and are in parentheses. *, **, and *** denote 10%, 5%, and 1% significance.

Table 5: **Effects on Fishing Production**

**Panel A: Annual Landings by Large Trawlers**

| | $\Delta_{13-21}$ Landings | | | | | |
|---|---|---|---|---|---|---|
| | Large | | Trawler | | Large Trawlers | |
| | (1) | (2) | (3) | (4) | (5) | (6) |
| ITQ Adoption | 0.295* | | 0.263** | | 0.375** | |
| | (0.165) | | (0.133) | | (0.158) | |
| Transferability | | 0.324 | | 0.121 | | 0.644** |
| | | (0.270) | | (0.224) | | (0.286) |
| Fish Family FE | Yes | Yes | Yes | Yes | Yes | Yes |
| Sea Zone FE | Yes | Yes | Yes | Yes | Yes | Yes |
| Observations | 1,289 | 1,289 | 1,418 | 1,418 | 1,186 | 1,186 |
| $R^2$ | 0.130 | 0.129 | 0.135 | 0.133 | 0.163 | 0.164 |

**Panel B: Annual Landings by Small Non-Trawlers**

| | $\Delta_{13-21}$ Landings | | | | | |
|---|---|---|---|---|---|---|
| | Small | | Non-Trawler | | Small Non-Trawlers | |
| | (1) | (2) | (3) | (4) | (5) | (6) |
| ITQ Adoption | -0.707** | | -0.494 | | -0.781** | |
| | (0.308) | | (0.420) | | (0.308) | |
| Transferability | | -1.412* | | -1.144** | | -1.768** |
| | | (0.729) | | (0.472) | | (0.753) |
| Fish Family FE | Yes | Yes | Yes | Yes | Yes | Yes |
| Sea Zone FE | Yes | Yes | Yes | Yes | Yes | Yes |
| Observations | 794 | 794 | 1,032 | 1,032 | 768 | 768 |
| $R^2$ | 0.182 | 0.183 | 0.152 | 0.159 | 0.161 | 0.165 |

**Notes:** This table reports the coefficients of the nation–fish–sea-zone OLS regression, in which the dependent variable is the change in landings from 2013 to 2021. The two explanatory variables are an indicator variable that equals 1 if a nation implements a market system of ITQ, and a continuous measurement of market transferability rights conducted by the Marine Research Assessment Group. In Panel A, Columns 1–2 present this analysis for the sample of vessels that are over 24 meters in length; Columns 3–4 restrict the sample to vessels that use trawling; Columns 5–6 restrict to trawlers that are over 24 meters. In Panel B, Columns 1–2 restrict the sample to vessels under 12 meters; 3–4 for non-trawlers; and 5–6 for non-trawlers under 12 meters. The sample of landings is restricted to observations with landings greater than or equal to 10 tons in 2013. Standard errors are clustered at the sea-zone level and are in parentheses. *, **, and *** denote 10%, 5%, and 1% significance.

Table 6: **Vessel Level Effects**

**Panel A: Effects on Fishing Vessel Retirement**

| | Vessel Exit | | | | | |
|---|---|---|---|---|---|---|
| | All | | Trawler | | Large | |
| | (1) | (2) | (3) | (4) | (5) | (6) |
| ITQ Adoption | 1.450 | 1.068 | 4.131*** | 2.897*** | 1.806*** | 1.693*** |
| | (1.217) | (0.150) | (5.956) | (5.421) | (3.258) | (3.904) |
| ITQ Adoption × Year ≥ 2013 | | 2.478** | | 3.139*** | | 1.984 |
| | | (2.469) | | (2.666) | | (1.486) |
| Country Strata | Yes | Yes | Yes | Yes | Yes | Yes |
| Observations | 1632533 | 1632533 | 297615 | 297615 | 97736 | 97736 |

**Panel B: Effects on National Fishing Fleet Retirements**

| | All | | Trawler | | Large | |
|---|---|---|---|---|---|---|
| | Exit | Export | Exit | Export | Exit | Export |
| | (1) | (2) | (3) | (4) | (5) | (6) |
| ITQ Adoption | 0.114 | 0.770* | 1.290* | 1.017 | 0.864 | 0.824** |
| | (0.540) | (0.391) | (0.639) | (0.545) | (0.464) | (0.314) |
| Year FE | Yes | Yes | Yes | Yes | Yes | Yes |
| Country FE | Yes | Yes | Yes | Yes | Yes | Yes |
| Observations | 288 | 288 | 288 | 288 | 288 | 288 |
| $R^2$ | 0.620 | 0.654 | 0.648 | 0.674 | 0.534 | 0.676 |

**Notes:** Panel A reports the hazard ratios from a Cox proportional hazard model of the survival of a European fishing vessels starting in 1990. Using the European Vessel Register, we define a vessel as exited if it was scrapped, retired, exported, or destroyed. Column 1-2 provides the ratio for all vessels. Column 3-4 provide the ratio when restricted to the universe of trawling vessels. Columns 5-6 provide the ratio when restricted to the universe of vessels over 24 meters in length. Panel B presents the coefficients of the OLS regressions, in which the dependent variable is either vessel exit or vessel export. Column 1-2 presents the coefficient for all vessels. Column 3-4 presents the coefficient for the universe of trawlers. Column 5-6 presents the coefficient for the universe of vessels over 24 meters. Standard errors are clustered at the national level and are in parentheses. *, **, and *** denote 10%, 5%, and 1% significance.

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# Appendix A Identification Analysis

In this section, we provide more formal analysis of the validity of our main explanatory instrument. As a reminder, the shares from our shift-share instrument are primarily from a 1983 European agreement, which agreed to freeze shares at their average level from 1973 to 1978. Our central concern is that these historical shares are correlated with unobserved characteristics that effect future trends in fish population.

## A.1 Correlates of Shares

Following Goldsmith-Pinkham et al. (2020) and Borusyak et al. (2025), we begin by exploring what observable characteristics may explain the cross-sectional variation of our historical fishing shares. Because comprehensive ecological data is relatively rare for the years before our analysis, we turn our attention to geographic features such as distance from coastline. Table A1 examines how geographic distance shapes these historical fishing shares.

This analysis reveals that geography and fish fixed effects explains a considerable amount of the cross-sectional variation of our initial shares, especially in the three nations that adopt ITQ. For instance, in Panel A, Denmark and the United Kingdom both have a coefficient of determination that is 0.37 and the Netherlands has a coefficient of 0.213. We find that across the board, nations have higher catch shares in fish populations that overlap with their exclusive economic zone. Controlling for other variables, we also find that being the closest nation to a sea zone tends to predict higher fishing shares. Although, this relationship is weaker for countries that share their closest sea zones with many other nations such as the Netherlands and Belgium.

INSERT TABLE A1

In Table A2, we show these observation remain relatively robust when restricting to categories of different fish types. While this analysis cannot rule out unobservable confouding variables, it is reassuring that so much variation is explained by simple geographic features which are captured by controls in our main specification.

INSERT TABLE A2

### A.2 Unpooled Analysis

As a next test, we provide unpooled versions of our main analysis. Because our shift-share estimate can be interpreted as pooled version of multiple national share-instruments, it is useful to understand which of these results is driving our main results Goldsmith-Pinkham et al. (2020). In our case, we restrict our analysis to the shares of nations that have variation in their fishery market structure–Netherlands, United Kingdom, and Denmark.

In Table A3 we provide this analysis for each nation by rerunning our main specification with at 'one-at-a-time' share instrument estimates. This table reveals that the United Kingdom's adoption of ITQ is a primary driver of our findings. Table A4, which presents the unpooled regressions by nation pairs, further confirms the importance of the United Kingdom to our analysis.

The fact the United Kingdom is highly influential is not entirely surprising. In terms of scale, the United Kingdom is one of the largest producers of fish with its industry landing 624 thousand tones in 2013. Table C2 shows, conditional on a fishing presence, the United Kingdom fishes 27%, while the Netherlands fishes 11%. Moreover, the United Kingdom fishes over a very large swathe of the Atlantic. Of the 54 fish populations in the A3, the United Kingdom has a non-zero presence in 35. In contrast, Denmark has a presence in only 22 populations. In the following section, we provide more specific evidence that our

identifying assumptions are valid in the case of the United Kingdom.

INSERT TABLE A3

### A.3 United Kingdom Specific Tests

As further analysis, we examine how factors such as Brexit and the United Kingdom's unique quasi-system of ITQ may threaten our analysis.

One reasonable concern is that Britain's 2016 withdraw from the European Union and renegotiation of its quota allocation from the European Common Fishery may weaken our instrument in later years. To test this concern in our data, we first provide evidence that the UK's fishing shares are relatively unchanged in the years that follow Brexit. Figure A1 shows that 1983 fishing shares are highly predictive of both 2010 and 2020 shares. Moreover, the distribution of differences looks similar pre and post-brexit. This data also matches with institutional announcements promising to preserve the fishing status quo during negotiation.

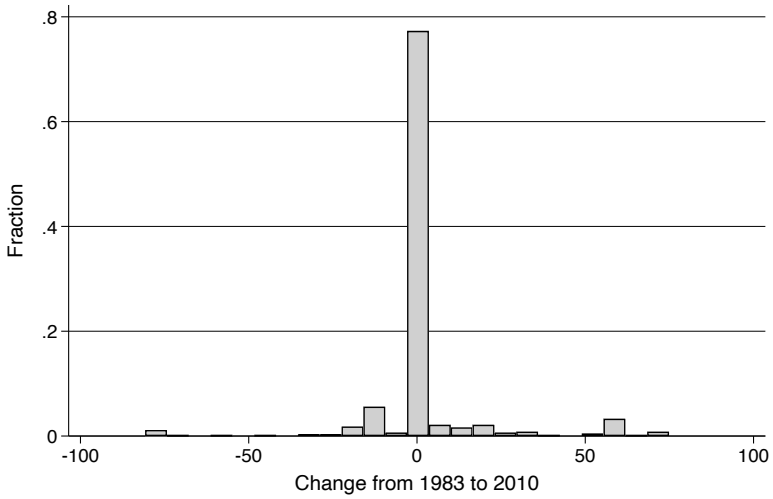
INSERT FIGURE A1

As a further robustness check, Figure A2 provides an event study figure restricted only to the UK's adoption of ITQ. Reassuringly, although under powered, the trends looks highly similar to our main analysis with limited pre-trends and gradually increasing negative effects over the following 15 years.

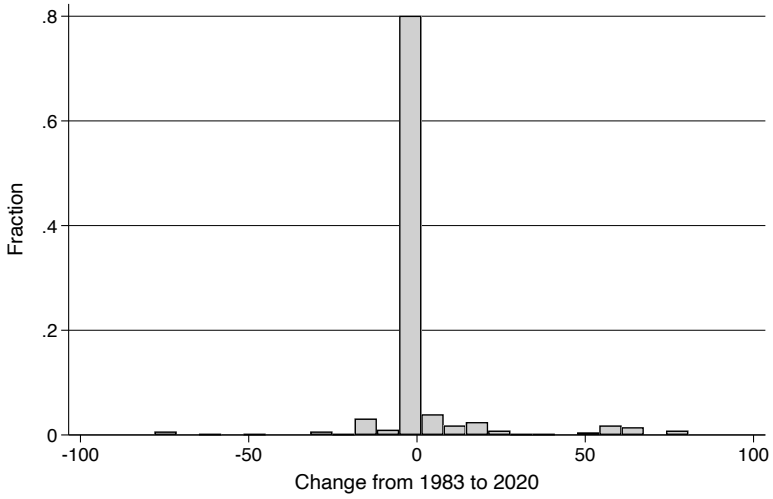
INSERT FIGURE A2

Figure A1: **Brexit and Fishing Shares**

**Panel A: Pre-Brexit**



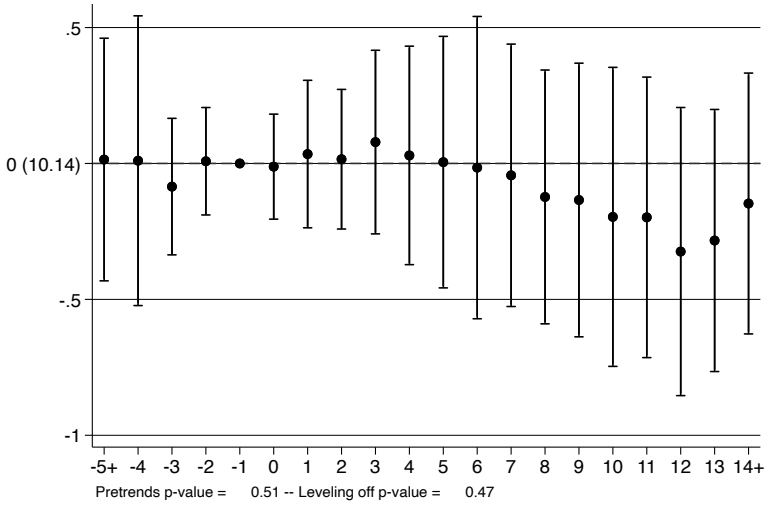

**Panel B: Post-Brexit**



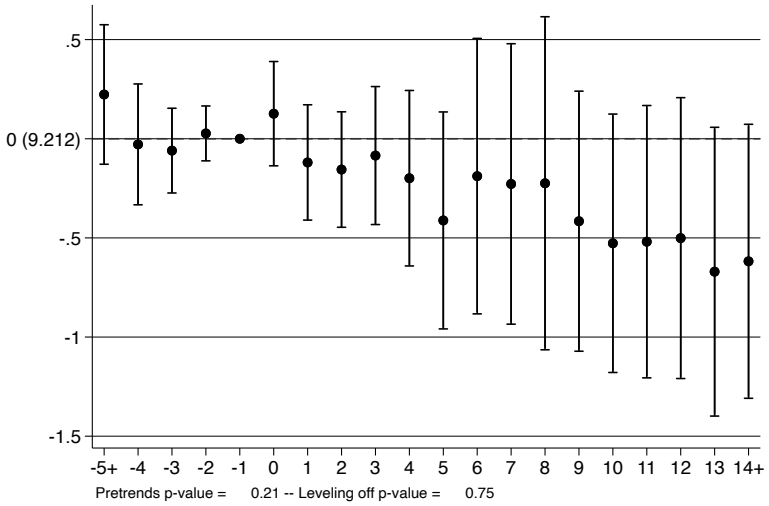

**Notes:** This figure plots the difference in the UK's 1983 shares before and after Brexit. Panel A presents a histogram of the difference from 1983 to 2010 and Panel B presents the difference from 1983 to 2020.

Figure A2: **United Kingdom- Event Study**

**Panel A: Groundfish Population**




**Panel B: Post-Groundfish Landings**




**Notes:** This figure plots the OLS coefficients and their corresponding 95% intervals from an event study on UK ITQ Exposure on fish population. Panel A provides an event study figure regressing log(biomass) of a fish population on an indicator variable that is 1 when the treatment is above median of the maximum of property strength from the UK. Panel B provides identical analysis, but for annual landings. This figure restricts the sample to those of groundfish. This model employs fish population and year fixed effects and standard errors are clustered at the fish population level.

Table A1: **Correlates of Shares**

**Panel A: 1983 Shares**

| | 1983 National Shares | | | | | | | |
|---|---|---|---|---|---|---|---|---|
| | All | BEL | DEU | DNK | FRA | UK | IRL | NLD |
| | (1) | (2) | (3) | (4) | (5) | (6) | (7) | (8) |
| Log(Distance) | -0.205 | -0.934 | 1.939 | 2.558** | -0.004 | -0.665** | -0.659** | -0.393* |
| | (0.216) | (0.542) | (1.894) | (0.965) | (1.003) | (0.265) | (0.252) | (0.212) |
| Closest Nation (1/0) | 5.998*** | -5.720 | 12.962 | 27.951*** | 14.156* | 2.582 | -0.544 | -9.681 |
| | (1.664) | (3.788) | (12.769) | (7.807) | (7.053) | (2.390) | (0.841) | (9.992) |
| Exclusive Economic Zone (%) | 0.151*** | | 0.164 | 0.416*** | 0.064 | 0.050** | 0.156* | 0.332 |
| | (0.033) | | (0.099) | (0.121) | (0.053) | (0.022) | (0.075) | (0.303) |
| Fish FE | Yes | Yes | Yes | Yes | Yes | Yes | Yes | Yes |
| Observations | 4,256 | 608 | 608 | 608 | 608 | 608 | 608 | 608 |
| $R^2$ | 0.182 | 0.176 | 0.143 | 0.373 | 0.453 | 0.376 | 0.293 | 0.213 |

**Panel B: 2001 Shares**

| | 2001 National Shares | | | | | | | |
|---|---|---|---|---|---|---|---|---|
| | All | BEL | DEU | DNK | FRA | UK | IRL | NLD |
| | (1) | (2) | (3) | (4) | (5) | (6) | (7) | (8) |
| Log(Distance) | -0.541** | -1.007** | 2.504 | -3.768** | -2.726*** | -0.383 | -0.237 | -0.429* |
| | (0.204) | (0.447) | (1.528) | (1.437) | (0.817) | (0.525) | (0.233) | (0.226) |
| Closest Nation (1/0) | 4.404*** | -6.626* | 17.311 | 1.977 | -19.941*** | 8.521*** | 5.651* | -17.219* |
| | (1.261) | (3.291) | (10.186) | (2.180) | (5.954) | (1.881) | (2.795) | (9.216) |
| Exclusive Economic Zone (%) | 0.087*** | | 0.075 | -0.386** | 0.226*** | -0.009 | 0.064 | 0.510* |
| | (0.028) | | (0.113) | (0.170) | (0.063) | (0.046) | (0.054) | (0.294) |
| Fish FE | Yes | Yes | Yes | Yes | Yes | Yes | Yes | Yes |
| Observations | 9,126 | 1,014 | 1,014 | 1,014 | 1,014 | 1,014 | 1,014 | 1,014 |
| $R^2$ | 0.125 | 0.166 | 0.196 | 0.180 | 0.464 | 0.277 | 0.344 | 0.283 |

**Notes:** Shares are multiplied by 100 for visual clarity. Because multiple nations may be equally close to the same sea zone (For example, France and UK are equally close to the English Channel), the closest nation variable is not always unique. As Spain and Portugal shares are not included in the original 1983 documents, we exclude them in our nation by nation breakdown of 2001 shares. However, their shares are included in the aggregate analysis in column A in Panel B. Standard errors in both panels are clustered at the fish species level and are reported in the parentheses. *, **, and *** represent statistical significance at the 10%, 5%, and 1% confidence levels, respectively.

Table A2: **Correlates of Shares By Fish Category**

**Panel A: 1983 Shares**

| | Groundfish | | | Pelagic | | |
|---|---|---|---|---|---|---|
| | DNK | GBR | NLD | DNK | GBR | NLD |
| | (1) | (2) | (3) | (4) | (5) | (6) |
| Log(Distance) | 1.458 | -0.524** | -0.152 | 5.639** | -1.060 | -1.069 |
| | (1.026) | (0.242) | (0.120) | (1.782) | (0.789) | (0.702) |
| Closest Nation (1/0) | 24.350** | 4.584** | -20.627* | 38.033* | -3.023 | 20.969 |
| | (9.277) | (2.113) | (11.428) | (15.132) | (6.861) | (14.262) |
| Exclusive Economic Zone (%) | 0.311** | 0.048 | 0.651* | 0.707** | 0.057 | -0.561 |
| | (0.139) | (0.028) | (0.359) | (0.222) | (0.037) | (0.353) |
| Fish FE | Yes | Yes | Yes | Yes | Yes | Yes |
| Observations | 448 | 448 | 448 | 160 | 160 | 160 |
| $R^2$ | 0.400 | 0.352 | 0.221 | 0.343 | 0.427 | 0.312 |

**Panel B: 2001 Shares**

| | Groundfish | | | Pelagic | | |
|---|---|---|---|---|---|---|
| | DNK | GBR | NLD | DNK | GBR | NLD |
| | (1) | (2) | (3) | (4) | (5) | (6) |
| Log(Distance) | -1.732** | -0.456 | -0.366** | -9.297* | -0.185 | -0.601 |
| | (0.706) | (0.632) | (0.166) | (4.590) | (1.004) | (0.752) |
| Closest Nation (1/0) | 1.357 | 11.019*** | -28.463** | 3.661 | 1.738 | 13.302 |
| | (1.323) | (2.130) | (10.979) | (7.691) | (2.720) | (11.038) |
| Exclusive Economic Zone (%) | -0.083 | -0.042 | 0.869** | -1.206** | 0.083 | -0.465 |
| | (0.094) | (0.045) | (0.344) | (0.472) | (0.120) | (0.401) |
| Fish FE | Yes | Yes | Yes | Yes | Yes | Yes |
| Observations | 741 | 741 | 741 | 273 | 273 | 273 |
| $R^2$ | 0.199 | 0.251 | 0.205 | 0.227 | 0.387 | 0.517 |

**Notes:** Shares are multiplied by 100 for visual clarity. Because multiple nations may be equally close to the same sea zone (For example, France and UK are equally close to the English Channel), the closest nation variable is not always unique. Standard errors in both panels are clustered at the fish species level and are reported in the parentheses. *, **, and *** represent statistical significance at the 10%, 5%, and 1% confidence levels, respectively.

Table A3: **Unpooled Results by Nation**

## Panel A: Netherlands

| | Log(Biomass) | | | Log(Landing) | | |
|---|---|---|---|---|---|---|
| | All Fish | Groundfish | Other | All Fish | Groundfish | Other |
| | (1) | (2) | (3) | (4) | (5) | (6) |
| ITQ Exposure | -0.006 | -0.004 | -0.018 | -0.002 | 0.002 | 0.777*** |
| | (0.004) | (0.004) | (0.041) | (0.004) | (0.004) | (0.143) |
| Year FE | Yes | Yes | Yes | Yes | Yes | Yes |
| Fish Population FE | Yes | Yes | Yes | Yes | Yes | Yes |
| Observations | 1,731 | 1,330 | 401 | 1,625 | 1,402 | 223 |
| $R^2$ | 0.942 | 0.907 | 0.940 | 0.924 | 0.923 | 0.873 |

## Panel B: United Kingdom

| | Log(Biomass) | | | Log(Landing) | | |
|---|---|---|---|---|---|---|
| | All Fish | Groundfish | Other | All Fish | Groundfish | Other |
| | (1) | (2) | (3) | (4) | (5) | (6) |
| ITQ Exposure | -0.008 | -0.008 | -0.001 | -0.019** | -0.016* | 0.088*** |
| | (0.005) | (0.005) | (0.009) | (0.008) | (0.008) | (0.020) |
| Year FE | Yes | Yes | Yes | Yes | Yes | Yes |
| Fish Population FE | Yes | Yes | Yes | Yes | Yes | Yes |
| Observations | 1,731 | 1,330 | 401 | 1,625 | 1,402 | 223 |
| $R^2$ | 0.943 | 0.910 | 0.940 | 0.932 | 0.929 | 0.856 |

## Panel C: Denmark

| | Log(Biomass) | | | Log(Landing) | | |
|---|---|---|---|---|---|---|
| | All Fish | Groundfish | Other | All Fish | Groundfish | Other |
| | (1) | (2) | (3) | (4) | (5) | (6) |
| ITQ Exposure | -0.000 | 0.001 | -0.003 | 0.002 | -0.003 | -0.002 |
| | (0.003) | (0.003) | (0.005) | (0.006) | (0.005) | (0.008) |
| Year FE | Yes | Yes | Yes | Yes | Yes | Yes |
| Fish Population FE | Yes | Yes | Yes | Yes | Yes | Yes |
| Observations | 1,731 | 1,330 | 401 | 1,625 | 1,402 | 223 |
| $R^2$ | 0.942 | 0.907 | 0.941 | 0.924 | 0.923 | 0.834 |

**Notes:** . Standard errors in all panels are clustered at the fish population level and are reported in the parentheses. *, **, and *** represent statistical significance at the 10%, 5%, and 1% confidence levels, respectively.

Table A4: **Unpooled Results by Nation Pairs**

**Panel A: Netherlands and Denmark**

| | Log(Biomass) | | | Log(Landing) | | |
|---|---|---|---|---|---|---|
| | All Fish | Groundfish | Other | All Fish | Groundfish | Other |
| | (1) | (2) | (3) | (4) | (5) | (6) |
| ITQ Exposure | -0.001 | -0.000 | -0.004 | 0.001 | -0.001 | -0.002 |
| | (0.002) | (0.003) | (0.005) | (0.004) | (0.004) | (0.008) |
| Year FE | Yes | Yes | Yes | Yes | Yes | Yes |
| Fish Population FE | Yes | Yes | Yes | Yes | Yes | Yes |
| Observations | 1,731 | 1,330 | 401 | 1,625 | 1,402 | 223 |
| $R^2$ | 0.942 | 0.907 | 0.941 | 0.924 | 0.923 | 0.833 |

**Panel B: Netherlands and United Kingdom**

| | Log(Biomass) | | | Log(Landing) | | |
|---|---|---|---|---|---|---|
| | All Fish | Groundfish | Other | All Fish | Groundfish | Other |
| | (1) | (2) | (3) | (4) | (5) | (6) |
| ITQ Exposure | -0.008** | -0.008* | -0.002 | -0.016** | -0.012* | 0.086*** |
| | (0.004) | (0.004) | (0.008) | (0.006) | (0.006) | (0.019) |
| Year FE | Yes | Yes | Yes | Yes | Yes | Yes |
| Fish Population FE | Yes | Yes | Yes | Yes | Yes | Yes |
| Observations | 1,731 | 1,330 | 401 | 1,625 | 1,402 | 223 |
| $R^2$ | 0.944 | 0.910 | 0.940 | 0.931 | 0.928 | 0.860 |

**Panel C: United Kingdom and Denmark**

| | Log(Biomass) | | | Log(Landing) | | |
|---|---|---|---|---|---|---|
| | All Fish | Groundfish | Other | All Fish | Groundfish | Other |
| | (1) | (2) | (3) | (4) | (5) | (6) |
| ITQ Exposure | -0.004* | -0.004 | -0.004 | -0.009** | -0.011** | -0.000 |
| | (0.002) | (0.003) | (0.005) | (0.004) | (0.004) | (0.008) |
| Year FE | Yes | Yes | Yes | Yes | Yes | Yes |
| Fish Population FE | Yes | Yes | Yes | Yes | Yes | Yes |
| Observations | 1,731 | 1,330 | 401 | 1,625 | 1,402 | 223 |
| $R^2$ | 0.943 | 0.908 | 0.941 | 0.927 | 0.928 | 0.833 |

**Notes:** . Standard errors in all panels are clustered at the fish population level and are reported in the parentheses. *, **, and *** represent statistical significance at the 10%, 5%, and 1% confidence levels, respectively.

# Appendix B Background on Fish

Table B1: **Fish Summary Statistics From 2015**

**Panel A: Top 10 Fish in 2015 by Value**

| | Sample (1/0) | Euros (millions) | Tonnage (thousands) | Groundfish (1/0) |
|---|---|---|---|---|
| Swordfish | 0 | 193.9 | 34.33 | 0 |
| Sardine | 1 | 249.7 | 193.68 | 0 |
| Sole | 1 | 288.3 | 25.73 | 1 |
| Yellow Fin Tuna | 0 | 311.0 | 147.19 | 0 |
| Norway Lobster | 1 | 327.4 | 49.94 | 0 |
| Greenland Halibut | 1 | 358.8 | 14.22 | 1 |
| Herring | 1 | 416.2 | 755.08 | 0 |
| Cod | 1 | 471.5 | 151.62 | 1 |
| Hake | 1 | 483.1 | 134.68 | 1 |
| Mackerel | 1 | 525.1 | 572.64 | 0 |

**Panel B: Top 10 Fish in 2015 by Tonnage**

| | Sample (1/0) | Euros (millions) | Tonnage (thousands) | Groundfish (1/0) |
|---|---|---|---|---|
| Yellow Fin Tuna | 0 | 311.0 | 147.19 | 0 |
| Cod | 1 | 471.5 | 151.62 | 1 |
| Horse Mackerel | 1 | 92.7 | 159.17 | 0 |
| Skipjack Tuna | 0 | 28.1 | 164.21 | 0 |
| Sardine | 1 | 249.7 | 193.68 | 0 |
| Sand-eel | 1 | 51.4 | 222.32 | 0 |
| Blue Whiting | 1 | 130.8 | 223.51 | 0 |
| Sprat | 1 | 135.0 | 534.59 | 0 |
| Mackerel | 1 | 525.1 | 572.64 | 0 |
| Herring | 1 | 416.2 | 755.08 | 0 |

**Notes:** This table presents basic summary statistics for the most caught fish in the Northeast Atlantic in 2015. Column 1 presents whether the fish is used in our main analysis. For economics and ecological reasons, the fishing of swordfish and tuna is regulated differently than other fish and so is excluded. Column 2 presents the value of landings in units of million Euros. Column 3 presents weight of landings in units of thousands tons. Column 4 presents an indicator variable that is 1 when the fish is a groundfish. Panel A presents the top ten fish by value in ascending order and Panel B shows the top ten by landing. The source of this data is from the STECF.

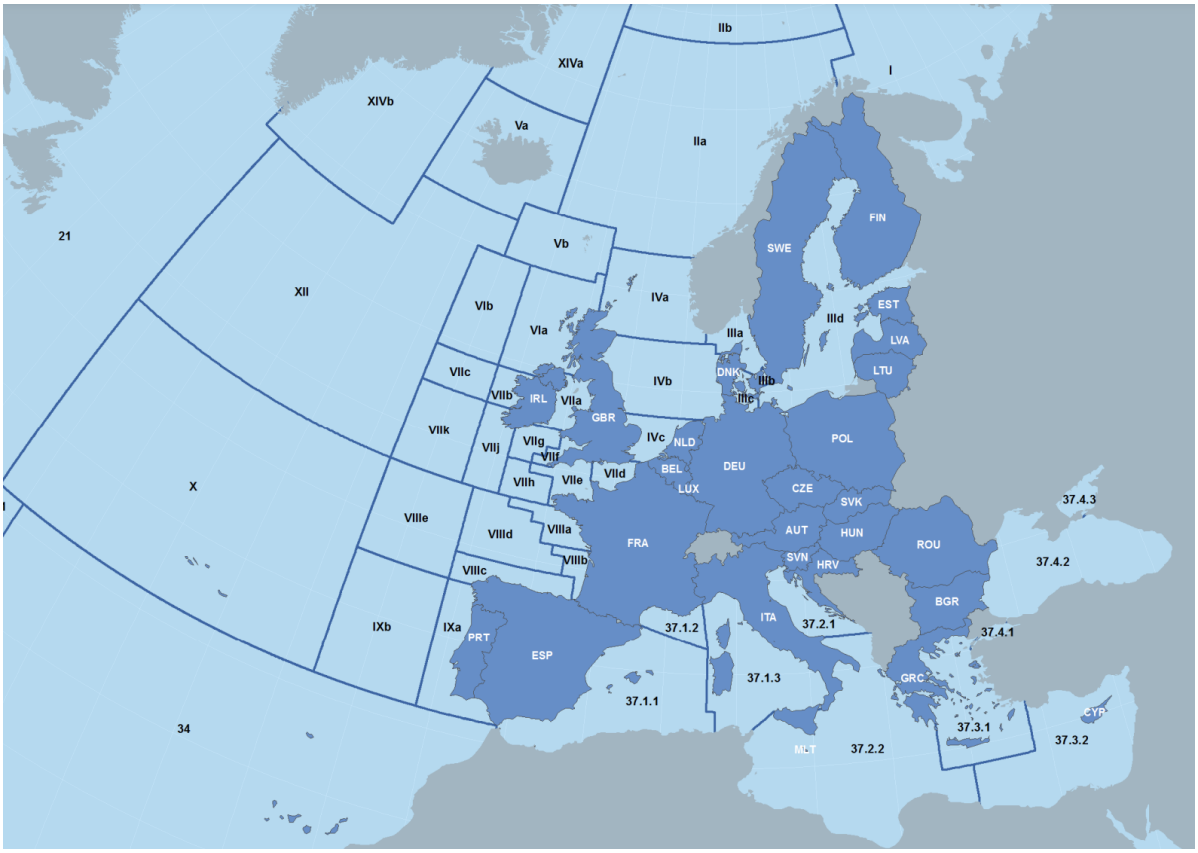
Table B2: **Fish Summary Statistics from 1983**

| | 1983 Quota Tonnage (thousands) | Groundfish (1/0) | Gadidae (1/0) |
|---|---|---|---|
| Sand-eel | 0.00 | 0 | 0 |
| Capelin | 0.00 | 0 | 0 |
| Witch | 0.41 | 1 | 0 |
| American Plaice | 0.55 | 1 | 0 |
| Halibut | 1.22 | 1 | 0 |
| Salmon | 2.17 | 0 | 0 |
| Anchovy | 3.00 | 0 | 0 |
| Catfish | 6.00 | 1 | 0 |
| Greenland Halibut | 10.45 | 1 | 0 |
| Hake | 30.30 | 1 | 0 |
| Shrimps | 31.70 | 0 | 0 |
| Sole | 32.31 | 1 | 0 |
| Redfish | 54.50 | 1 | 0 |
| Saithe | 101.76 | 1 | 1 |
| Plaice | 159.41 | 1 | 0 |
| Haddock | 189.77 | 1 | 1 |
| Whiting | 201.21 | 1 | 1 |
| Herring | 219.40 | 0 | 0 |
| Horse Mackerel | 249.00 | 0 | 0 |
| Norway Pout | 321.00 | 1 | 1 |
| Mackerel | 375.00 | 0 | 0 |
| Sprat | 460.00 | 0 | 0 |
| Blue Whiting | 500.00 | 0 | 1 |
| Cod | 529.87 | 1 | 1 |

**Notes:** This table presents basic summary statistics for the fish that were regulated under the Council Regulation (EEC) No 172/83 of 25 January 1983. Column 1 presents the total allocated quotas in thousands of tons in 1983. It should be noted that although sand-eel and capelin were regulated under the 1983 treaty, the total allowable catch that year was set to zero. Column 2 indicates whether this fish is a groundfish and Column 3 whether the fish is a gadidae.

# Appendix C Additional Tables

Figure C1: **FAO Sea Zone Map**



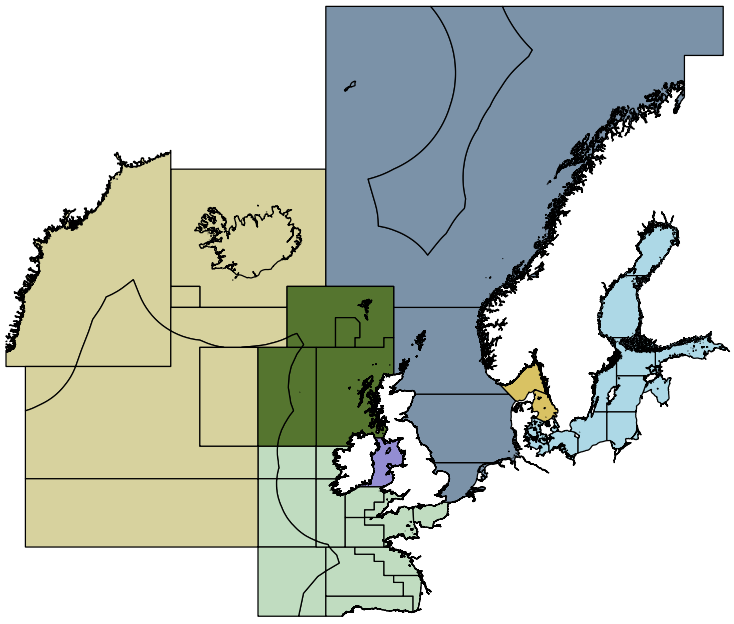

**Notes:** This figure details the Food and Agriculture Organization (FAO) major areas around Europe. Our analysis focuses on major area 27 and its underlying sea zones. These sea zones are labeled by their corresponding roman numerals on the map.

Figure C2: **Example of Cod Quota Ranges**

**Panel A: Example of 1982 Quota for North Sea**

| Stock | | | Member State | 1982 quota (tonnes) |
|---|---|---|---|---|
| Species | Geographical regions | ICES/NAFO division | | |
| Cod | North Sea, Norwegian Sea | IV<br>II a)<br>(EEC zone) | Belgium | 8 700 |
| | | | Denmark | 50 000 |
| | | | Federal Republic of Germany | 31 700 |
| | | | Greece | |
| | | | France | 10 750 |
| | | | Ireland | |
| | | | Italy | |
| | | | Luxembourg | |
| | | | Netherlands | 28 250 |
| | | | United Kingdom | 114 700 |
| | | | Available for Member States | |
| | | | EEC total | 244 100 |

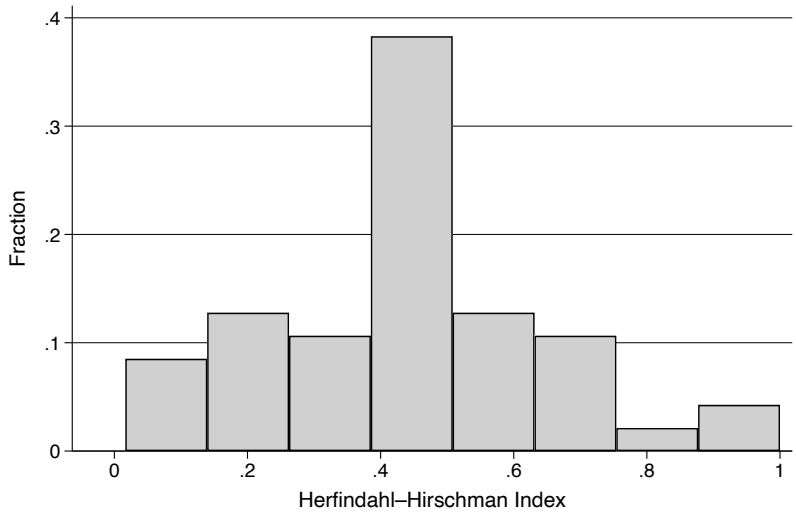
**Panel B: Cod Quota Ranges 1982**



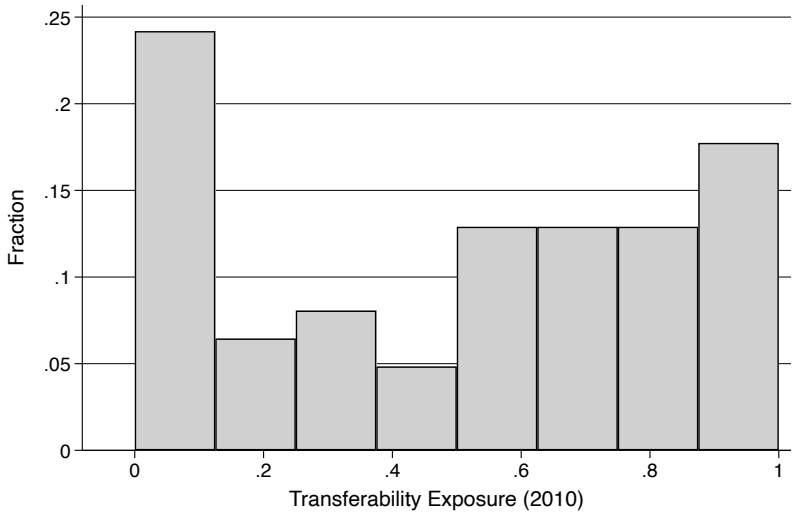
**Notes:** This figure provides an example of the fishing quota shares from 1982. Panel A presents an excerpt from the Council Regulation (EEC) No 172/83 of 25 January 1983 for cod in the north sea. Panel B presents these quota ranges for all cod with with each color showing a different range. For visual clarity, we excluded the arctic ICES sea zones 27.2.B, 27.14.A, 27.1, to avoid distortions caused by the Mercator projection.

Figure C3: **Quota Share and Property Rights Distribution**

**Panel A: Quota Concentration**

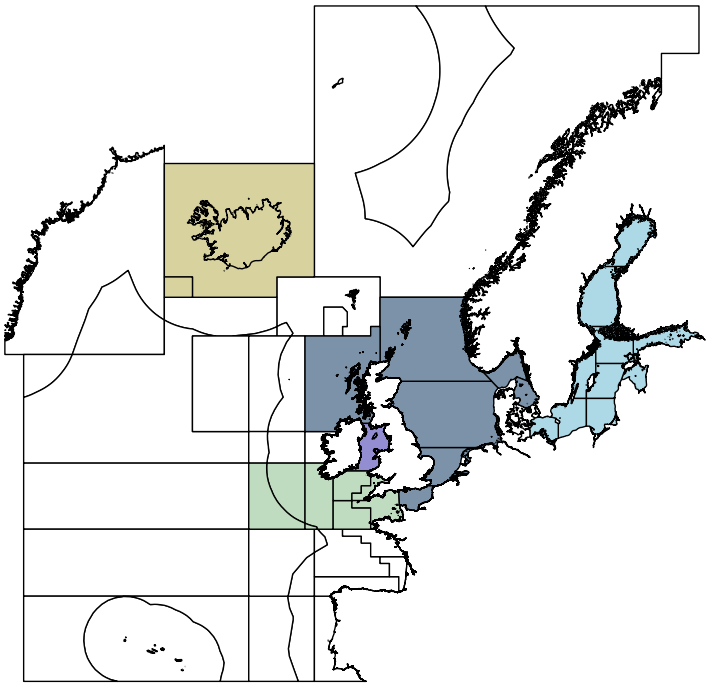



**Panel B: Property Rights Exposure**

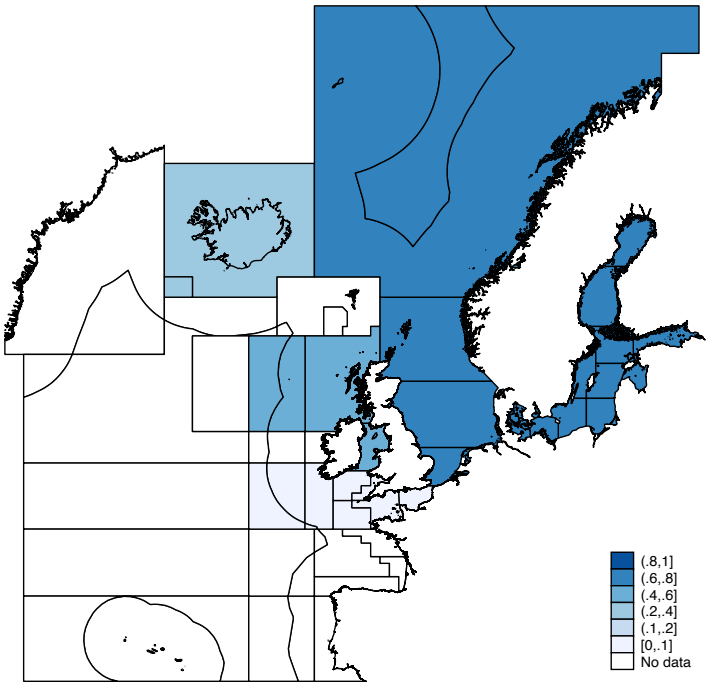



**Notes:** Panel A provides a histogram of the Herfindahl–Hirschman Index for each fish population based on their 1983 national shares. Panel B provides a histogram of the maximum of property strength exposure for each fish population. This is equivalent to the share of each fish population landed by either Netherlands, Denmark, and United Kingdom. Both of these panels are cross sectional.

Figure C4: **Map of Population Ranges and Exposure to ITQ**



Scientific Population Ranges




ITQ Average Exposure in 2010

**Notes:** This figure spatially presents key variables for cod. Panel A presents the quota ranges from 1983 with each color showing a different range. Panel B presents the population ranges as determined by ICES. Three populations of cod that also live in the North Sea with overlapping ranges are excluded for visual clarity. Panel C shows the strength of our main explanatory variable. For visual clarity, we excluded the arctic ICES sea zones 27.2.B, 27.14.A, 27.1, to avoid distortions caused by the Mercator projection.

Table C1: **Catch Shares Historical**

**Panel A: Summary Statistics of Catch Shares in 1982**

| Country | $n$ | Quota (1/0) | Conditional Ave. | SD | 99th pct. |
|---|---|---|---|---|---|
| Belgium | 40 | 0.50 | 10.14 | 12.93 | 62.5 |
| Denmark | 40 | 0.43 | 30.42 | 21.24 | 79.4 |
| West Germany | 40 | 0.60 | 13.09 | 14.38 | 66.7 |
| France | 40 | 0.73 | 21.64 | 21.78 | 76.4 |
| Ireland | 40 | 0.55 | 25.28 | 24.30 | 83.3 |
| Netherlands | 40 | 0.50 | 10.23 | 13.30 | 75.2 |
| United Kingdom | 40 | 0.70 | 33.03 | 22.15 | 80.8 |

**Panel B: Summary Statistics of Catch Shares in 2001**

| Country | $n$ | Quota (1/0) | Conditional Ave. | SD | 99th pct. |
|---|---|---|---|---|---|
| Belgium | 61 | 0.52 | 8.4 | 11.8 | 62.8 |
| Germany | 61 | 0.57 | 21.1 | 23.4 | 95.0 |
| Denmark | 61 | 0.49 | 50.5 | 34.8 | 98.8 |
| Spain | 61 | 0.16 | 35.8 | 18.7 | 92.4 |
| France | 61 | 0.75 | 20.0 | 22.3 | 76.4 |
| United Kingdom | 61 | 0.79 | 27.5 | 22.0 | 86.2 |
| Ireland | 61 | 0.41 | 32.7 | 23.5 | 83.8 |
| Netherlands | 61 | 0.48 | 11.9 | 12.1 | 75.2 |
| Portugal | 61 | 0.13 | 23.6 | 14.2 | 100.0 |

**Notes:** Panel A presents a summary table of the national catch share variable for the 61 observed fish populations using quota as reported by the New Economic Foundation. Column 1 presents an indicator variable that is equal to 1 when the nation has a fishing presence in that population. Column 2 presents a nation's average fishing share conditional on having a fishing presence. Column 3 presents the standard deviation of a nation's fishing shares across the 40 populations. Column 4 presents the 99th percentile of nation's fishing shares.

Table C2: **National Summary Table**

**Panel A: Fishing Shares (%) for Sea Zone × Fish**

| Country | N | Conditional Mean | Quota (1/0) | SD | p99 |
|---|---|---|---|---|---|
| BEL | 1120 | 3.69 | 0.22 | 3.69 | 9.26 |
| DEU | 1120 | 14.69 | 0.22 | 12.38 | 87.61 |
| DNK | 1120 | 41.27 | 0.15 | 19.80 | 98.41 |
| ESP | 1120 | 25.02 | 0.19 | 15.83 | 90.00 |
| FRA | 1120 | 31.10 | 0.37 | 22.79 | 76.43 |
| GBR | 1120 | 27.24 | 0.33 | 18.38 | 81.02 |
| IRL | 1120 | 21.58 | 0.25 | 13.50 | 80.00 |
| NLD | 1120 | 11.21 | 0.21 | 8.90 | 55.49 |
| PRT | 1120 | 20.87 | 0.09 | 10.09 | 62.25 |

**Panel B: Fishing Fleet in 1992**

| Country | Num. Vessels | Length (m) | Trawlers (1/0) | Large (1/0) | Small (1/0) |
|---|---|---|---|---|---|
| BEL | 210 | 26.02 | 0.99 | 0.54 | 0.00 |
| DEU | 2704 | 9.49 | 0.21 | 0.05 | 0.79 |
| DNK | 3618 | 13.08 | 0.24 | 0.08 | 0.56 |
| ESP | 19481 | 9.44 | 0.13 | 0.08 | 0.78 |
| FRA | 7738 | 10.85 | 0.00 | 0.04 | 0.75 |
| IRL | 2107 | 11.11 | 0.24 | 0.04 | 0.74 |
| NLD | 1502 | 19.74 | 0.54 | 0.33 | 0.38 |
| PRT | 14260 | 7.19 | 0.02 | 0.03 | 0.90 |

**Panel C: Fishing Fleet in 2022**

| Country | Num. Vessels | Length (m) | Trawlers (1/0) | Large (1/0) | Small (1/0) |
|---|---|---|---|---|---|
| BEL | 65 | 29.43 | 0.97 | 0.54 | 0.00 |
| DEU | 1250 | 9.82 | 0.20 | 0.05 | 0.78 |
| DNK | 1966 | 9.32 | 0.16 | 0.05 | 0.81 |
| ESP | 8731 | 10.98 | 0.10 | 0.08 | 0.73 |
| FRA | 6087 | 10.03 | 0.15 | 0.04 | 0.86 |
| IRL | 2032 | 9.74 | 0.13 | 0.06 | 0.86 |
| NLD | 832 | 20.15 | 0.45 | 0.30 | 0.42 |
| PRT | 7640 | 7.40 | 0.02 | 0.02 | 0.91 |

**Notes:** Panel A reports the national fishing shares for a given sea zone and fish species. This data is constructed from 1983 quota data and 2001 quota data as reported by the New Economic Foundation. Panel B reports summary statistics for fleet data in 1992 as reported by the European Union Vessel Registry. Panel C reports this fleet data for 2022.