How commercial fishing effort is managed

Article *in* Fish and Fisheries · December 2018

DOI: 10.1111/faf.12339

Received: 26 March 2018 | Revised: 28 September 2018 | Accepted: 17 October 2018

DOI: 10.1111/faf.12339

ORIGINAL ARTICLE

WILEY FISH and FISHERIES

How commercial fishing effort is managed

Christopher M. Anderson[1] | Melissa J. Krigbaum[1] | Martin C. Arostegui[1] | Megan L. Feddern[1] | John Zachary Koehn[1] | Peter T. Kuriyama[1] | Christina Morrisett[1] | Caitlin I. Allen Akselrud[1] | Melanie J. Davis[1] | Courtney Fiamengo[1] | Ava Fuller[1] | Qi Lee[1] | Katherine N. McElroy[1] | Maite Pons[1] | Jessica Sanders[1,2]

[1]School of Aquatic and Fishery Sciences, University of Washington, Seattle, Washington

[2]Food and Agriculture Organization of the United Nations, FAO Sub-Regional Office for the Pacific Islands, Apia, Samoa

Correspondence
Christopher M. Anderson, School of Aquatic and Fishery Sciences, University of Washington, Seattle, WA.
Email: cmand@uw.edu

Abstract

Wild capture fisheries produce 90 million tonnes of food each year and have the potential to provide sustainable livelihoods for nearly 40 million people around the world (http://www.fao.org/3/a-i5555e.pdf). After decades of overfishing since industrialization, many global fish stocks have recovered, a change brought about through effective management. We provide a synthetic overview of three approaches that managers use to sustain stocks: regulating catch and fishing mortality, regulating effort and regulating spatial access. Within each of these approaches, we describe common restrictions, how they alter incentives to change fishing behaviour, and the resultant ecological, economic and community-level outcomes. For each approach, we present prominent case-studies that illustrate behaviour and the corresponding performance. These case-studies show that sustaining target stocks requires a hard limit on fishing mortality under most conditions, but that additional measures are required to generate economic benefits. Different systems for allocation allow stakeholder communities to strike a locally acceptable balance between profitability and employment.

KEYWORDS

catch management, effort management, fishery management, spatial management, triple bottom line outcomes

> ...probably all the great sea fisheries...are inexhaustible.... And any attempt to regulate these fisheries seems, consequently, from the nature of the case, to be useless.
>
> T.H. Huxley (1883)
> Quoted in M. Graham, 1943, *The Fish Gate*, London, p. 111.

1 | INTRODUCTION

Huxley's comment reflects the once widely held perspective, even among biologists, that the oceans could provide functionally limitless fish for human consumption. However, following the World Wars, a large influx of effort eventually outstripped the natural productivity of many fish stocks, driven by changes in fishing technology such as diesel engines, steel vessels and mechanized gear, combined with the increase in demand from global markets made possible by refrigeration and rapid shipping.

The 1982 UN Convention on the Law of the Sea allowed countries to establish 200-mile exclusive economic zones (EEZs), evicting foreign vessels in an attempt to reserve fish for their domestic fleets. New domestic vessels replaced the foreign capacity, and reports of stock collapses arose from global keystone fisheries (Weber & Gradwohl, 1995). With the Pew Oceans Commission (Panetta, 2003) identifying overfishing as a major threat, scientists documented a levelling of global catch—including dramatically falling catches within keystone fisheries—and made sensationalist headlines that the world would be out of fish in our lifetimes (e.g., Dean, 2006).

Propelled into action by this characterization of the impending collapse of fish stocks, over the last quarter century many governments have implemented management measures that have curtailed overfishing (Worm et al., 2009) and rebuilt fish stocks (e.g., Murphy, Kitts, Demarest, & Walden, 2015). However, not all countries have found the governance capacity or political will to regulate their harvesters, and not all of the adopted management methods have been equally successful; indeed, many fisheries had some form of management in place when global attention was drawn to overfishing.

Further, even where management has achieved sustainable harvests, it has not consistently led to profitable fishing businesses that offer stable, well-paying jobs that support the communities out of which the fisheries are based (Branch et al., 2006). A joint World Bank–FAO report estimates we forego US$83 billion per year in fishery benefits globally. These losses arise from stocks that have been fished beyond their optimal productivity, and from investing in more capacity than necessary to catch the available fish (World Bank 2017; cf. Costello et al., 2016). The impact of these losses is particularly acute in developing countries, where fisheries provide both food security and critical livelihoods in vulnerable communities (Béné, Hersoug, & Allison, 2010).

This paper provides a critical survey of the major approaches that fisheries managers have used to constrain the behaviour of fish harvesters in order to achieve sustainable fisheries. We identify archetypical case-studies to illustrate how individual fish harvesters respond to the incentives presented by each approach and offer an empirical characterization of the outcomes that arise. Our analysis is rooted in the bioeconomic model of harvester behaviour in order to extend our characterization of these management tools beyond their ability to achieve ecosystem health. We use the terminology "fish" throughout, but only for simplification as the characterization of these management schemes extends to fisheries targeting invertebrates and chondrichthyans. We focus primarily on stock status as our measure of biological outcomes; harvesters' profitability as our measure of economic performance; and employment and safety as our measures of social outcomes (cf. Anderson et al., 2015). While fisheries may select management programmes to achieve different objectives, we evaluate the extent to which each programme supports biological, economic and social performance.

This paper also contributes to the understanding of management tools by characterizing the relationships among the major fishery management approaches and representing them in a Venn diagram (Figure 1). Individual fisheries often try a succession of approaches and make refinements as they learn what does and does not work in each case (Branch et al., 2006) and as stakeholder communities act to improve their outcomes. We approach the diagram by describing the incremental effects of these adjustments along three typical paths: limiting catch, limiting effort and limiting spatial access. Managers' choices among these paths are often governed by their specific enabling legislations, governance conditions and philosophies of management. Each path begins with unregulated open access and moves towards the centre of the diagram, with each step incorporating a new feature of management, while inheriting many of the traits from the previous step. The first path, characterized by limiting catch, travels down the left of the diagram, first establishing limited entry to the fishery and then adding binding constraints on the total quantity of fish caught collectively or individually. The second path, characterized by limiting effort, travels down the right of the diagram, utilizing a system of restrictions on harvesting inputs such as time fishing or gear usage in order to reduce mortality. The third path, characterized by limiting spatial access, travels right to left across the bottom, specifying spatial regulated-take or no-take zones, with a range of effort or catch controls where fishing is permitted.

2 | NO REGULATION: OPEN ACCESS

The outer region of Figure 1 represents unregulated open access, where there are no managerial constraints imposed on the fishery, neither limiting the number of harvesters that enter the fishery, nor limiting the quantity of harvested product. Although nearly every fishery has basic restrictions on fishing practices (e.g., mist nets, dynamite) that are not calibrated to sustainable levels of effort, fisheries with only those measures are considered unregulated open access here.

The key decision for an individual harvester is whether, and how intensively, to participate in the fishery. This is represented in the bioeconomic model of harvester behaviour, which relates stock dynamics, fishing revenues and costs (see Field, 2008, ch. 13). If the revenue from catching additional fish is expected to be higher than the costs of doing so, new participants will decide to enter the fishery, and existing harvesters will continue to invest in more harvesting capacity, hoping to capture a larger share of the fish and profits for themselves. As a result, total fleet capacity and effort increase, each vessel pays more to operate their increased capacity, and the per-vessel harvest decreases due to increased competition and declining stocks. This entry dynamic will only end once no harvester has an incentive to enter the fishery, which occurs when harvesters would rather take other jobs. Economists consider this a "zero-profit" outcome because economic profit represents benefits above what people or resources could earn in their next-best employment; a fishery can be "zero profit" while firms make an accounting profit and participants still make a normal wage. The result is an economic "tragedy of the commons" where, despite the fact that fish are provided for free, no profit is made (Gordon, 1954; Hardin, 1968; Scott, 1955). Without regulation, self-interested harvesters will not slow fishing when the stocks reach low levels if there is still profit to be made. In many fisheries, by the time this zero-profit level of effort is reached, fishing pressure is higher than the stocks can sustain, leading to collapsing fish stocks and a biological tragedy of the commons.

These biological and economic tragedies are illustrated by the New England groundfish fishery, which had unregulated open access from the colonization of New England until the mid-1970s. Catches peaked in the 1860s, but the fishery remained stable until the 1960s. Sail-powered boats and an inability to refrigerate catches meant fishing was costly enough per unit that participation was held in equilibrium at low effort and high stock levels. The introduction of new technologies, such as diesel engines, made it possible for vessels to harvest more efficiently, and improvements in shipping and refrigeration expanded markets. As a result, per-unit costs decreased, revenues increased and profit opportunities returned. In response, existing vessels increased their harvesting power, and new ones entered: the fleet grew from 825 vessels in 1977 to 1662 in 1990, during which time the populations of key species declined 65%. With few fish, 14,000 fishing jobs were lost, remaining harvesters were unprofitable, and coastal communities suffered (Weber & Gradwohl, 1995).

The same dynamic is currently playing out in the developing world, especially as export markets are established. In the Indonesian blue swimming crab (*Portunus pelagicus*, Portunidae) fishery, fishing effort increased 475% between 2006 and 2016, while catch increased only 52% (Hamid, Wardiatno, Lumbanbatu, & Riani, 2016). On Lake Victoria, gear used to catch Nile perch (*Lates niloticus*, Latidae) and tilapia (*Oreochromis niloticus*, Cichlidae) increased from 99,800 gillnets to 161,800 gillnets between 2000 and 2004, while catch of the two species fell from a combined 132,000 tonnes to 75,000 tonnes over the same period (Njiru et al., 2007). As is typical of open access fisheries, the profits of early entrants in these fisheries attracted additional vessels, leading to lower profits as fishing costs increased and high rates of exploitation resulting in the catch of smaller or immature individuals (De Alessi & Warmbrunn, 2014; Njiru et al., 2006). In both cases, as in many communities throughout the developing

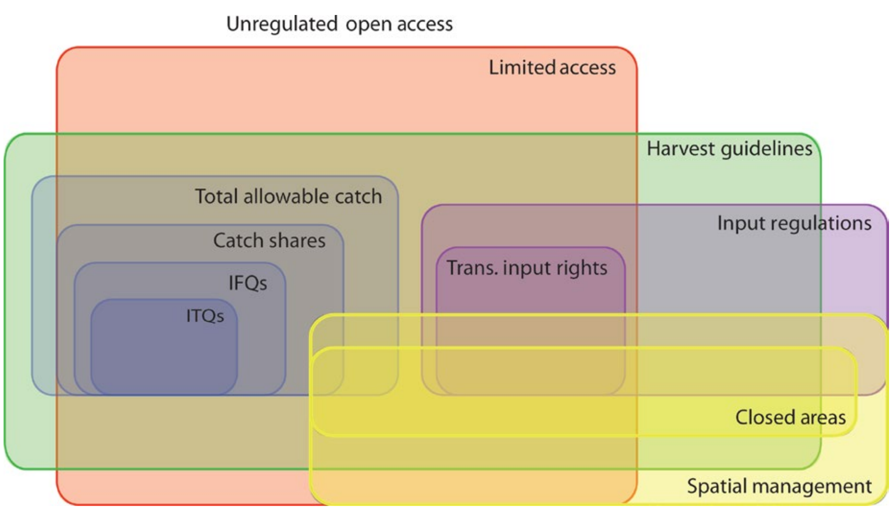

FIGURE 1 Venn diagram representing the relationships among common approaches to fishery management, with regions of greater overlap indicating additional restrictions. Beginning from unregulated open access, the diagram represents three pathways: 1) limiting catch, beginning with limited access and adding restrictions on total allowable catch, allocating harvest rights through catch shares, individual allocation through individual fishing quota (IFQ) and individual transferable quota (ITQ); 2) limiting effort through establishing non-binding harvest guidelines, imposing input restrictions and then transferable input rights; and 3) controlling spatial access by establishing regulated-take or closed no-take areas, with the range of effort or catch controls applying within regions where fishing is permitted [Colour figure can be viewed at wileyonlinelibrary.com]

world, there are few other employment opportunities, so vulnerable harvesters have turned to processors for loans, committing their future fish landings to individual buyers who need not pay competitive prices and exacerbating income inequality (De Alessi & Warmbrunn, 2015; Geheb et al., 2008). This pattern is mirrored in the small-scale fisheries of West Africa where the number of harvesters increased from 953,000 in the 1950s to 1.74 million in 2010 and catch quadrupled, but, despite increases in landed value, fisher income is very low (Belhabib, Sumaila, & Pauly, 2015).

While entry eroded the profits of these potentially valuable fisheries, it is important to understand the tragedy of the commons as the consequence of the choices individual harvesters make in pursuit of their own profits. There are many fish populations with low market demand or high harvesting costs—like lanternfish (Myctophidae) and jellyfish (Medusozoa)—which are in bioeconomic equilibrium at low fishing effort and sustainable stock levels, with little threat of overfishing even without regulation. In other cases, especially in the developing world, limited technology, limited capital or a limited population of potential harvesters may limit effort sufficiently to allow sustainable profitability. In the small-scale fisheries of northern Sulawesi, large villages with better employment alternatives and villages that are less connected to urban demand centres see reduced levels of overfishing (Liese, Smith, & Kramer, 2007). However, when the target species is valuable, unregulated open access is characterized by over-exploitation, excessive harvesting capital, little or no economic profit, and the collapse of fishing communities. Managers have attempted to address these negative impacts by adopting management measures from the three paths outlined below.

3 | PATH 1: LIMITING CATCH

The management approaches in this section attempt to limit catches by regulating who can fish and how much they can catch.

3.1 | Limited entry

Limited entry management forms the base of the first path of management approaches commonly used in commercial fisheries. This strategy limits the number of vessels or harvesters that participate in a fishery in an attempt to restrict total fishing mortality to sustainable levels. The number of vessels is often calibrated to an intended level of mortality, sometimes supported by limits on gear or per-trip landings quantities.

While the number of harvesters is capped, individual harvesters find they can increase their catches by fishing more frequently, for longer amounts of time, or by investing in improved technologies. Harvesters make costly investments in vessel or gear improvements, also known as “capital stuffing,” to outcompete others for a larger share of available fish (Townsend, 1985), often leading to higher levels of catch than managers intended. Although harvesters are not able to freely enter the fishery, limited entry fisheries have insufficient control over fishing mortality and often continue to exhibit overfishing.

For example, although the U.S. West Coast groundfish trawl fishery successfully reduced the number of vessels fishing with a limited entry programme that began in 1994, this management system failed to curtail the high levels of overfishing, resulting in low catches for valuable species and therefore depressed revenues (The Nature Conservancy, 2008). Ultimately, in 2000, the decreased catches led the US government to declare the fishery a federal disaster.

Other implementations of limited entry programmes include the Bristol Bay drift gillnet salmon (*Oncorhynchus nerka*, Salmonidae) fishery, which introduced 1669 limited entry permits in 1974 (Schelle, Iverson, Free-Sloan, & Carlson, 2004). These fishermen compete to place their nets closest to “the line” where fish approach the mouth of the river in which they will spawn. Though boats are restricted to 32 feet, fishermen continually invest in stronger, more powerful vessels to race for position and push others out of the way. Once pursued by one-tonne wooden sailboats, new vessels are aluminium, weigh 20–40 tonnes and can top 1,000 horsepower. While in-season closures prevent overfishing, the fishery has at least five times the capacity necessary to harvest the available fish, meaning profits have been reduced by five times the necessary expenditure on vessels and fuel. Capital stuffing was also seen in the British Columbia salmon fishery, which implemented limited entry to decrease the number of vessels from 6500 in 1968 to 5300 in 1977. The value of the fleet increased from $73.4 to $273 million over the same period. This capital stuffing was paired with a decline in the quantity of harvesting jobs from 9,600 to 8,600 labourers (Fraser, 1979).

Some short-lived, highly fecund species, such as shrimp, are robust enough to be adequately managed biologically through limited entry. In many of these fisheries, high recruitment variability and technological limitations prevent over-exploitation and dampen capital stuffing. For example, the South Australian Spencer Gulf prawn (*Melicertus latisulcatus*, Penaeidae) stock is estimated to be at or above maximum sustainable yield, and has seen increased catch per unit effort since implementing limited entry with supporting vessel and gear size restrictions, providing jobs and income to local communities (Dixon, Noell, & Hooper, 2013; Noell & Hooper, 2015). Similarly, the North Carolina brown shrimp (*Penaeus aztecus*, Penaeidae) fishery is biologically sustainable, though the high number of licensed vessels leads to inefficient competition and excessive costs, amounting to a 20% reduction in revenue (Huang & Smith, 2014).

As a stand-alone management system, limited entry does little to disincentivize increases in fishing power to capture any existing profit. Like open access, limited entry programmes are characterized by biological over-exploitation, excess harvesting capital, and little or no economic profit.

3.2 | Total allowable catch

Once they experience the weak control over fishing mortality seen in limited entry systems, management agencies often move towards the centre of Figure 1 and implement a fleetwide total allowable catch (TAC). TAC management establishes an annual or seasonal limit to fishing mortality, usually based on scientific advice or catch data. Catch is tracked during the season using a catch-accounting system, and a regulator closes the fishery once the TAC threshold is met so overfishing is unable to occur.

This strict control over the total quantity harvested supports biological sustainability. However, a TAC system establishes a competition among fishermen: the only way an individual can increase their catch is to catch it before other harvesters. This induces a "derby" or a "race-to-fish," where participants invest in larger and faster boats, more powerful engines, additional gear and more crew, so as to give themselves an advantage. This continues until investing in more input does not yield more profit— another form of the economic tragedy of the commons where the total harvesting capacity of the fleet far exceeds what is needed to harvest the TAC, and fleetwide economic profit is zero.

Ecological success, coupled with poor economic performance, is seen in the British Columbia halibut (*Hippoglossus stenolepis*, Pleuronectidae) fishery, which was TAC-managed prior to 1991. With the impending threat of fishery closure, the harvesters engaged in a competitive race-to-fish, which incentivized capital stuffing and increased crew size, rewarding vessels that invested in competitive catching power, thus dramatically increasing the cost to fish (Casey, Dewees, Turris, & Wilen, 1995). Over the 1980s, the race to catch the TAC shortened the open season from 2 weeks to merely one or two days a year. During these short seasons, harvesters would fish regardless of bad weather or hazardous working conditions, compromising safety for revenue. Since all the halibut was landed over a short period of time, these high landings resulted in low market prices and low product quality, as processors sacrificed product development and opted for the frozen market to minimize landings waste (Casey et al., 1995). Other TAC-managed fisheries, like the "Deadliest Catch" Bering Sea Aleutian Island crab (*Paralithodes* sp. [Lithodidae] and *Chionoecetes* sp. [Oregoniidae]) fishery before implementation of quota shares (Fina, 2005), the Finnish herring (*Clupea harengus*, Clupeidae) fishery from 2001 to 2016 (Kulmala, Peltomäki, Lindroos, Söderkultalahti, & Kuikka, 2007) and the Rhode Island fluke (*Paralichthys dentatus*, Paralichthyidae) fishery pre-2009 (Scheld, Anderson, & Uchida, 2012) also observed sustainable stocks, but low economic efficiency and social trade-offs: crew jobs were numerous, but duration, working conditions and safety were poor.

Once a management system includes a TAC as part of its management plan and moves within the TAC area on the diagram, if the TAC is scientifically informed and adequately enforced, the fishery typically maintains or increases target stocks (Da Rocha, Cerviño, & Villasante, 2012) and improves biological outcomes. However, it is important that the scientific TACs be followed. Under the EU Common Fisheries Policy, TACs are recommended by ICES scientists and approved by the EU, but then increased through the political process of allocating them among member countries, leading to biological and economic failure (Khalilian, Froese, Proelss, & Requate, 2010). Similarly, if TAC recommendations are scientifically uncertain or heavily reliant on fisheries-dependent data where increases in catchability due to technological advances are not well accounted for, biological sustainability can be undermined (Eigaard, Marchal, Gislason, & Rijnsdorp, 2014; O'Leary et al., 2011).

3.3 | Catch shares

Mitigating the race-to-fish requires providing harvesters security in their shares of landings. Catch share management allocates a secure share of an established TAC to communities, cooperatives or individuals for their exclusive use (NOAA 2017). Group-based catch shares, the focus of this section, allow groups of harvesters to cooperatively decide how to manage their collective catch quota allocation to maximize their benefits. Each group's allocation is generally based on member harvesters' catch history; harvesters typically need to join a group to participate in the catch share. In practice, most groups partition this historical share to individual harvesters to fish under cooperatively established rules, creating a de facto individual quota system, described in the following two sections.

Since catch share groups have a secure portion of the TAC to manage, they can cooperate to solve the internal allocation problem, eliminate the race-to-fish and address associated problems that limit the value of the fishery. Slowing fishing leads to decreasing costs, reducing capital stuffing and increasing efficient harvesting practices to maximize profit (Birkenbach, Kaczan, & Smith, 2017). Slower, more careful fishing may also allow harvesters to more fully utilize target and non-target quota to increase revenues within the group (Brinson & Thunberg, 2016). However, increases in utilization in a multispecies fishery can be limited due to relative TAC values that differ considerably from harvest ratios. For example, the US West Coast groundfish trawl fishery implemented catch limits designed to rebuild populations, but the constraining non-target species quotas likely led to limited utilization of target species since the fishery would stop once quota for jointly harvested non-target species was exhausted, regardless of the amount of target quota remaining (Kuriyama, Branch, Bellman, & Rutherford, 2016).

Overcapitalized like the Bristol Bay salmon fishery, 77 of 100 salmon fishermen in Chignik, Alaska, formed a co-op which received a catch share allocation. They fished using only the 19 most efficient member vessels, but paid out dividends equally to all co-op members. This consolidation of effort led to reduced maintenance, gas and labour costs. The co-op partnered with a processor to arrange for live fish deliveries, improving product quality. The salmon was marketed under a special brand, leading to a price premium (Metzner & Ward, 2002). This strategy led to a 20%-40% increase in net revenues (Deacon, Parker, & Costello, 2008). Because proceeds were shared equally among members, the business owners who benefitted most were those who would not have fished otherwise, while members in the top 25% of previous earners did not see increases or decreases in annual income (Knapp, 2008).

The Rhode Island fluke sector pilot programme was a group of eight mid-sized New England groundfish trawlers who organized a catch share around an allocation of 11% of Rhode Island's fluke TAC. The rest of the groundfish fleet targeted fluke jointly with other groundfish species, in a competitive derby, with a regulatory requirement to discard fluke once the landings limit was met. The sector members, by agreeing not to discard and count all catch against their catch share allocation, sat out the race-to-fish when the prices were low, and instead landed their allocation when other vessels were forced to discard and prices were high. This reduced dead discards of fluke, and raised revenue an average of $70,000 for sector members, and a total of $250,000 for non-sector members, as less fish were landed in the derby (Scheld et al., 2012). Similar structures relaxing closed seasons have increased temporal flexibility and prices for groundfish fishermen on Cape Cod (Pinto da Silva & Kitts, 2006).

The business flexibility allowed by group-based catch shares is often provided to industry when strict biological reductions must be implemented. Following the 2010 implementation of allocations to 17 self-identifying groups in the New England multispecies groundfish fishery, stocks are recovering under catch share management (Murphy et al., 2015). Flexibility in harvest strategies led to an estimated 18% increase in ex-vessel revenue gains and an 8% increase in job duration in the first year of implementation (Scheld & Anderson, 2014).

Also facing depletion of target stocks, Japanese coastal fisheries implemented community-based catch shares in 1949. Cooperative strategies have improved stocks and habitat, and harvesters have profited from increased revenue per unit effort (Ministry of Agriculture Forestry and Fisheries, 2016; Uchida & Makino, 2008). Japan's coastal community has also benefited from an increase in employment opportunities in both the harvesting and processing sectors (Mcilwain & Hill, 2013; Ministry of Agriculture Forestry and Fisheries, 2016). Korean coastal cooperatives have seen similar improvements (Uchida, Uchida, Lee, Ryu, & Kim, 2010). Likewise, converting a fleetwide halibut by-catch cap to an allocation to co-ops in the Bering Sea/Aleutian Islands non-pollock groundfish fleet led to collaborative gear development and information sharing about by-catch hot spots (Abbott & Wilen, 2010).

Group-based catch share systems have a record of resolving the race-to-fish, reducing costs and improving product quality and enhancing profitability. The extent of the cost reduction is determined by incumbent fishermen, who collectively determine a balance between harvesting efficiency and maintaining broad levels of employment within fishing dependent communities. Since fishing mortality is based on a TAC, catch share fisheries inherit the positive ecological outcomes associated with TAC management for the target species. Enhanced cooperation and information sharing can also support improved quota utilization and by-catch avoidance in multispecies fisheries. However, to realize these benefits, it is essential that the groups receiving the allocation be able to agree on an allocation and cooperate, and as a result, most cooperative catch shares have been implemented in community-based fisheries or fisheries with large industrial participants. When this does not happen, managers can set allocations for each harvester.

3.4 | Individual fishing quota

Individual fishing quota (IFQ) systems are a type of catch share where managers allocate non-transferable shares of the TAC to individual vessels or fishermen, rather than a group. Typically, individuals are given quota shares, which are permanent and denominated in a percentage of the TAC. Annually, these quota shares beget quota pounds, or pounds that can be landed in the current year, reflecting the individual's percentage quota share of the current year's TAC. Like other catch share systems, IFQ management attempts to counteract the harmful economic incentives associated with the race-to-fish. It does so by focusing individual harvesters on increasing their profit per pound by improving market timing and handling practices to get the highest prices (Homans & Wilen, 2005) and by minimizing harvesting costs.

Individual fishing quota systems strengthen TAC-based management strategies by disincentivizing discarding, quota overages and politically inflated TAC levels. Individual stakeholders are granted a secure stake in the future health of the fishery and are therefore driven to maintain or increase the value of their property via ensuring the productivity of the fishery (GMFMC 2013; National Research Council 1999; SEDAR 2013). As a result, individual quota-based fisheries have a record of reducing overfishing and rebuilding stocks (Branch, 2009).

In 1991, the British Columbia longline halibut fishery implemented a non-transferable quota system. Prior TAC-based management saw extensive capital stuffing and a race-to-fish that reduced season length to 6 days (Dewees, 1998). In the first year of IFQs, fish were landed throughout the 8-month season and in smaller daily quantities that allowed 21% more processors to participate. Most of the new firms specialized in fresh products, increasing fresh market production from 42% to 94%—leading to a 55% increase in ex-vessel price (Casey et al., 1995). Crew sizes were somewhat smaller, but those crew that did work earned more money. While transferability was initially prohibited out of concern for consolidation, after 2 years (1991–1993) harvesters decided they wanted to scale their businesses and approved a set of rules for trading quota pounds, or leasing. Full transferability of quota shares followed in 1999 (Gilroy, Erikson, & Mactavish, 2011).

3.5 | Individual transferable quota

Individual transferable quotas (ITQs), found at the inner end of the limiting catch path in Figure 1, are IFQs that allow individual quota shares to be traded, or quota pounds to be leased, in exchange for money, other quota or a share of landings revenue. Quota share is an asset that, like a stock certificate, provides access to an annual stream of profits and is valued on that basis: to a fishing business, the value of a single-year lease of a quota pound is the increase in profit provided by the opportunity to harvest the additional pound; the value of quota share is the present value of that expected increase in profit into the future (Grainger & Costello, 2016).

The addition of transferability incentivizes inefficient harvesters to sell their quota to more profitable harvesters who are willing to pay more for the quota than the inefficient harvesters would earn by catching it themselves. As such, ITQs support the positive biological and individual-level profit maximizing outcomes of IFQ systems, but further improve the economic efficiency of the fishery as a whole, redirecting harvest towards those who can earn the most profit with it. Over time, the transferability of quota can reduce fleet overcapacity and increase fishery rents, to varying extents based on the fishery (Connor, 2001; Färe, Grosskopf, & Walden, 2015; Haynie, 2014; Yandle & Dewees, 2008).

ITQs inherit the positive biological outcomes of IFQ and TAC systems and retain the incentives to reduce cost and improve value through increased season length. This allows harvesters the flexibility to land their fish during favourable market conditions with improved product handling, thereby increasing product quality and value (Ginter, 1995; Tveteras, Paredes, & Pena-Torres, 2011). For example, the Gulf of Mexico red snapper (*Lutjanus campechanus*, Lutjanidae) fishing season extended to year-round, from a pre-ITQ average of 109 days, enabling harvesters to improve their scale of operation and time their harvest to acquire higher sale prices (Agar, Stephen, & Strelcheck, 2014). Since quota prices reflect expected profits in the fishery, increases in quota share purchase and lease prices of 145% and 37%, respectively, from 2007 to 2011, show that profits increased (Agar et al., 2014).

While biological performance and profitability increase, fisheries transitioning to ITQs often undergo significant structural transformations (Brinson & Thunberg, 2016). Quota market forces drive changes, some of which may be undesirable to managers or stakeholders, especially adverse social impacts (Branch et al., 2006). First, the transfer of quota to those who can use it most profitably often leads to consolidation, especially if implementation is accompanied by a reduction in TAC. For example, 61% of Bering Sea crab vessels exited in the first year of ITQ management (NPFMC, 2017), the British Columbia halibut fleet decreased in size by 66% in the 12 years after legalizing full transferability (Gilroy et al., 2011), while the Gulf of Mexico red snapper fleet decreased in size by 17% in the first 5 years of its ITQ programme (Agar et al., 2014). This is often an intended outcome in fisheries that are known to be overcapitalized, as quota sale provides a voluntary mechanism for some quota holders to leave the fishery, with compensation from those who remain. However, this process can favour larger operations at the expense of small-scale harvesters or native communities (Carothers, Lew, & Sepez, 2010; Connor, 2001). Additionally, the profitability supported by ITQs leads to share price increases, which presents a high financial barrier to entry by aspiring owners (McCay, 2004). Importantly, quota allocations are nearly always granted only to boat owners, so hired captains and crew are put out of work and not compensated when the new asset incentivizes high-cost owners to sell and the fleet consolidates; in isolated communities without other sources of employment, this effect can lead to social collapse (cf. Carothers, 2008).

Second, transition to an ITQ system often results in changes in the character of employment in the fishery. Numerous short-term fishing jobs can be lost and replaced by fewer long-term jobs: in the Bering Sea king crab fishery, average season length increased 2–3 times, with 87% of the remaining crew experiencing some increase in earnings (Abbott, Garber-Yonts, & Wilen, 2010). As the market decreases the total number of crew needed to harvest in an ITQ system, the remaining or entering participants receive a higher potential income, as they each earn a higher share of revenue (Carothers et al., 2010). Furthermore, ending the race-to-fish means jobs are safer, as harvesters are less likely to go out in bad weather as in the US West Coast groundfish trawl fishery (Pfeiffer & Gratz, 2016) and the Gulf of Mexico red snapper fishery (Boen & Keithly, 2012). In the "Deadliest Catch" Alaska crab fishery, crew fatalities dropped from an average of over one per year to one total in the nine years following ITQs (NPFMC, 2017).

More recent ITQ implementations often include measures to mitigate adverse social outcomes. One common strategy is to cap individual quota holdings, to limit consolidation. The pioneering New Zealand mixed-stock inshore and offshore quota management system saw large companies acquire quota at the expense of small-scale harvesters, who now mostly fish-for-hire and express dissatisfaction with the system (Connor, 2001; Yandle & Dewees, 2008). In the Gulf of Mexico red snapper fishery, there was a restrictive 6.0203% cap on shares, which kept the market for shares competitive and prevented market concentration (Agar et al., 2014). The Alaska halibut programme supplemented consolidation restrictions with prohibitions on quota transfer from smaller to larger vessel categories, and a quota owner-on-board requirement. These measures have maintained benefits to active fishermen in the small boat fleet, but at a cost of $117 million in quota value (Kroetz, Sanchirico, & Lew, 2015).

Another approach to managing adverse community effects is to allocate some quota share to community groups. The rural western Alaska Community Development Quota (CDQ) programme grants a portion of the commercial groundfish and crab quotas for coastal communities (Ginter, 1995; Holland, 2000). Some CDQ quota is leased to harvesters, with the proceeds used to support community programmes, while other quota is distributed to community residents or leased with the expectation that landings be processed in the community. Over a 19-year period, this programme increased native organization assets from $13 million to $938 million USD and local jobs by more than sixfold (Haynie, 2014), demonstrating how ITQs can provide community benefits while enhancing fishery efficiency.

4 | PATH 2: LIMITING EFFORT

In situations where governance and enforcement capacity are limited or not cost-justified, rather than monitor and regulate total catch, many fisheries instead choose to limit fishing mortality by regulating technical inputs with restrictions on fishing effort by size of vessels, type and amount of fishing gear, or number of open fishing days

(Branch et al., 2006; Pope, 2009). This second path through Figure 1 begins by establishing a biological goal and then layering restrictions to limit overall fishing mortality.

4.1 | Guideline harvest levels

When managers first recognize the need to constrain harvests, they often begin by identifying guideline harvest levels (GHLs) or target levels of fishing mortality that they hope the fishery will not exceed. GHLs can be based on past harvests that are believed to be sustainable or estimates of sustainable mortality from advanced stock assessment models. GHLs differ from TACs in that a TAC will close the fishery once reached, whereas a GHL will not. Although a biological target exists, TACs or any other management measures are not implemented.

As a primary management tool, GHLs are prevalent where measuring fishing intensity is difficult, capacity to implement stronger measures is limited, or enforcement is poor. In these cases, stating a desired mortality with a GHL fails to provide an incentive for reducing catch or operating more profitably. For example, eastern Atlantic sailfish (*Istiophorus albicans*, Istiophoridae) are targeted by small-scale fisheries in West Africa and caught incidentally by industrial tuna long-liners. While the International Commission for the Conservation of Atlantic Tunas (ICCAT) establishes a harvest guideline, it has not been able to coordinate measures to stay within the GHLs across the different fleets and flag states. As a consequence, there are no regulations to limit individual harvesters and this stock is currently heavily over-exploited (ICCAT, 2017), and the West African fishermen targeting sailfish from canoes are quite poor, with fewer than 40% having running water (Brinson, Die, Bannerman, & Diatta, 2009). Without any restrictions on effort or spatial access, GHLs result in the same deleterious economic and biological outcomes as unregulated open access.

4.2 | Input controls without guideline harvest levels

When a management agency lacks the necessary scientific or regulatory basis to establish biological targets, input controls are implemented as the sole form of management. This is especially common in complex multispecies fisheries, especially reef fisheries, or where the objective is to protect traditional or historical fishing habits without a specific mortality goal. With limits on only some inputs, individual harvesters begin capital stuffing like in limited entry systems, scaling up all unregulated inputs to increase their harvesting power in the face of regulations. Without species-specific harvest guidelines, harvesters can increase earnings by targeting the most profitable species, accelerating their decline.

For example, in the East China Sea, fisheries managers have enacted seasonal closures, mesh size restrictions, gear restrictions and other input controls, to maintain production for the many subsistence harvesters that rely on this resource (Zou, 2003). Absent limited entry, the number of vessels and their engine power has increased, leading to over-capacity (Yu & Yu, 2008). While historical participation continues, the restrictions mandate the use of old, often inefficient technology, increasing the cost of fishing and diminishing profits for these harvesters, as catch per unit effort has decreased. This pattern of investing to scale up unregulated inputs and sapping profits while undermining ecological sustainability has also been observed in Malaysian marine fisheries (Saharuddin, 1995) and the Gulf of Thailand mackerel (*Scomberomorus* sp., Scombridae) fisheries (Panayotou & Jetanavanich, 1987).

4.3 | Input controls with guideline harvest levels

When selecting the level of input controls, fisheries managers often try to select restrictions that will achieve the GHL. However, even when input controls are well-calibrated to status quo effort levels, harvesters still increase their own catch by fishing more intensively through capital stuffing. This causes fishing mortality to decrease less than intended, or even continue to increase, exceeding the GHL and at a higher cost.

For example, the New England groundfish fishery implemented the Atlantic Demersal Fisheries Plan in 1985 to combat declining fish stocks. While this plan included input controls such as mesh size restrictions and seasonal limits, the rules resulted from a contentious debate between the National Marine Fisheries Service, the New England Fishery Management Council and industry members, leading to strong political influence over the policy and lenient restrictions (Acheson & Gardner, 2011). Coupled with a large increase in the number of vessels due to federal loan programmes financing additional permits, these input controls did very little to limit effective fishing effort. As a result, stocks continued to plummet, Atlantic cod (*Gadus morhua*, Gadidae) continued to collapse, and total catches in New England declined from over 200 million pounds in 1983 to less than 50 million pounds in 1993 (Acheson & Gardner, 2011). This fishery demonstrates that lenient input regulations, especially without limited entry, fail to curtail high fishing effort or end biological decline.

Beginning in 1993, the New England Fishery Management Council sought to further combat declining groundfish stocks by establishing a moratorium on new permits and by limiting harvesters to 88 days-at-sea each season (Acheson & Gardner, 2011). With restrictions on the time allotted for fishing, harvesters focused their effort on areas closer to shore and on higher value, often overfished, species to maximize their profits for each day allowed (Holland & Sutinen, 1999). They invested in the size and power of fishing vessels, so total fishing capacity was not reduced despite the time constraint on fishing. Ecologically, limiting the number of days-at-sea did not rebuild the most important overfished stocks, such as Atlantic cod (Brewer, 2011). Managers reacted by reducing annual days-at-sea fourfold to as few as 20, inducing considerable consolidation of the fishery, but with limited success in rebuilding key stocks (Acheson & Gardner, 2011; Brewer, 2011; Thunberg, Kitts, & Walden, 2007). Fishing towns throughout New England saw social collapse as a fleet that supported 3,033 permits in 1994 had only 574 active permits by 2007, with declines varying by port (Brewer, 2011). Distrust of management persisted as the remaining fishery participants faced rising costs, diminished profits and declining stocks (Acheson & Gardner, 2011).

Empirically, the unregulated effort increases associated with input regulations are less problematic for highly abundant, highly fecund species—such as small pelagic fish, skipjack tuna or shrimp—for which precise effort control is often less consequential. The Parties of the Nauru Agreement (PNA) established a vessel-day scheme for skipjack tuna (*Katsuwonus pelamis*, Scombridae), where distant water fishing nations (e.g., Korea and Japan) must bid for vessel-day units that allow them to fish within the EEZs of PNA countries (including Papua New Guinea, Kiribati and six other Pacific Island countries) with one vessel for 1 day. The auction efficiently allocates fishing rights to the harvester that can fish them the most profitably (Yeeting, Bush, Ram-Bidesi, & Bailey, 2016). While the input control does not tightly limit joint harvest of overfished bigeye tuna (*Thunnus obesus*, Scombridae), a majority of the catch is from the healthy skipjack tuna stock (Havice, 2013) by successful distant water fishing businesses. Still, capital stuffing occurs for these vessels, as the fishing operations utilize an increased number of fish-aggregating devices (FADs) to maximize their catch for a given vessel-day. While the use of these devices increases the total mortality of skipjack, the stock is sufficiently robust to this additional pressure, due to their high fecundity. However, the use of FADs has negative consequences for by-catch such as turtles and sharks (Yeeting et al., 2016). Revenue from the vessel-days has also allowed the developing PNA countries to capture economic benefits from fishing resources, though they are facing challenges associated with the transition to a cash-based economy, including social strain, shifting gender roles and a potential loss in historical participation in traditional fishing for food (Barclay, 2010).

4.4 | Individual transferable input controls

Transferable input control systems, the end of the input control pathway through Figure 1, set a fisherywide cap on a given input, allocate it among fishermen and allow fishermen to trade it. This system combines the weak harvest limitations of input controls with incentives to transfer inputs to harvesters who can use them most profitably, like an ITQ system, while providing a voluntary mechanism for some incumbents to leave the fishery with compensation from those who remain.

Adding transferability to an input control system does little to counteract the weak regulation of harvest associated with input controls, perhaps even exacerbating the problem as input controls are traded to the most effective and productive harvesters. As the number of days-at-sea allocated in the New England multispecies groundfish fishery declined below the level sustainable for all businesses, transfer of days-at-sea was introduced in 2004 (NEFMC & NMFS, 2003). During the first 3 years of the transferability programme, the lack of mortality controls on vulnerable and valuable stocks allowed for the number of stocks that are both overfished and experiencing overfishing to increase from 7 to 11 (Northeast Fisheries Science Center, 2008). Despite a steep 69% reduction of total days-at-sea allocated between 2004 and 2006, harvesters were still incentivized to maximize their fishing power within the given input restrictions, by fishing more hours per day with more gear to maximize the value of the catch for each day-at-sea permit, regardless of the ecological health of the target stocks (Acheson & Gardner, 2011). The transferable days-at-sea programme also attempted to control species-specific catch through stock-specific trip limits, intended to encourage harvesters to be more selective and avoid overfished stocks. However, these limits caused harvesters to discard—often dead—any of that species above the limit rather than avoid catching them, or return to port early. Such costly behaviours resulted in considerably reduced profits (NEFMC 2006).

As each year's measures failed to curtail overfishing, and new ones were necessitated, social conflict and mistrust of the management agency arose. Once again, the many input restrictions forced harvesters to undermine their own cost-effectiveness, spending more money to fish harder within the days allowed (Acheson & Gardner, 2011). Harvesters were frustrated that costly restrictions that were still ineffective at rebuilding stocks or increasing catch per unit effort, to the extent that when interviewed in 2008, only 17% of interviewed groundfish harvesters would recommend that their children enter the industry (Acheson & Gardner, 2011). The consolidating effect of tradeable rights, combined with the continuing declines in days-at-sea, reduced the total number of active vessels from 515 in 2004 to 328 in 2009 (Walden, 2013). However, the vessels that exited the fishery faced uncertain economic futures regardless, and the tradeable rights system allowed them to be compensated for exiting by selling their days-at-sea. Furthermore, while total revenues fell by 21% in the first 2 years of the programme, the average vessel that remained saw an increase in revenue since fewer vessels were participating (NEFMC 2006; Walden, 2013).

5 | PATH 3: SPATIAL ACCESS CONTROLS

The third major approach to regulating fishery effort is to implement physical boundaries for controlling fishery resources and associated ecosystem services. A wide range of terms are used, including marine protected areas (MPAs), marine reserves, marine parks and closed areas, but they are all applied regardless of whether fishing access is allowed (de facto or de jure) within the established boundaries. As a fishery management tool, no-take areas exclude fishing effort in order to preserve habitat and stock to serve as a seed population, with the expectation that dispersal will supply a sustainable fishery in adjacent areas (Hilborn et al., 2004). In practice, the areas closed are typically prime fishing grounds because harvesters target areas of high abundance, and displaced harvesters shift their effort to the most profitable areas they may still access, a response that depends on the habitat and gear (Horta e Costa et al., 2013). When effectively communicated and enforced, ecological outcomes within well-designed and enforced no-take boundaries are typically positive, including increased size and abundance (Russ, Alcala, Maypa, Calumpong, & White, 2004). However, a recent meta-analysis showed that these improvements were most strongly predicted by increased staff and budget and were not without numerous examples of ineffective or inequitable management processes (Gill et al., 2017; Lester et al., 2009). For a static closure to improve stock status, it must be appropriately scaled to the life history of the target species; most no-take areas are best suited to benthic fisheries that are less mobile throughout their life history (Hilborn et al., 2004).

Spatial management that identifies areas for regulated fishing typically designates a group—often local fishermen or leaders—to design the management system governing access and harvest. As with no-take areas, the ecological effectiveness of regulated-take areas is constrained by the biology of the target organisms, but ecological success also requires that the governing body be capable of establishing and enforcing regulations that sustain the resource. In some cases, enabling legislation establishes TACs, creating a spatially explicit catch share system. In general, fishermen will respond to the incentives for catching the available fish, and approaches that limit effort and achieve healthy stocks when applied fisherywide can sustain a healthy stock within a biologically appropriate MPA, as described above.

Comparing spatial case-study outcomes to those from catch limit and effort limit approaches is challenging because spatial measures are often implemented with different objectives. First, spatial measures are often motivated primarily by preservation, or conservation interests drive project design even when fishery goals are articulated (e.g., Castrejón & Charles, 2013). Second, while management objectives typically envision fisheries as enhancing communities' wealth and employment by selling fish in an exchange economy (cf., Cunningham, Neiland, Arbuckle, & Bostock, 2009), spatial measures are often applied where implementers are more concerned about welfare concepts such as food security, equity of access and self-determination (cf., Béné et al., 2010). As a result, many spatial initiatives do not track the same economic and social measures as are typically used to assess other approaches' outcomes. This information gap fuels a vigorous debate among development scholars about whether poor fishing communities are best supported by developing local management institutions or providing market opportunities (Foale et al., 2013).

We focus on case-studies that include a primary goal related to sustaining commercial fisheries and that specifically seek to improve fishery outcomes by constraining fishing effort. Whether a healthy stock is supported through a closed area seeding an adjacent fishery, or through effective management of effort within a designated area, access to that stock can be achieved through harvest guidelines, limited access, TAC limits, catch share cooperatives, input regulations or transferable input rights. Since harvesters respond to the incentives present where they can fish, spatial management typically inherits the outcomes of those measures.

5.1 | Open access

In 1990, the Columbretes Islands in the Spanish Mediterranean established a no-take area to support lobster (*Palinurus elephas*, Palinuridae) breeding. By 1998, the benthic crustacean fishery saw an increase in lobster size within the reserve, paired with a 10% increase in fishery catch outside of the reserve, due to the local spillover effect of this low-mobility species (Goñi, Hilborn, Díaz, Mallol, & Adlerstein, 2010). However, several vessels exited the area when the closure occurred, and during the subsequent study period, the number of vessels fishing the boundary of the reserve dropped from three to two. Thus, while catch slightly increased, the loss of the high-density fishing ground reduced the number of vessels the fishery was able to support, underscoring that using spatial measures to address the biological tragedy of the commons does not necessarily resolve the economic tragedy of the commons associated with open access.

For example, the Galapagos Marine Preserve was designed in 1999 as a network of closed and multi-use (fishing) areas. Since the fishery was open to any Ecuadorian resident, the number of vessels increased from 795 to over 1,200 in the first year; areas were finally closed in 2006 (Castrejón & Charles, 2013). Poorly enforced TACs led to over-exploitation of key cucumber (*Isostichopus* sp., Stichopodidae) and spiny lobster (*Palinurus* sp.) stocks. A race-to-fish has emerged as fishermen are investing in more equipment and faster vessels, and are diving in deeper waters, leading to more cases of decompression sickness (Castrejón & Charles, 2013).

Spatial restrictions are also sometimes used to prevent conflict or implement allocative policies between large- and small-scale vessels. For example, Indonesia maintains waters within 12 miles of shore for vessels under 30 tonnes, and waters inside 4 miles of shore are reserved for vessels under 5 tonnes. Open access in each zone allowed vessels to enter, expanding the fleet 28% between 2002 and 2013 (Stobutzki, Stephan, & Mazur, 2014), and fully or over-exploiting most stocks. Although more than 2.4 million people participate in capture fishing, many artisanal fishermen are poor and food-insecure (Stobutzki et al., 2014). In both examples, spatial restrictions without limited entry or well-enforced TACs function like open access, where fishers lack an incentive to exit or reduce their effort until it is no longer profitable to fish.

5.2 | Limited entry

In 1991, the Chilean government established management and exploitation areas with local-only access, and governance, for artisanal dive fisheries targeting high-value benthic crustaceans, loco (*Concholepas concholepas*, Muricidae), urchin (*Loxechinus albus*, Parechinidae) and stone crab (*Metacarcinus edwardsii*, Cancridae). Limited access or no-take areas saw significant overall increases in the size and abundance of loco and urchin relative to open access areas (Castilla & Fernandez, 1998), with bigger differences in better enforced areas (Gelcich et al., 2012), but much smaller or no gains for the more mobile stone crabs, underscoring that the scale of spatial management must match individual range and other characteristics of the target species. A survey of 55 local associations indicated participants felt conservation goals had mostly been met, but despite the increased resource in no-take areas, a meaningful increase in aggregate harvest has not been realised (Gelcich et al., 2010). The median fisher receives only 20% of their income from limited access areas, and only 15% of participants across territories have identified economic benefits (Gelcich et al., 2017). While local management led to various governance and access arrangements, aggregate results reflect that widely adopted limited access and no-take areas improve stock health, but additional measures are required to generate economic gains. Similar concerns about food security persist amidst stock improvements in a six-nation network of closed areas in the Coral Triangle Initiative (Christie et al., 2016; Foale et al., 2013).

The importance of controlling catch or effort in conjunction with spatial approaches is often confounded because spatial methods are often applied in contexts where the potential for entry is limited, because little capital is available and the human population is small relative to the size of the resource. For example, the small-scale reef fishery in Apo Island, Philippines, implemented a no-take zone in 10% of the coral area for dive tourism, prohibited destructive methods and established local-only access to the area around the reserve. Biomass for the two target stocks increased substantially within 250 m of the reserve. With only 500 residents, 200 of whom fish, local-only access constrained entry enough that the CPUE of the hook fisheries increased 50% from 1998 to 2001 compared to years immediately preceding and concurrent with initial implementation of the MPAs (Russ et al., 2004).

Welfare objectives are primary in Samoa's community-based fishery management programme, which provides technical assistance from national entities to long-standing spatially explicit community tenure organizations. By working with the national government and its mandates, habitat-destroying fishing practices (e.g., dynamite fishing) and poisoning of fish were banned (King & Faasili, 1998). As a result of central support, communities increased enforcement of access limitations as well as other national regulations within the spatial management area and seafood consumption increased (Govan, 2011; Tiitii, Sharp, & Ah-Leong, 2014). However, no significant economic improvements or increases in income or market purchases were observed (Tiitii et al., 2014). As in the Philippines, this management is adequate to support local subsistence use, but may not be robust to changes in harvest technology or to harvesters shifting to sell into wider geographic markets (Cinner & McClanahan, 2006).

In the English Beam Trawl fishery, managers used a spatial restriction to limit fishing within 12 miles of the coast to vessels under 9 m in length and 221 kW in engine power, reducing gear conflicts between small and large vessels and addressing a 20-fold increase in total effort between the 1970s and 1990s. Some larger vessels reduced their engine sizes to stay inshore, to reduce fuel costs and transit time. License restrictions prevented new entry, but as with limited entry without a spatial component, incumbents substituted unregulated inputs, weakening control over fishing mortality and increasing harvesting costs (Pascoe & Robinson, 1998).

5.3 | Total allowable catch

In some instances, TAC limits are used in conjunction with spatial closures in order to better achieve the desired biological outcomes. In Japan, the snow crab (*Chionoecetes opilio*, Oregoniidae) fishery in Kyoto prefecture experienced a large decline in catch in the late 1970s. This decline was associated with increasing investment in harvest technology exerting pressure on stocks and high regulatory discard mortality of crabs in flatfish trawling activity by the same vessels after the crab season. Both permanent and seasonal closures were implemented in the early 1980s, and expanded in 1991, and a TAC limiting take outside the areas was introduced in 1997. This led to an 8,300% increase in revenue per day in 2001–2005 as compared to the pre-MPA period; this increase was not seen in nearby regions that did not use MPAs (Makino, 2008). By capping the total removals outside the closures and creating areas without fishing pressure for the crab populations to grow, the management system encouraged rebuilding populations, increased catch per unit effort and revenue per day.

5.4 | Using spatial approaches to limit growth overfishing

The US Atlantic scallop (*Placopecten magellanicus*, Pectinidae) fishery added rotating fishing areas into their management scheme in 2003, opening areas of several hundred square nautical miles only when the density of large scallops was sufficiently high, yielding a much higher price due to their size (Edwards, 2001). Prior to adding this spatial element, limited access vessels were losing money (Edwards, 2001). Vessels are allocated days-at-sea for fishing inside the management areas and separate days-at-sea for fishing outside the management areas. In combination with trip limits, there is a de facto quota that maintains the fishery below the harvest guideline (Olson, 2006). Although participants still race to the most productive patches within an area during openings, allowing the scallops to grow to the highest-valued market size has made this one of the most valuable fisheries in the United States, and vessels are extremely profitable (Georgianna, Lee, & Walden, 2017). Similarly, the Australian Bass Strait central zone scallop fishery utilizes spatial closures and spatially explicit quota systems to ensure that no particular area is overharvested (Australian Fisheries Management Authority, 2017) For slow-moving or sessile organisms such as the Atlantic and Australian scallops, limiting spatial access combined with effort or catch limits has facilitated strong biological outcomes and growth of individual organisms leading to positive economic outcomes as well.

6 | DISCUSSION

Historical attitudes towards fishing were that resources were available for access and that over-exploitation was impossible. While this may have been true with the technology and market conditions that prevailed for much of history, advances in harvesting technology such as mechanization and refrigeration have led to widespread over-exploitation in the absence of limitations on catch. There is building evidence that science-based management methods are reducing the incidence of biological overfishing (Worm et al., 2009), and in many cases, stocks are recovering (e.g., Murphy et al., 2015). This is largely the result of moving from management systems which less effectively restrict catch, towards the centre of Figure 1 to systems that enact biologically informed regulatory practices. But even in fisheries with good biological management, harvesters often still struggle to make money and support their communities (World Bank 2017).

Table 1 facilitates direct comparisons of the observed triple bottom line outcomes across management approaches. The left column relates the economic outcome to the behavioural change induced by the management approach. The behavioural changes observed in the fisheries reviewed above underscore that economic success requires more than selecting a target biomass which maximizes profit—often called B_{MEY} in the bioeconomic model—given a fixed industry-wide cost and market structure. Rather, effort management induces behavioural responses to the need to compete for fish, which change the economic and social structure of the industry in predictable ways, at any target biomass. With open access or harvest guidelines, summarized in the top two rows, there is no restriction on entry, and additional harvesters enter whenever there is profit to be made. As

TABLE 1 Summary of behavioural changes observed under each approach to effort management, with associated economic, ecological and community outcomes. Background shading indicates generally negative (red), mixed (yellow) or positive (green) outcomes; gradients reflect outcomes depend on other features of management [Colour in online version]

Management Approach	Economic	Ecological	Community
Open Access	Profit attracts entry into fishery until depleted stocks, saturated markets or costs of competitive fishing eliminates profit, the bioeconomic equilibrium.	Determined by stock pressure at point of zero profit. For valuable species, entry leads to overfishing. Can be sustainable for species with limited markets.	High employment during depletion phase, but low profits and wages mean poor jobs. Displacement and community disruption when stocks collapse.
Harvest Guidelines	Unenforced guidelines do not change Open Access outcomes.	Unenforced guidelines do not change Open Access outcomes.	Unenforced guidelines do not change Open Access outcomes
Limited Access	Incumbent fishermen invest in harvesting power to compete for more fish until profit is eliminated, leading to bioeconomic equilibrium.	Effort increases by permit holders lead to higher fishing pressure, depleting stocks, except for short-lived, highly fecund species	Low profits and wages mean poor jobs. Displacement and community disruption when stocks collapse.
Input Regulations	Fishermen increase unregulated inputs, capital stuffing, until profit is eliminated.	Capital stuffing increases effort and stock pressure.	Employment can increase if crew not regulated input, but low profit means poor jobs.
Tradable Input Regulations	Input rights shifted to those who can capital stuff most efficiently. They continue to do so until profit is eliminated.	Shifting input rights to more effective capital stuffers exacerbates resource depletion.	Crew displaced from selling vessels; adverse effects in communities whose residents sell; remaining jobs still low-paying.
Total Allowable Catch (TAC)	Fishermen invest to compete more effectively for fish, until profit is eliminated.	Correctly set and enforced TACs support sustainability.	Race-to-fish leads to seasonal, high-paying and dangerous jobs.
Catch Shares	Groups receiving collective allocations can coordinate rather than compete, reduce costs to and improve price.	Correctly set and enforced TACs support sustainability.	Stakeholders make tradeoff between number and quality of jobs, but non-members disenfranchised.
Individual Fishing Quota (IFQs)	Individually fixed catch quantity induces fishermen to maximize profit per fish by cutting costs, improving price.	Correctly set and enforced TACs support sustainability.	Less intensive fishing may reduce employment, but profits mean jobs are safer, better paying.
Individual Transferable Quota (ITQs)	Like IFQ, but additionally quota moves to more efficient vessels, increasing profitability of the fleet.	Correctly set and enforced TACs support sustainability.	Crew displaced from consolidation; remaining jobs safer, better paying; disproportionate adverse effects in communities whose residents sell.
Spatial Management	Behavioral response and economic outcomes from fishing zone determined by approach, above, in place there.	Closing areas calibrated to the life history of the species increases biomass within the area, and may create spillover to be caught.	Community outcomes from where fishing allowed determined by approach, above, in place there.

with the post-WWII New England groundfish fishery, entry drives the fishery towards bioeconomic equilibrium, unsustainable levels of effort, and low or no profits. Simply presenting scientific estimates of harvest guidelines, without regulations or enforcement to limit mortality, does not change these incentives, as efforts to manage Atlantic sailfish demonstrate. In these cases, declining stocks are unable to support those who entered when fish were more abundant, leading to community disruption. With a well-designed spatial component (bottom row of Table 1), stock collapse can be averted, but Galapagos Marine Reserve demonstrates that the economic tragedy of the commons remains.

Limited access (third row of Table 1) and input regulations (rows four and five) attempt to control the effect of increases in effort through new entry, but fail: as in the Bristol Bay salmon fishery, limiting entry still induces incumbents to try to increase their share of the competitive harvest by capital stuffing. However, these competition-driven investments in vessels increase costs, and any biological benefit arises because the point where additional investment is no longer profitable is reached at higher biomass. Eigaard et al.'s (2014) survey concludes that the biological success of effort-based management controls is highly dependent on the ability to anticipate input substitution and changes in catchability attributable to technological change. New England groundfish's efforts with limited access and days-at-sea demonstrate that sufficiently comprehensive controls are elusive, especially as transferability reallocated effort to those who could capital stuff more effectively, so that social and economic outcomes mirror those in unregulated fisheries.

When a hard total allowable catch is implemented (sixth row of Table 1), additional effort, through new entry or capital stuffing, does not result in additional mortality. As in British Columbia halibut, an appropriately set and enforced TAC ensures biological sustainability, even in the face of increasing capital. With a spatial component, when the scale matches the range of the target species' life history, stock health can be improved, as in Japanese snow crab. However, harvesters must race to fish to effectively compete for a share of the catch, so they invest in more capital, engage in risky fishing practices and erode value through poor handling and flooding markets; those who refrain from the derby will lose market share to those who participate. Harvesters invest until the additional profits from more investment are less than its cost: a bioeconomic equilibrium at the TAC, with a combination of higher costs, lower revenue, increased employment volatility and reduced safety.

The competition for fish that drives overcapitalization and value-dissipating behaviour can be eliminated by determining the allocation of TAC among fishermen through IFQs or ITQs (rows eight and nine of Table 1), or through a catch share programme (row seven). Unable to increase their catch through additional effort, harvesters instead focus on maximizing profit from their fixed allocation by reducing costs and maximizing value. This shift from choosing effort to capture more, or a greater share, of the fish means the bioeconomic model, and the prediction of bioeconomic equilibrium, no longer applies. Correctly set and enforced TACs ensure biological sustainability, and the focus on maximizing profits makes the fishery economically successful. This is exemplified by the British Columbia halibut harvesters' development of a fresh market that more than doubled the value of the product and dramatically increased profitability. Less intensive harvesting can affect the structure of the crew labour market, often reducing the number of short-season jobs in favour of fewer longer term, higher paying and safer jobs.

The Bering Sea crab fishery demonstrates that allowing the transfer of quota facilitates the compensated exit of high-cost operators, further enhancing fleetwide profitability. However, transferability also lets the market determine who benefits from fishing, which often leads to adverse social effects. Smaller, more isolated communities, which are often more dependent on fisheries, may have less well-capitalized harvesters and higher cost processors, leading quota to flow to other communities where operations are more profitable. Although the boat owners who are typically initially allocated quota sell voluntarily, their local crew do not have a choice in the sale decisions that erode an essential employment base within a community of people or of place. Further, the additional profits may accrue to active fishermen, as in Alaska halibut, or initial quota holders can choose to collect lease payments without fishing, as in Bering Sea crab.

Catch share programmes mitigate these adverse community outcomes by allocating quota to groups rather than individuals. Allocations to fishing groups provide incentives to coordinate on harvest effort and joint marketing as in Chignik, and catch timing and by-catch management as in the Rhode Island fluke sector. Such groups can also collaborate to manage "choke" species or by-catch species that limit the harvest of otherwise abundant species, by sharing information and establishing avoidance incentives (e.g., Holland & Jannot, 2012). Catch shares empower the community of incumbent harvesters, rather than markets, to determine allocations. Quota can be additionally allocated to non-harvester community groups, as in the Alaskan CDQ programme, to generate local fishery benefits, giving other stakeholders control over who fishes and receives fishery benefits.

These new tools allow participants to strike a locally acceptable balance between sustaining broad participation that provides high levels of social benefit, and providing high levels of economic benefits to participants. The collective experience in different management methods is that there is a trade-off, whose resolution depends both on the priorities of regulators and on the structure of the fisheries: social benefits accrue differently based on whether vessel owners, crew, processing owners and their workers reside in, and spend their income in, the fishing community of concern, and on where benefits are created in the supply chain (Branch et al., 2006).

Approaches that require limiting access, in particular, often engender controversy because designating a group that has the exclusive right to fish implies designating a group that does not. This is often minimized by granting access to all incumbents. However, in some fisheries, there are loosely invested people, or people who leverage access only when other resources (e.g., other fisheries or agriculture) are performing poorly. In areas with diverse employment opportunities, forcing harvesters to specialize within the portfolio of fisheries in which they have participated in the recent past increases their exposure to risks of biological and market variations in those fisheries (e.g., Kasperski & Holland, 2013). In poorly integrated economies, fisheries are sometimes viewed as an "employer of last resort" for coastal residents who need subsistence food or income in the event of crop failure or personal financial shocks, and limited access exchanges better outcomes for incumbent fishermen for a social safety net.

Community-based management approaches are not separately included in Figure 1, because community management refers to who has the right to regulate access and harvest, not the approaches to regulating effort analysed here. Regardless of how incentives are established, we expect a community-based management process selecting any of the effort controls discussed to observe the same outcomes. However, involving the regulated community in the management process may still improve outcomes through two mechanisms. First, involving the community in management draws on their expertise about the actual operations in the fishery, and provides managers a sense of what types of measures will meet management goals at least cost. This increases the legitimacy of management and increases compliance and effectiveness (Jentoft, Mccay, & Wilson, 1998; Kuperan & Sutinen, 1998). Second, communities may be able to work together to establish a political consensus around more effective management methods. Both the Chignik co-op and Rhode Island Fluke Sector, which achieved significant triple bottom line successes for their fisheries, were discontinued following disagreements by non-participating harvesters, reflecting the importance of community acceptance of even very effective measures.

The economic effectiveness of catch share programmes leads to a perverse argument that selling access and harvest rights makes it more difficult for new entrants to participate in the fishery. Fishing quota and permits are valued, like all assets, as the present discounted value of the stream of profits to which the permit provides access. Entering open access fisheries is free, but that right also has little value because open access fisheries generate little profit. The value of a limited entry permit in a fishery where limited entry is binding is the present discounted value of the profits from fishing in that fishery. The value of an individual quota share is the present discounted value of the profits from fishing that quota. Therefore, if entry is more expensive in individual quota-based fisheries, it is because they are more profitable, which is likely to be the case given the incentives for maximizing market value and minimizing costs established by individual allocation. In fact, quota which subdivides the fishery rents into more units may make participating in ownership more accessible, as younger fishermen can slowly acquire shares, increasing their desirability as crew, without purchasing a critical mass of capital to have a fully independent business. Auctioning these permits or quota allows the government to capture much of the fishery's rent, rather than harvesters; allocating permits to community leaders, rather than members of the fishing industry, allows local political processes to determine who benefits.

Over the last three decades, fisheries management has demonstrated the ability to attenuate overfishing and sustain stocks and global catches around 90 million tonnes (FAO, 2016). However, not all approaches to management lead to ecological success in all cases. Some are easily circumvented in most applications. Others may be effective for highly fecund species with a weak correlation between spawning stock size and the number of young (e.g., shrimp, forage fish), but not work with more structured stocks. Still others may sustain stocks in geographically isolated fishing communities, but not be robust to the pressures of globalization. Even among biologically effective approaches, methods differ in how they trade off harvesting at low cost and paying many fishermen, and how they distribute the benefits of fishing among industry and community stakeholders. How best to strike these balances in any fishery is ultimately not a question of science, but rather one of the politics: scientists' role is to advise the decision-makers recognized by civil society on the most likely outcomes of alternative approaches. Scientists should help decision-makers draw the correct lessons from data, models and past experiences. That different sustainable approaches support different suites of outcomes provide a powerful policy lever that enables policymakers to select the fishery benefits that best suit the needs and values of their stakeholder communities.

ACKNOWLEDGEMENTS

This research is the product of a FISH513 graduate seminar on "What works in fishery management?" It did not receive any specific grant from funding agencies in the public, commercial or not-for-profit sectors. The framework presented evolved over several years of lectures in Anderson's FISH230 and FISH561 courses at the University of Washington, and this paper benefitted considerably from the questions and suggestions of students in those courses.

ORCID

Christopher M. Anderson http://orcid.org/0000-0003-0472-7218

Qi Lee http://orcid.org/0000-0001-7346-5077

Maite Pons http://orcid.org/0000-0002-4127-7300

Martin C. Arostegui http://orcid.org/0000-0002-9313-9487

REFERENCES

Abbott, J. K., Garber-Yonts, B., & Wilen, J. E. (2010). Employment and remuneration effects of IFQs in the Bering Sea/Aleutian Islands Crab Fisheries. *Marine Resource Economics*, *25*(4), 333–354. https://doi.org/10.5950/0738-1360-25.4.333

Abbott, J., & Wilen, J. (2010). Voluntary cooperation in the commons? Evaluating the sea state program with reduced form and structural models. *Land Economics*, *86*(1), 131–154. https://doi.org/10.3368/le.86.1.131

Acheson, J. M., & Gardner, R. (2011). Modeling disaster: The failure of the management of the New England groundfish industry. *North American Journal of Fisheries Management*, *31*, 1005–1018. https://doi.org/10.1080/02755947.2011.635119

Agar, J. J., Stephen, J. A., & Strelcheck, A. (2014). The Gulf of Mexico Red Snapper IFQ Program: The first five years. *Marine Resource Economics*, *29*, 117–198. https://doi.org/10.1086/676825

Anderson, J. L., Anderson, C. M., Chu, J., Meredith, J., Asche, F., Sylvia, G., … Valderrama, D. (2015). The fishery performance indicators: A management tool for triple bottom line outcomes. *PLoS One*, *10*(5), e0122809. https://doi.org/10.1371/journal.pone.0122809

Australian Fisheries Management Authority (2017). *Bass strait central zone scallop fishery management arrangements booklet*. Retrieved from http://www.afma.gov.au/fisheries-services/fisheries-management-plans/

Barclay, K. (2010). Impacts of tuna industries on coastal communities in Pacific Island countries. *Marine Policy*, *34*(3), 406–413. https://doi.org/10.1016/j.marpol.2009.09.003

Belhabib, D., Sumaila, R., & Pauly, D. (2015). Feeding the poor: Contribution of West African fisheries to employment and food security. *Ocean and Coastal Management*, *111*, 72–81. https://doi.org/10.1016/j.ocecoaman.2015.04.010

Béné, C., Hersoug, B., & Allison, E. H. (2010). Not by rent alone: Analysing the pro-poor functions of small-scale fisheries in developing countries. *Development Policy Review*, *28*(3), 325–358.

Birkenbach, A. M., Kaczan, D. J., & Smith, M. D. (2017). Catch shares slow the race to fish. *Nature*, *544*, 223–226. https://doi.org/10.1038/nature21728

Boen, C., & Keithly, W. (2012). *Gulf of Mexico Red Snapper IFQ Program: Survey results and analysis*. Unpublished manuscript, Department of Agricultural Economics, Louisiana State University.

Branch, T. A. (2009). How do individual transferable quotas affect marine ecosystems? *Fish and Fisheries*, *10*, 39–57. https://doi.org/10.1111/j.1467-2979.2008.00294.x

Branch, T. A., Hilborn, R., Haynie, A. C., Fay, G., Flynn, L., Griffiths, J., … Young, M. (2006). Fleet dynamics and fishermen behavior: Lessons for fisheries managers. *Canadian Journal of Fisheries and Aquatic Sciences*, *63*, 1647–1668. https://doi.org/10.1139/f06-072

Brewer, J. F. (2011). Paper fish and policy conflict: Catch shares and ecosystem-based management in Maine's groundfishery. *Ecology and Society*, *16*(1), 15. https://doi.org/10.5751/ES-03765-160115

Brinson, A. A., Die, D. J., Bannerman, P. O., & Diatta, Y. (2009). Socioeconomic performance of West African fleets that target Atlantic billfish. *Fisheries Research*, *99*(1), 55–62. https://doi.org/10.1016/j.fishres.2009.04.010

Brinson, A. A., & Thunberg, E. M. (2016). Performance of federally managed catch share fisheries in the United States. *Fisheries Research*, *179*, 213–223. https://doi.org/10.1016/j.fishres.2016.03.008

Carothers, C. (2008). "Rationalized out": Discourses and realities of fisheries privatization in Kodiak, Alaska. *American Fisheries Society Symposium*, *68*, 55–74.

Carothers, C., Lew, D. K., & Sepez, J. (2010). Fishing rights and small communities: Alaska halibut IFQ transfer patterns. *Ocean and Coastal Management*, *53*, 518–523. https://doi.org/10.1016/j.ocecoaman.2010.04.014

Casey, K. E., Dewees, C. M., Turris, B. R., & Wilen, J. E. (1995). The effects of individual vessel quotas in the British Columbia Halibut Fishery. *Marine Resource Economics*, *10*(3), 211–230. https://doi.org/10.1086/mre.10.3.42629588

Castilla, J. C., & Fernandez, M. (1998). Small-scale benthic fisheries in Chile: On co-management and sustainable use of benthic invertebrates. *Ecological Applications*, *8*, S124–S132. https://doi.org/10.1890/1051-0761(1998)8[s124:sbfico]2.0.co;2

Castrejón, M., & Charles, A. (2013). Improving fisheries co-management through ecosystem-based spatial management: The Galapagos Marine Reserve. *Marine Policy*, *38*, 235–245. https://doi.org/10.1016/j.marpol.2012.05.040

Christie, P., Pietri, D. M., Stevenson, T. C., Pollnac, R., Knight, M., & White, A. T. (2016). Improving human and environmental conditions through the Coral Triangle Initiative : Progress and challenges. *Current Opinion in Environmental Sustainability*, *19*, 169–181. https://doi.org/10.1016/j.cosust.2016.03.002

Cinner, J. E., & McClanahan, T. R. (2006). Socioeconomic factors that lead to overfishing in small-scale coral reef fisheries of Papua New Guinea. *Environmental Conservation*, *33*(1), 73–80. https://doi.org/10.1017/S0376892906002748

Connor, R. (2001). Changes in fleet capacity and ownership of harvesting rights in New Zealand fisheries. In R. Shotton (Ed.), *Case studies on the effects of transferable fishing rights on fleet capacity and concentration of quota ownership – Changes in fleet capacity and ownership of harvesting rights in New Zealand fisheries*. Rome: FAO Fisheries Technical Paper No. 412. Retrieved from http://www.fao.org/docrep/005/y2498e/y2498e0e.htm

Costello, C., Ovando, D., Clavelle, T., Strauss, C. K., Hilborn, R., Melnychuk, M. C., … Leland, A. (2016). Global fishery prospects under contrasting management regimes. *Proceedings of the National Academy of Sciences of the United States of America*, *113*(18), 5125–5129. https://doi.org/10.1073/pnas.1520420113

Cunningham, S., Neiland, A. E., Arbuckle, M., & Bostock, T. (2009). Wealth-based fisheries management: Using fisheries wealth to orchestrate sound fisheries policy in practice. *Marine Resource Economics*, *24*(3), 271–287. https://doi.org/10.5950/0738-1360-24.3.271

Da Rocha, J.-M., Cerviño, S., & Villasante, S. (2012). The common fisheries policy: An enforcement problem. *Marine Policy*, *36*(6), 1309–1314. https://doi.org/10.1016/j.marpol.2012.02.025

De Alessi, M., & Warmbrunn, A. (2014). *Markets, moneylenders, and minimum sizes: Pathways to reform in blue swimming crab fishing communities of the Tiworo Strait, SE Sulawesi*. Report prepared for the Walton Family Foundation, 15pp.

De Alessi, M., & Warmbrunn, A. (2015). *Markets, moneylenders, and minimum sizes part II: Blue swimming crab fisheries in Southern Sumatra*. Report prepared for the Walton Family Foundation, 21pp.

Deacon, R. T., Parker, D. P., & Costello, C. (2008). Improving efficiency by assigning harvest rights to fishery cooperatives: Evidence from the Chignik Salmon Co-op. *Arizona Law Review*, *50*, 479–509. Retrieved from http://heinonline.org/HOL/Page?handle=hein.journals/arz50&id=487&div=26&collection=journals

Dean, C. (2006). Study sees "Global Collapse" of fish species. *The New York Times: Science*. November 3, 2006.

Dewees, C. M. (1998). Effects of individual quota systems on New Zealand and British Columbia fisheries. *Ecological Applications*, *8*, S133–S138. https://doi.org/10.1890/1051-0761(1998)8[s133:eoiqso]2.0.co;2

Dixon, C., Noell, C., & Hooper, G. (2013). *Spencer Gulf prawn Penaeus (Melicertus) latisulcatus fishery 2011/12: Fishery assessment report to PIRSA Fisheries and Aquaculture*. Retrieved from http://trove.nla.gov.au/work/178568791?selectedversion=NBD50725544

Edwards, S. F. (2001). Rent-seeking and property rights formation in the rent-seeking and property rights format in the U.S. Atlantic Sea scallop fishery. *Marine Resource Economics*, *16*(16), 263–275. https://doi.org/10.1086/mre.16.4.42629337

Eigaard, O. R., Marchal, P., Gislason, H., & Rijnsdorp, A. D. (2014). Technological development and fisheries management. *Reviews in Fisheries Science & Aquaculture*, *22*, 156–174. https://doi.org/10.1080/23308249.2014.899557

FAO (2016). *The State of World Fisheries and Aquaculture 2016: Contributing to food security and nutrition for all*. Rome. Retrieved from http://www.fao.org/3/a-i5555e.pdf

Färe, R., Grosskopf, S., & Walden, J. (2015). Productivity change and fleet restructuring after transition to individual transferable quota management. *Marine Policy*, *62*, 318–325. https://doi.org/10.1016/j.marpol.2015.05.015

Field, B. C. (2008). *Natural resource economics: An introduction*. Long Grove, IL: Waveland Press.

Fina, M. (2005). Rationalization of the Bering Sea and Aleutian Islands crab fisheries. *Marine Policy*, *29*(4), 311–322. https://doi.org/10.1016/j.marpol.2004.05.005

Foale, S., Adhuri, D., Aliño, P., Allison, E. H., Andrew, N., Cohen, P., … Weeratunge, N. (2013). Food security and the Coral Triangle Initiative. *Marine Policy*, *38*, 174–183. https://doi.org/10.1016/j.marpol.2012.05.033

Fraser, A. (1979). Limited entry: Experience of the British Columbia salmon fishery. *Journal of the Fisheries Research Board of Canada*, *36*(7), 754–763. https://doi.org/10.1139/f79-113

Geheb, K., Kalloch, S., Medard, M., Nyapendi, A. T., Lwenya, C., & Kyangwa, M. (2008). Nile perch and the hungry of Lake Victoria: Gender, status and food in an East African fishery. *Food Policy*, *33*, 85–98. https://doi.org/10.1016/j.foodpol.2007.06.001

Gelcich, S., Cinner, J., Donlan, C. J., Tapia-Lewin, S., Godoy, N., & Castilla, J. C. (2017). Fishers' perceptions on the Chilean coastal TURF system after two decades: Problems, benefits, and emerging needs. *Bulletin of Marine Science*, *93*, 53–67. https://doi.org/10.5343/bms.2015.1082

Gelcich, S., Fer Andez, M., Godoy, N., Canepa, A., Prado, L., & Castilla, J. C. (2012). Territorial user rights for fisheries as ancillary instruments for marine coastal conservation in Chile. *Conservation Biology*, *26*, 1005–1015. https://doi.org/10.1111/j.1523-1739.2012.01928.x

Gelcich, S., Hughes, T. P., Olsson, P., Folke, C., Defeo, O., Fernández, M., … Castilla, J. C. (2010). Navigating transformations in governance of Chilean marine coastal resources. *Proceedings of the National Academy of Sciences of the United States of America*, *107*, 16794–16799. https://doi.org/10.1073/pnas.1012021107

Georgianna, D., Lee, M.-Y., & Walden, J. (2017). Contrasting trends in the Northeast United States groundfish and scallop processing industries. *Marine Policy*, *85*, 100–106. https://doi.org/10.1016/j.marpol.2017.08.025

Gill, D. A., Mascia, M. B., Ahmadia, G. N., Glew, L., Lester, S. E., Barnes, M., … Fox, H. E. (2017). Capacity shortfalls hinder the performance of marine protected areas globally. *Nature*, *543*, 665–669. https://doi.org/10.1038/nature21708

Gilroy, H. L., Erikson, L. M., & Mactavish, K. A. (2011). *2011 Commercial and regulation changes*. *IPHC Report of assessment and research activities 2011*. Retrieved from http://www.iphc.int/publications/rara/2011/2011.33.2011commercialfisheryandregulationchange.pdf

Ginter, J. J. C. (1995). The Alaska community development quota fisheries management program. *Ocean and Coastal Management*, *28*, 147–163. https://doi.org/10.1016/0964-5691(95)00069-0

GMFMC (2013). *Red snapper individual fishing quota program 5-year review*. Retrieved from http://www.nmfs.noaa.gov/sfa/laws_policies/national_standards/documents/red-snapper-5-year-review.pdf

Goñi, R., Hilborn, R., Díaz, D., Mallol, S., & Adlerstein, S. (2010). Net contribution of spillover from a marine reserve to fishery catches. *Marine Ecology Progress Series*, *400*, 233–243. https://doi.org/10.3354/meps08419

Gordon, S. H. (1954). The economic theory of a common-property resource: The fishery. *Journal of Political Economy*, *62*(2), 124–142. https://doi.org/10.1086/257497

Govan, H. (2011). *Good coastal management practices in the Pacific: Experiences from the field*. *Sprep*. Apia, Samoa: SPREP.

Grainger, C. A., & Costello, C. (2016). Distributional effects of the transition to property rights for a common-pool resource. *Marine Resource Economics*, *31*(1), 1–26. https://doi.org/10.1086/684132

Hamid, A., Wardiatno, Y., Lumbanbatu, D., & Riani, E. (2016). Stock status and fisheries exploitation of blue swimming crab *Portunus pelagicus* (Linnaeus 1758) in Lasongko Bay, Central Buton, Indonesia. *Asian Fisheries Science*, *29*, 206–219.

Hardin, G. (1968). The tragedy of the commons. *Science*, *162*, 1243–1248. https://doi.org/10.2307/1724745

Havice, E. (2013). Rights-based management in the Western and Central Pacific Ocean tuna fishery: Economic and environmental change under the Vessel Day Scheme. *Marine Policy*, *42*, 259–267. https://doi.org/10.1016/j.marpol.2013.03.003

Haynie, A. C. (2014). Changing usage and value in the Western Alaska Community Development Quota (CDQ) program. *Fisheries Science*, *80*, 181–191. https://doi.org/10.1007/s12562-014-0723-0

Hilborn, R., Stokes, K., Maguire, J. J., Smith, T., Botsford, L. W., Mangel, M., … Walters, C. (2004). When can marine reserves improve fisheries management? *Ocean and Coastal Management*, *47*, 197–205. https://doi.org/10.1016/j.ocecoaman.2004.04.001

Holland, D. S. (2000). Fencing the Fisheries Commons: Regulatory Barbed Wire in the Alaskan Groundfish Fisheries. *Marine Resource Economics*, *15*, 141–149. https://doi.org/10.1086/mre.15.2.42629297

Holland, D. S., & Jannot, J. E. (2012). Bycatch risk pools for the US West Coast Groundfish Fishery. *Ecological Economics*, *78*, 132–147. https://doi.org/10.1016/j.ecolecon.2012.04.010

Holland, D. S., & Sutinen, J. G. (1999). An empirical model of fleet dynamics in New England trawl fisheries. *Canadian Journal of Fisheries and Aquatic Sciences*, *56*, 253–264. https://doi.org/10.1139/f98-169

Homans, F. R., & Wilen, J. E. (2005). Markets and rent dissipation in regulated open access fisheries. *Journal of Environmental Economics and Management*, *49*(2), 381–404. https://doi.org/10.1016/J.JEEM.2003.12.008

Horta e Costa, B., Batista, M. I., Goncalves, L., Erzini, K., Caselle, J. E., Cabral, H. N., & Goncalves, E. J. (2013). Fishers' behaviour in response to the implementation of a marine protected area. *PLoS One*, *8*(6), e65057. https://doi.org/10.1371/journal.pone.0065057

Huang, L., & Smith, M. D. (2014). The dynamic efficiency costs of common-pool resource exploitation. *American Economic Review*, *104*(12), 4071–4103. https://doi.org/10.1257/aer.104.12.4071

ICCAT (2017). Report of the 2016 Sailfish stock assessment. *Collective Volume of Scientific Papers. ICCAT SCRS*, 1–68.

Jentoft, S., Mccay, B. J., & Wilson, D. C. (1998). Social theory and fisheries co-management. *Marine Policy*, *22*(45), 423–436. https://doi.org/10.1016/S0308-597X(97)00040-7

Kasperski, S., & Holland, D. S. (2013). Income diversification and risk for fishermen. *Proceedings of the National Academy of Sciences of the United States of America*, *110*(6), 2076–2081. https://doi.org/10.1073/pnas.1212278110

Khalilian, S., Froese, R., Proelss, A., & Requate, T. (2010). Designed for failure: A critique of the common fisheries policy of the European Union. *Marine Policy*, *34*, 1178–1182. https://doi.org/10.1016/j.marpol.2010.04.001

King, M., & Faasili, U. (1998). A network of small, community-owned village fish reserves in Samoa. *PARKS*, *8*(2), 11–16.

Knapp, G. (2008). The Chignik Salmon Cooperative. In R. Townsend, R. Shotton, & H. Uchida (Eds.), *Case studies in fisheries self-governance*. Rome, Italy: Food and Agriculture Organization of the United Nations.

Kroetz, K., Sanchirico, J. N., & Lew, D. K. (2015). Efficiency costs of social objectives in tradable permit programs. *Journal of the Association of Environmental and Resource Economists*, *2*, 339–366. https://doi.org/10.1086/681646

Kulmala, S., Peltomäki, H., Lindroos, M., Söderkultalahti, P., & Kuikka, S. (2007). Individual transferable quotas in the Baltic Sea herring fishery: A socio-bioeconomic analysis. *Fisheries Research*, *84*, 368–377. https://doi.org/10.1016/j.fishres.2006.11.029

Kuperan, K., & Sutinen, J. G. (1998). Blue water crime: Deterrence, legitimacy, and compliance in fisheries. *Law & Society Review*, *32*(2), 309–337. https://doi.org/10.2307/827765

Kuriyama, P. T., Branch, T. A., Bellman, M. A., & Rutherford, K. (2016). Catch shares have not led to catch-quota balancing in two North American multispecies trawl fisheries. *Marine Policy*, *71*, 60–70. https://doi.org/10.1016/j.marpol.2016.05.010

Lester, S., Halpern, B., Grorud-Colvert, K., Lubchenco, J., Ruttenberg, B., Gaines, S., … Warner, R. (2009). Biological effects within no-take marine reserves: A global synthesis. *Marine Ecology Progress Series*, *384*, 33–46. https://doi.org/10.3354/meps08029

Liese, C., Smith, M. D., & Kramer, R. A. (2007). Open access in a spatially delineated artisanal fishery: The case of Minahasa, Indonesia. *Environment and Development Economics*, *12*, 123–143. https://doi.org/10.1017/S1355770X06003421

Makino, M. (2008). Marine protected areas for the snow crab bottom fishery off Kyoto Prefecture, Japan. In R. Townsend, R. Shotton, & H. Uchida (Eds.), *Case studies in fisheries self-governance* (FAO Fisher, pp. 335–348). Rome. Retrieved from http://www.fao.org/3/a-a1497e/a1497e19.pdf

McCay, B. J. (2004). ITQs and community: An essay on environmental governance. *Agricultural and Resource Economics Review*, *33*(02), 162–170. https://doi.org/10.1017/S1068280500005748

Mcilwain, K., & Hill, J. (2013). *Catch Shares In Action: United States Bering Sea and Aleutian Islands Crab Rationalization Program*. Environmental Defense Fund. Retrieved from http://fisherysolutionscenter.edf.org/sites/catchshares.edf.org/files/US_BSAI_Crab_Rationalization.pdf

Metzner, R., & Ward, J. M. (2002). *Report of the expert consultation on catalysing the transition away from overcapacity in marine capture fisheries*. Rome.

Ministry of Agriculture Forestry and Fisheries of Japan (2016). *FY2015 Trends in fisheries, FY 2016 Fisheries policy white paper on fisheries: Summary*. Retrieved from http://www.maff.go.jp/e/data/publish/attach/pdf/index-49.pdf

Murphy, T., Kitts, A., Demarest, C., & Walden, J. (2015). 2013 Final report on the performance of the Northeast multispecies (Groundfish) fishery. *Northeast Fisheries Science Center Reference*. Retrieved from https://www.nefsc.noaa.gov/read/socialsci/pdf/groundfish_report_fy2013.pdf

National Research Council (1999). *Sharing the fish: Toward a national policy on individual fishing quotas*. Washington, DC: National Academies Press. https://doi.org/10.17226/6335

NEFMC (2006). *Framework adjustment 42 to the northeast multispecies fishery management plan*. Retrieved from http://www.nero.noaa.gov/nero/hotnews/multifw42fr/

NEFMC & NMFS (2003). *Amendment 13 to the northeast multispecies fishery management plan*. Retrieved from http://archive.nefmc.org/nemulti/planamen/Amend%2013/Final%20Amendment%2013%20SEISVol.%20I-II.pdf

Njiru, M., Nzungi, P., Getabu, A., Wakwabi, E., Othina, A., Jembe, T., & Wekesa, S. (2006). Are fisheries management, measures in Lake Victoria successful? The case of Nile perch and Nile tilapia fishery. *African Journal of Ecology*, *45*(3), 315–323.

Njiru, M., Nzungi, P., Getabu, A., Wakwabi, E., Othina, A., Jembe, T., & Wekesa, S. (2007). Are fisheries management, measures in Lake Victoria successful? The case of Nile perch and Nile tilapia fishery. *African Journal of Ecology*, *45*(3), 315–323. https://doi.org/10.1111/j.1365-2028.2006.00712.x

NOAA (National Oceanic and Atmospheric Administration) (2017). NOAA catch share policy. *Oceans*, *1853*, 25. Retrieved from http://www.nmfs.noaa.gov/sfa/management/catch_shares/about/documents/noaa_cs_policy.pdf

Noell, C. J., & Hooper, G. E. (2015). *Spencer Gulf Prawn Penaeus (Melicertus) latisulcatus fishery 2013/14* Fishery assessment report to PIRSA fisheries and aquaculture. South Australian Research & Development Institute (Aquatic Sciences), Adelaide. SARDI Research Report Series no. 788. Retrieved from https://trove.nla.gov.au/work/198855540

Northeast Fisheries Science Center (2008). *Assessment of 19 northeast groundfish stocks through 2007: Report of 3rd groundfish assessment review meeting (GARM III)*. Woods Hole. Retrieved from https://www.nefsc.noaa.gov/publications/crd/crd0815/

NPFMC (2017). *BSAI Crab 10-year review ten-year program review for the crab rationalization management program in the Bering Sea/Aleutian Islands North Pacific Fishery Management Council*. Retrieved from https://www.npfmc.org/wp-content/PDFdocuments/catch_shares/Crab/Crab10yrReview_Final2017.pdf

O'Leary, B. C., Smart, J., Neale, F. C., Hawkins, J. P., Newman, S., Milman, A. C., & Roberts, C. M. (2011). Fisheries mismanagement. *Marine Pollution Bulletin*, *62*(12), 2642–2648. https://doi.org/10.1016/j.marpolbul.2011.09.032

Olson, J. (2006). Changing property, spatializing difference: The sea scallop fishery in New Bedford, Massachusetts 1. *Human Organization*, *65*(3), 307–318. https://doi.org/10.17730/humo.65.3.d87bu307bxbwdcw1

Panayotou, T., & Jetanavanich, S. (1987). *The economics and management of thai marine fisheries*. Manila, Philippines: International Center for Living Aquatic Resources Management.

Panetta, L. E. (2003). *America's Living Oceans: Charting a course for Sea Change*. Summary Report. Pew Oceans Commission. Retrieved from https://www.pewtrusts.org/~/media/assets/2003/06/02/poc_summary.pdf

Pascoe, S., & Robinson, C. (1998). Input controls, input substitution and profit maximisation in the English channel beam trawl fishery. *Journal of Agricultural Economics*, *49*, 16–33. https://doi.org/10.1111/j.1477-9552.1998.tb01249.x

Pfeiffer, L., & Gratz, T. (2016). The effect of rights-based fisheries management on risk taking and fishing safety. *Proceedings of the National Academy of Sciences of the United States of America*, *113*(10), 2615–2620. https://doi.org/10.1073/pnas.1509456113

Pinto da Silva, P., & Kitts, A. (2006). Collaborative fisheries management in the Northeast US: Emerging initiatives and future directions. *Marine Policy*, *30*(6), 832–841. https://doi.org/10.1016/j.marpol.2006.04.003

Pope, J. G. (2009). Input and output controls: The practice of fishing effort and catch management in responsible fisheries. In K. L. Cochrane, & S. M. Garcia (Eds.), *A fishery manager's guidebook* (2nd ed., pp. 220–252). Oxford, UK: Wiley-Blackwell. https://doi.org/10.1002/9781444316315.ch9

Russ, G. R., Alcala, A. C., Maypa, A. P., Calumpong, H. P., & White, A. T. (2004). Marine reserve benefits local fisheries. *Ecological Applications*, *14*(2), 597–606. https://doi.org/10.1890/03-5076

Saharuddin, A. H. (1995). Development and management of Malaysian marine fisheries Technical conservation measures. *Marine Policy*, *19*(2), 115–126. https://doi.org/10.1016/0308-597X(94)00011-G

Scheld, A. M., & Anderson, C. M. (2014). Market effects of catch share management: The case of New England multispecies groundfish. *ICES Journal of Marine Science*, *71*(7), 1835–1845. https://doi.org/10.1093/icesjms/fsu001

Scheld, A. M., Anderson, C. M., & Uchida, H. (2012). The economic effects of catch share management: The Rhode Island fluke sector pilot program. *Marine Resource Economics*, *27*(3), 203–228. https://doi.org/10.5950/0738-1360-27.3.203

Schelle, K., Iverson, K., Free-Sloan, N., & Carlson, S. (2004). *Bristol Bay Salmon drift gillnet fishery optimum number report*. Retrieved from https://www.cfec.state.ak.us/RESEARCH/salmon/XSUM_10_21_04.pdf

Scott, A. (1955). The fishery: The objectives of sole ownership. *Journal of Political Economy*, *63*(2), 116–124. https://doi.org/10.1086/257653

SEDAR (2013). *Stock assessment report: Gulf of Mexico Red Snapper*. North Charleston. Retrieved from http://sedarweb.org/sedar-31-stock-assessment-report-gulf-mexico-red-snapper

Stobutzki, I., Stephan, M., & Mazur, K. (2014). *Overview of Indonesia's capture fisheries, 2013*. Retrieved from http://aciar.gov.au/files/app5_indonesian_capture_fisheries.pdf

The Nature Conservancy (2008). *History of the U.S. West Coast Groundfish Fishery*.

Thunberg, E., Kitts, A., & Walden, J. (2007). A case study of New England groundfish fishing capacity reduction. In R. Curtis & D. Squires (Eds.), *Fisheries buybacks*. Ames, IA: Blackwell Publishing.

Tiitii, U., Sharp, M., & Ah-Leong, J. (2014). *Samoa socioeconomic fisheries survey report 2012/2013*. Noumea, New Caledonia: Secretariat of the Pacific Community (SPC).

Townsend, R. E. (1985). On capital-stuffing in regulated fisheries. *Land Economics*, *61*(2), 195. https://doi.org/10.2307/3145812

Tveteras, S., Paredes, C., & Pena-Torres, J. (2011). Individual vessel quotas in Peru: Stopping the race for anchovies. *Marine Resource Economics*, *26*(3), 225–232. https://doi.org/10.5950/0738-1360-26.3.225

Uchida, H., & Makino, M. (2008). Japanese coastal fishery co-management: An overview. *FAO Fisheries Technical Paper*, *504*(Table 1), 221.

Uchida, H., Uchida, E., Lee, J.-S., Ryu, J.-G., & Kim, D.-Y. (2010). Does self management in fisheries enhance profitability? Examination of Korea's Coastal. *Marine Resource Economics*, *25*(1), 37–59. https://doi.org/10.5950/0738-1360-25.1.37

Walden, J. B. (2013). Economic health of the northeast (U.S.) multispecies trawl fleet 1996–2010. *Fisheries Research*, *139*, 98–104. https://doi.org/10.1016/j.fishres.2012.10.002

Weber, M., & Gradwohl, J. (1995). *The wealth of oceans*. New York, NY: W.W. Norton.

World Bank (2017). *The Sunken billions revisited: Progress and challenges in global marine fisheries*. Washington, DC: The World Bank. https://doi.org/10.1596/978-1-4648-0919-4

Worm, B., Hilborn, R., Baum, J. K., Branch, T. A., Collie, J. S., Costello, C., … Zeller, D. (2009). Rebuilding Global Fisheries. *Science*, *324*(5940), 578–585. https://doi.org/10.2307/20544204

Yandle, T., & Dewees, C. M. (2008). Consolidation in an individual transferable quota regime: Lessons from New Zealand, 1986-1999. *Environmental Management*, *41*, 915–928. https://doi.org/10.1007/s00267-008-9081-y

Yeeting, A. D., Bush, S. R., Ram-Bidesi, V., & Bailey, M. (2016). Implications of new economic policy instruments for tuna management in the Western and Central Pacific. *Marine Policy*, *63*, 45–52. https://doi.org/10.1016/j.marpol.2015.10.003

Yu, H., & Yu, Y. (2008). Fishing capacity management in China: Theoretic and practical perspectives. *Marine Policy*, *32*, 351–359. https://doi.org/10.1016/j.marpol.2007.07.004

Zou, K. (2003). Sino-Japanese joint fishery management in the East China Sea. *Marine Policy*, *27*, 125–142. https://doi.org/10.1016/S0308-597X(02)00086-6

How to cite this article: Anderson CM, Krigbaum MJ, Arostegui MC, et al. How commercial fishing effort is managed. *Fish Fish*. 2018;00:1–18. https://doi.org/10.1111/faf.12339