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An Evaluation of Harvest Control Methods for Fishery Management

Owen R. Liu[a], Lennon R. Thomas[a,b], Michaela Clemence[a,b], Rod Fujita[c], Jacob P. Kritzer[d], Gavin McDonald[a,b], and Cody Szuwalski[a,b]

[a]Bren School of Environmental Science and Management, University of California, Santa Barbara, CA, USA; [b]Marine Science Institute, University of California, Santa Barbara, CA, USA; [c]Environmental Defense Fund, San Francisco, CA, USA; [d]Environmental Defense Fund, Boston, MA, USA

ABSTRACT
Fisheries managers seek to maintain sustainable fisheries production, but successful management often requires the pursuit of multiple biological, ecological, and socioeconomic objectives simultaneously. Fisheries managers must choose among a broad range of harvest control methods (HCMs) to meet management objectives. This review identifies strengths and weaknesses of eight HCMs and evaluates their ability to meet a multitude of common biological, ecological, and socioeconomic management objectives such as protecting spawning biomass, reducing bycatch, and sustaining fishers' profit. Evidence suggests that individual HCMs often fail to meet management objectives and may unintentionally create incentives to race to fish, discard catch and overcapitalize fishing operations. These limitations can be overcome by strategically combining multiple controls or incorporating rights-based and spatial management.

KEYWORDS
fishery management; harvest control; rights-based management; race to fish; discard

Introduction

Fisheries around the world struggle to implement effective fishery management. Sixty-three percent of assessed stocks across the world are in need of rebuilding (Worm et al., 2009), and evidence suggests that unassessed fisheries, contributing up to 80% of global catch, may be in even worse shape (Costello et al., 2012). Overfishing globally is a threat to food security, particularly in the developing world (Pauly et al., 2005). Even in the developed world, some have argued that the economic costs of fishing outweigh its benefits because of overfishing: current resource rents of global fisheries (revenues minus costs and subsidies) are estimated at -$13 billion annually (Sumaila et al., 2012).

At the same time, the majority of fisheries that are properly scientifically assessed and actively managed are rebuilding (Hilborn and Ovando, 2014). Appropriate, not necessarily more precautionary, fishery management is effective at promoting sustainable fisheries. Rebuilding global fisheries through proper management could lead to enormous gains in fish abundance, fishery yield, and profitability (Costello et al., 2012; Sumaila et al., 2012). Regardless of the state of fisheries, selecting an appropriate harvest controls (or suite of controls) is necessary to accomplish fishery management objectives.

Objectives for fisheries management are often poorly defined, and the specific factors that lead to management success remain elusive (Hilborn, 2007a; Cochrane, 2000). Managers are typically faced with the challenge of achieving multiple objectives simultaneously, and therefore, the challenges of fishery management lie in first setting appropriate objectives, and then choosing feasible harvest control methods (HCMs) that allow for the simultaneous pursuit of those multiple objectives (Sissenwine and Kirkley, 1980; Hilborn, 2007a).

All methods of fisheries harvest control focus around directly or indirectly limiting catch. Direct controls such as catch limits impose a maximum number or weight of fish that can be caught. Alternatively, effort controls that limit gear type, or temporal and spatial restrictions, are indirect in that they assume that the restriction put on fishers (the limitation of fishing exploitation rate) will lead to a large enough biomass of fish left in the water to support harvestable populations. Both direct and indirect harvest controls are used commonly around the world in a great diversity of small-scale to large and industrial fisheries.

There are many potential HCMs, but little practical guidance available on which HCMs are most appropriate to achieve different objectives, particularly in data poor environments where information available for management is scarce (Carruthers et al., 2014). Previous reviews have incorporated a comparison of different HCMs, but they have focused primarily on simple description and

CONTACT Owen R. Liu oliu@ucsb.edu Bren School of Environmental Science and Management, University of California, Santa Barbara, CA 93106, USA.

generalized effects (e.g., Pope, 2002; Sissenwine and Kirkley, 1980), the problems created by fishers' response to fishing restrictions (Branch and Hilborn 2006), the scientific assessment of fish stocks (Deroba and Bence, 2008), or the effects of different HCMs on exploitation rate alone (Worm et al., 2009). This review expands that literature by offering a comprehensive look across a range of HCMs and objectives to illuminate tradeoffs and identify synergies in combining harvest controls to mitigate weaknesses of using them individually. The following sections first define specific biological, ecological, and socioeconomic management objectives, and then evaluate the efficacy of eight HCMs that are commonly used in fisheries management in efforts to achieve a suite of objectives.

A primary finding is that single HCMs have limitations in their ability to achieve multiple objectives simultaneously. Thus, a critical task is to determine the common limitations and unintended consequences that arise with HCMs and investigate situations where those drawbacks have been alleviated. The latter part of the review identifies cases where the combination of different HCMs helps to achieve multiple objectives, and discuss the ability of incentive-based approaches and spatial management to help alleviate the problems of discarding, uncertainty, and the impacts of fisher behavioral responses.

Objectives of harvest control methods

The first step in successful fisheries management is to clearly define and prioritize management objectives (Barber and Taylor, 1990, Beddington et al., 2007). This section describes a set of common biological, ecological and economic objectives of fisheries management used to compare and contrast HCMs (Table 1).

Biological objectives

Fisheries managers are tasked with the design and implementation of strategies to ensure sustainable production. Indeed, the FAO Code of Conduct for Responsible Fisheries advises that, "states should prevent over fishing and excess fishing capacity and should implement management measures to ensure that fishing effort is commensurate with the productive capacity of the fishery resources and their sustainable utilization" (FAO, 1995). Preservation of spawning stock biomass (SSB) is often the central objective that managers define in pursuit of sustainable stock productivity. Sufficient biomass must be protected in order to avoid recruitment failure and sustain fishing yields. Moreover, for many species, large, old, fecund females can disproportionately contribute to the reproductive output of a population (the BOFFFF hypothesis; Berkeley et al., 2004). Fisheries by their nature often preferentially target larger fish, and over time fishing truncates the 'natural' age structure of the fished population (i.e. that which would emerge in the absence of fishing), which can have negative consequences beyond any reduction in SSB. First noticed by Murphy (1967) in the Pacific sardine fishery, accumulating evidence of the effects of age truncation include elevated variability in population abundance, increased sensitivity to climate change, and a reduction in species' "bet-hedging" capability to withstand years of poor recruitment (Froese, 2004; Berkeley et al., 2004; Hsieh et al., 2006; Hsieh et al., 2010). Therefore, constraining the negative impacts of fishing on age structure may be an additional management objective beyond simple preservation of SSB (Berkeley et al., 2004; Birkeland and Dayton, 2005; Hsieh et al., 2010). Some HCMs may be designed to disproportionately protect smaller fish as well, in systems where growth overfishing is occurring (Waters and Huntsman 1986; Hill, 1992).

Management controls can also be selected to protect essential species behaviors or life cycle stages that are more vulnerable to fishing. Spawning aggregations of species such as orange roughy or Nassau grouper (Bax et al., 2005, Sala et al., 2001), synchronous migrations like those of Pacific salmon (Cooke et al., 2012), and schooling behavior as in many forage fish (Radovich, 1982) are all important behaviors that can be disrupted by fishing activity, thereby threatening fishery sustainability (Sadovy and Domeier, 2005; Nemeth, 2005). For example, fishing on a camouflage grouper spawning aggregation in just one year caused a decline in female size and a decline in mean female age of 3 years in Micronesia (Rhodes et al., 2011). Dean et al. (2012) documented the complete dispersal of an Atlantic cod spawning aggregation caused by a gillnet fishery. McQuinn (1997) suggested that productivity and maintenance of metapopulation structure in Atlantic herring is partly due to the social

Table 1. List of common fishery management objectives and considerations used to evaluate HCMs.

Biological objectives	Ecological objectives	Socioeconomic objectives	Costs and feasibility of implementation
Protect spawning stock biomass	Protect essential habitat	Increase fishing efficiency	Cost of implementation
Limit fishery truncation of age structure	Decrease bycatch	Increase fishing safety	Ease of enforcement
Protect essential behaviors		Increase product quality	Data requirements
		Increase fishing profits	

transmission of behaviors from older to younger fish, and that the removal or overfishing of particular contingents can threaten population resilience. All of these examples suggest that protection of behavior can be an important consideration for fishery managers.

Ecological objectives

The importance of considering potential impacts on ecosystems when managing fisheries is well documented (e.g., Sainsbury and Sumaila, 2003; Pikitch et al., 2004; Polovina, 2002; Hall and Mainprize, 2005). The maintenance of spawning stock biomass can be an ecological as well as a biological objective, if overfishing a given species affects system-wide productivity or trophic dynamics (Garrison and Link, 2000). In multispecies fisheries, unwanted bycatch of protected or threatened species, or co-occurring species with prohibitively low catch limits, can constrain or eliminate yield of target species, so management controls may be selected on their ability to control the number of nontarget species in the catch. For example, in the Eastern Bering Sea, management agencies have enacted specific regulations to minimize bycatch of co-occurring species in the groundfish fishery, primarily to balance fishing opportunities and maximize sustainable yield between fishers primarily targeting different species (Witherell and Pautzke, 1997). In addition, habitats damaged by fishing activity can compromise fishery yields (Turner et al., 1999), and an objective could be to safeguard habitat integrity essential to ecosystem or target species' productivity.

Socioeconomic objectives

Management strategies can also be formulated to achieve socioeconomic objectives related to the efficiency and operation of the fishery itself (Hilborn, 2007a). Managers and fishers can strive to maximize different types of efficiency such as economic efficiency (maximum yield for minimum cost, Dichmont et al., 2010) or technical efficiency (maximum catch relative to capital inputs, Kirkley et al., 1998). Managers often wish to promote fisher profits (e.g., Mardle et al., 2002), fishing safety (e.g., Hughes and Woodley, 2007; Kaplan and Kite-Powell, 2000), and increased value through better product quality (Anderson, 1989; Carroll et al., 2001). Multiple objectives are common in fisheries—managers and fishermen want high yields, good profits, and stable jobs which depend on achieving conservation objectives such as protecting spawning stock biomass, age structure, and ecosystem productivity.

Costs and feasibility of implementation

Finally, the costs related to the design and implementation of an HCM must be considered along with the potential benefits (Anderson, 1989; Arnason et al., 2000). This is particularly salient in fisheries that generate relatively low revenues (but may be important for other reasons, e.g., food security or livelihoods). Ongoing costs related to monitoring, enforcement and assessment of the resource vary significantly among HCMs (Anderson, 1989; Arnason et al., 2000; Wallis and Flaaten, 2003). Calculation of sustainable catches usually requires scientific fishery stock assessment, which can be technically challenging, subject to uncertainty, and costly (Walters and Maguire, 1996, Fulton et al., 2011). Some methods for analyzing stock status in data-limited situations are less technically onerous, but can be sensitive to uncertain inputs (Carruthers et al., 2014). Furthermore, HCMs vary widely in their costs of effective monitoring and enforcement (Anderson, 1989). In fact, considerations relating to the costs of harvest controls often dominate decision-making in implementation, rather than careful consideration of desired biological outcomes (Beddington and Rettig, 1983; Bunnefeld et al., 2011). Therefore, costs and feasibility must be considered at the outset of HCM design.

Harvest control methods

Fishery managers develop specific policies, or harvest strategies, to achieve the objectives detailed above. Harvest strategies are the framework and rules under which harvest is conducted, and are often operationalized relative to biological reference points. For example, a harvest strategy could be developed to maintain the fished stock at or above B_{MSY}, the biomass level required for maximum sustainable yield. Alternatively, a harvest strategy could attempt to maintain a sustainable fishing mortality rate (F_{MSY}), or ensure an adequate "floor" biomass through escapement (e.g., in the Pacific herring fishery, Hall et al., 1988).

Regardless of the harvest strategy, specific regulations or tactics (rules that fishers are required to follow) must be developed to control fishing, which in this review are termed HCMs. Eight common HCMs are the focus of this review (Table 2). The HCMs fall into two general categories, and are defined here.

Output controls

Output controls are direct limits on the number or weight of fish caught by a fishery. Fishery stock assessments are used to calculate how many fish can be

Table 2. Description of common HCMs in fisheries included in this article. HCMs are organized by example harvest strategy and type.

Example harvest strategy	Type of HCM	Method	Description
Maintain B_{MSY} | Preserve target SSB	Output control	Catch limit	Sets an upper limit on how many fish can be removed by a fishery in a given time
		Escapement threshold	Allows a certain number of fish to escape a fishery before harvest
		Bag or trip limit	Limits the number of fish that can be landed by an individual fisher or vessel on a single day or fishing trip
		Size limit	Sets minimum and/or maximum bounds on the size of fish that can be legally landed in a fishery
		Sex-specific limit	Similar to catch limits, but broken down by sex within a target species
Fish at F_{MSY}	Input control	Temporal limit	Restricts the time period over which a fish can be legally landed
		Gear/ vessel restrictions	Restricts the dimensions and characteristics of a gear or vessel allowed to participate in a fishery. May also restrict the quantity of gears allowed
		Deployment limit	Places a cap on the individual fishers' use of fixed gears

sustainably removed from the water, and the output control is based on that calculation (Hilborn and Walters, 1992). On the surface, output controls are straightforward, but their implementation can be complicated, because the controls' effectiveness depends upon the ability of managers to carefully monitor not just landings but total catch (landings plus discards) and incorporate those data into robust models (Pope, 2002). If, for example, unwanted catch is discarded dead, and not monitored and accounted for, there will be a large discrepancy between measured and actual fishing mortality (Gillis et al., 1995). Because output controls constrain allowable catch directly, they can result in attempts to maximize catch quality, or high-grade.

Catch limits

Catch limits are the most direct way to limit harvest to desired levels (FAO, 2003), by setting a Total Allowable Catch (TAC) for how many individuals or how much weight of fish can be removed by a fishery in a given time period. A TAC is normally set through a stock assessment process and sometimes includes an uncertainty buffer, which reduces the TAC by some fraction to account for both scientific and management uncertainty and reduce the risk of severely exceeding the TAC (Restrepo and Powers, 1999). Catch is monitored in the fishery through a combination of fisher self-reporting, third party fishing observers, and/or dockside monitoring of landings, and once the TAC is reached (or exceeded), the fishery is shut down for the remainder of the fishing period.

Escapement thresholds

Escapement thresholds mandate a minimum release or protection of fish (biomass or numbers) from fishing mortality, before harvest is permitted on any surplus (e.g., in Alaskan salmon fisheries, Alaska Department of Fish and Game, 2001; or herring fisheries, Hall et al., 1988). Fishers are regulated by the number of fish left in the water to ensure that a sufficient number remain for reproduction and ecosystem functions. An escapement threshold is the only method that directly targets this desired outcome, rather than achieving it indirectly via constraints on catch or effort. Escapement thresholds are most applicable in the relatively few fisheries that are based on migratory species with predictable temporal and spatial patterns (e.g., diadromous species), and escapement thresholds have only been adopted in relatively few of those.

Escapement targets can also be viewed as a higher level harvest strategy (e.g., Hall et al., 1988), and pursued with other HCMs described below, such as temporal limits or mesh size restrictions. For example, in the state of Maine, annual river herring escapement targets are achieved by restricting harvest to four days a week, along with gear restrictions (ASMFC, 2010). For the purposes of this review, however, escapement thresholds are considered a HCM to facilitate its comparison with other methods.

Bag or trip limits

Bag or trip limits are derivatives of fishery-wide catch limits, and limit the amount of fish one fisher can catch in a given time period. Bag limits are used most often in recreational or sport fisheries, e.g., an allowance of three legal-sized fish per fisher per day (Woodward and Griffin, 2003; Cox et al., 2002). Trip limits are based on an overall catch target for a time period divided by the number of vessels in a fishery and the number of expected trips per fisher. Trip limits are often used in an attempt to keep a fishery open all year (Pikitch and Wallace, 1988), by indirectly attempting to spread a catch limit over the length of a season.

Size limits

Minimum and maximum size limits set bounds on the size of a given species that can be legally landed by the fishery. Size limits are often implemented in an attempt to protect certain life stages of target species, on the theory that fishing mortality on these life stages may be disproportionately constraining stock productivity. A minimum size limit can be set above a species' size at maturity in order to allow most fish to spawn at least once. A maximum size limit can be set in order to preserve large, mature fish, termed "megaspawners", who in general have a disproportionately large role in reproduction and stock productivity (Berkeley et al., 2004). Size limits do not explicitly place a cap on total catch, but rather impose a restriction on the selectivity of the fishery, controlling the size, not number, of fish that are removed from the water. The biomass protected by size limits will be comprised of all fish below and/or above the size limit, and therefore will fluctuate with recruitment. Many fisheries employ size limits because of their simplicity and ease of enforcement (Anderson, 1989).

Sex-specific limits

Similarly to size limits, sex-specific limits place a restriction on the composition of the catch of a fishery, by managing the catch of mature individuals of each sex separately (usually by placing lower limits on reproductive females). By placing restrictions or bans on the capture of reproductively active individuals, sex-specific controls attempt to actively preserve spawning capacity (Zhou et al., 2010). Like size limits, the biomass protected by sex-specific limits will depend upon recruitment patterns. Sex-specific controls are most applicable in fisheries where distinguishing males from females is simple, e.g., in many crustacean fisheries. The catch of a certain sex can be regulated with numerical catch limits or simply banned (such as the "v-notching" policy and ban on the take of proven reproductive females in the American Lobster fishery, Daniel et al., 1989).

Input controls

In contrast to output controls, input controls restrict elements of the fishing operation itself as opposed to constraining catch. Input controls are based on the theory that restricting how or when fish can be caught will translate into a sustainable level of fishing mortality. In reality, input controls often reduce the efficiency (catch per unit cost or time) of fishing, in an attempt to curb longer-term negative impacts on the stock (Pope, 2002). Limiting the methods of fishing instead of the catch strongly incentivizes fishers to change their fishing technology and behaviors to improve efficiency or skirt the regulations (Branch and Hilborn, 2006). Implications of these incentives are discussed in section "Common outcomes of individual harvest control methods."

Temporal limits

Temporal limits manage harvest through a cap on the total number of days a fishery is open (Gulland, 1974; Sissenwine and Kirkley, 1980). The aim of a temporal limit is to constrain total harvest and effort occurring in the fishery (Sissenwine and Kirkley, 1980; Beddington et al., 2007), which relies on knowledge of the relationship between catch and effort. Individual fishers or vessels may be restricted temporally by having a set number of "days-at-sea" that they are allowed to fish, with the idea that the number of fishers multiplied by their days-at-sea will approximate a sustainable level of catch. Similarly, fishery managers may predict how long it will take for a fishery to meet a certain harvest level and based on this prediction, open a fishery for a set number of days or months (a season). A special case of temporal limits is a seasonal closure, often designed to protect spawning or other important behaviors (e.g., Beets and Friedlander, 1998). Because in general a seasonal closures target a behavioral rather than a fishing mortality objective, they are distinct from other types of temporal limits and are less likely to help meet biomass targets.

Gear/vessel restrictions

Gear/vessel restrictions place limits on the dimensions or type of vessel or gear allowed in a fishery, thereby restricting the efficiency and harvesting capacity of fishers (Metzner, 2005; Branch and Hilborn, 2006; McClanahan and Mangi, 2004). Gear restrictions can include gear bans, such as prohibiting destructive gear that impacts the seafloor, or gear modifications such as limiting mesh size or limiting the size of hooks in a line fishery. Vessel restrictions are used to restrict the dimensions or other attributes of vessels that can participate in a fishery (length, holding capacity, engine size, speed etc.). Effective gear and vessel restrictions require that all inputs be regulated, to avoid the incentive for fishers to increase fishing power in response to a constraint on one element of inputs by increasing their investment in other, less regulated inputs.

Deployment limits

In a fixed gear fishery, such as a gillnet, longline, or trap fishery, deployment limits are designed to limit overall fishing effort by placing a cap on individual fishers' use of gear. Deployment limits can include, for example, restrictions on the number of traps a fisher can deploy or a restriction on the number of hooks on a longline or the number of gillnets that can be set (e.g., Miller, 1976; Briand et al., 2004; Acheson 2001). For the purpose of this study, deployment limits refer to a cap on the amount of a certain gear that be fished, while gear restrictions refers to regulations on the type and design of the gear itself.

Common outcomes of individual harvest control methods

The ultimate outcomes of HCMs are determined not only by their technical design and implementation, but also by how the incentives of fishers change when a new regulation is instituted. Common potential positive and negative impacts of each HCM on the objectives introduced in section "Objectives of harvest control methods" are described below and outlined in Table 3. Example references are listed in Table 4.

The primary observation evident in Table 3 is that no single HCM can accomplish all management objectives, and rarely accomplish more than two or three objectives. In fact, many HCMs can result in negative impacts on objectives due to unanticipated changes in fisher behavior or technology. Therefore, once fisheries management objectives are articulated, managers should critically evaluate the expected strengths and limitations of HCMs to identify the most appropriate or combination of most appropriate methods to meet their objectives.

In practice, most fisheries are managed by more than one HCM, although many data-poor or developing world fisheries are managed only with a size limit or simple gear restrictions, if they are managed at all (e.g., McClanahan and Mangi, 2004), but to determine effects of HCMs on the objectives described above, it is illuminating to focus on each HCM individually. Moreover, even though fisheries are often managed with a suite of HCMs, the HCMs are not always combined strategically, and instead constitute a "band-aid" approach where increasingly more regulation is instituted in reaction to negative outcomes (Hilborn et al., 2004). The issue of the strategic combination of HCMs is considered in section "Combining harvest control methods."

Common positive outcomes of individual HCMs

Individual HCMs can have positive effects on biological objectives. Catch limits and escapement limits directly promote a sustainable spawning stock biomass (Cochrane and Garcia, 2009). They are purposefully designed to ensure that an adequate biomass of fish remain each year to sustain their population and the fishery. Size limits do not have an explicit effect on total catch, but instead impose management control on the composition of the catch (e.g., protecting all juveniles from fishing pressure). Gear restrictions can have a similar effect if they are designed around protecting a certain size class of fish (McClanahan and Mangi, 2004; Catchpole et al., 2005). Poorly designed size limits and/or overfishing of large, productive megaspawners as a result of restrictions on the harvest of smaller fish can have serious long-term implications for age/size structure of targeted fish populations, leading to declines in fish populations and fishery health (Berkeley et al., 2004; Conover and Munch, 2002; Fenberg and Roy, 2008).

If the protection of certain critical behaviors is a management objective, escapement limits and temporal limits (of the short-term, spawning closure type) can have beneficial effects, depending on the behavior of concern. For example, in an escapement management strategy in Alaska's Kuskokwim River, subsistence harvest by indigenous peoples is allowed during four consecutive days per week, followed by three days of unimpeded passage, to directly protect spawning biomass and the migration behavior itself (Linderman and Bergstrom, 2009).

Gear restrictions are the primary way through which the ecological objectives of habitat protection and/or control of bycatch are pursued in many fisheries (Isaksen et al., 1992; Roberts et al., 2005; Graham et al., 2007). Special cases of temporal limits can also work to reduce the bycatch in a fishery, if a fishing season (or closed season) is designed around a time of year when bycatch rates are known to be high (Dunn et al., 2011). In some fisheries, seasonal closures have been applied with the primary objective of reducing fisheries interaction with bycatch species, with varying success (Hood et al., 2007;

Table 3. Common HCMs and potential positive (plus sign), or negative (minus sign) effects on the defined biological, ecological, and socioeconomic objectives from Table 1, and implementation considerations based on a review of the available literature.

		Biological objectives			Ecological objectives		Socioeconomic objectives				Ease of implementation	
	Harvest control method	Protect SSB	Limit fishery truncation of age-structure	Protect behavior	Protect habitat	Decrease bycatch	Increase profits	Increase product quality	Increase efficiency	Fish safely	Cost of design and implementation	Cost of monitoring and enforcement
Output controls	Catch limits	O	–		–	–	–	–	O	–	–	O
	Bag or trip limits	O	–		–	–	–	+	O	–	–	–
	Escapement thresholds	+	O	+								
	Size limit	O	O			–	O	+	–		+	+
	Sex-specific restriction	O	O			–		–	–			
Input controls	Temporal limit	+	O	+		O	–	–	–	–	O	+
	Gear/vessel restrictions	O	O		+	+	–	+	–	–	O	+
	Deployment limit	+				+	+		O			

Circles indicate that the HCM has been observed to have positive or negative effects, depending on context. Grey indicates that the HCM is not expected to have an effect on the objective, and black indicates a lack of data in the literature. The table represents impacts of HCMs on objectives that are well documented in the literature and is not designed to represent or include the impacts of HCMs on management objectives in every potential scenario. Example references for this table are found in Table 4.

Table 4. Example references for the construction of Table 3.

	Harvest control method	Biological objectives: Protect SSB	Biological objectives: Limit fishery truncation ofage-structure	Biological objectives: Protect behavior	Ecological objectives: Protect habitat	Ecological objectives: Decrease bycatch	Socioeconomic objectives: Increase profits	Socioeconomic objectives: Increase product quality	Socioeconomic objectives: Increase efficiency	Socioeconomic objectives: Fish safely	Ease of implementation: Cost of design and implementation	Ease of implementation: Cost of monitoring and enforcement
Output controls	Catch limits	Pope, 2002 Cochrane and Garcia, 2009 Sissenwine and Kirkley, 1980	Sissenwine and Kirkley, 1980		Holland, 2007	Sissenwine and Kirkley, 1980 Ault et al., 2005 Vester gaard, 1996	Branch and Hilborn, 2006	Anderson, 1989 Waters, 1991	Branch and Hilborn, 2006 Anderson, 1989 Metzner, 2005	Coleman et al., 2004 Kaplan and Kite-Powell, 2000 Hanna and Smith, 1993	Sissenwine and Kirkley, 1980	Anderson, 1989 Sissenwine and Kirkley, 1980 Beddington et al., 2007
	Bag or trip limits	Woodward and Griffin, 2003 Cox et al., 2002 Pikitch and Wallace, 1988	Pikitch and Wallace, 1988		Fujita and Bonzon, 2005	Woodward and Griffin, 2003 Pikitch and Wallace, 1988 Gillis et al., 1995 Ault et al., 2005	Branch and Hilborn, 2006 Woodward and Griffin, 2003 Sissenwine and Kirkley, 1980 Waters, 1991		Branch and Hilborn, 2006 Cox et al., 2002 Woodward and Griffin, 2002 Pikitch and Wallace, 1988	Coleman et al., 2004 Hanna and Smith, 1993 Kaplan and Kite-Powell, 2000		
	Escapement thresholds	Cochrane and Garcia, 2009	Shaul et al., 2007 Morita and Fukuwaka, 2007	Linderman and Bergstrom, 2009								
	Size limit	Woodward and Griffin, 2003 Hennessey and Healey, 2000	Berkeley et al., 2004 Zhou et al., 2010 Fenberg and Roy, 2008 Hsieh et al., 2010			Alverson et al., 1994 Woodward and Griffin, 2003 Sissenwine and Kirkley, 1980 Ault et al., 2005	Woodward and Griffin, 2003 Waters, 1991	Donaldson and Donaldson, 1992	Woodward and Griffin, 2003 Waters, 1991	Rijnsdorp et al., 2008	Anderson, 1989	Anderson, 1989
	Sex-specific restriction	Zhou et al., 2010 Daniel et al., 1989 Miller, 1976	Zhou et al., 2010 Daniel et al., 1989	Daniel et al., 1989		Dew and McConnaughey, 2005						

Table 4. (*Continued*)

		Biological objectives			Ecological objectives		Socioeconomic objectives				Ease of implementation	
	Harvest control method	Protect SSB	Limit fishery truncation ofage-structure	Protect behavior	Protect habitat	Decrease bycatch	Increase profits	Increase product quality	Increase efficiency	Fish safely	Cost of design and implementation	Cost of monitoring and enforcement
Input controls	Temporal limit	Zhou et al., 2010 Sissenwine and Kirkley, 1980 Branch and Hilborn, 2006	Sissenwine and Kirkley, 1980 Berkeley et al., 2004	Sissenwine and Kirkley, 1980 Beets and Friedlander, 1998		Dunn et al., 2011 Hood et al., 2007 Murray et al., 2000 Catchpole and Gray, 2010 Vestergaard, 1996	Branch and Hilborn, 2006 Fulton et al., 2011 Sissenwine and Kirkley, 1980	Anderson, 1989 Branch et al., 2006 Waters, 1991	Branch and Hilborn, 2006 Fulton et al., 2011 Anderson, 1989 Sissenwine and Kirkley, 1980	Branch and Hilborn, 2006 Kaplan and Kite-Powell, 2000 Coleman et al., 2004	Anderson, 1989 Sissenwine and Kirkley, 1980	Sissenwine and Kirkley, 1980 Beddington et al., 2007
	Gear/vessel restrictions	McClanahan and Mangi, 2004 Catchpole et al., 2005 Sissenwine and Kirkley, 1980	Berkeley et al., 2004 Zhou et al., 2010 Fenberg and Roy, 2008		Roberts et al., 2005 Holland and Schnier, 2006 Bellman et al., 2005	Isaksen et al., 1992 Graham et al., 2007 Ault et al., 2005 Harrington et al., 2005	Branch and Hilborn, 2006 Waters, 1991	McClanahan, 2010	Branch and Hilborn, 2006 Anderson, 1989 Waters, 1991 Rijnsdorp et al., 2008 Metzner, 2005		Anderson, 1989 Sissenwine and Kirkley, 1980	Anderson, 1989 Sissenwine and Kirkley, 1980 Beddington et al., 2007
	Deployment limit	Ward and Luckhurst, 1991 Acheson and Acheson, 2010 Miller, 1976				Ward and Luckhurst, 1991	Briand et al., 2004 Acheson and Acheson, 2010		Acheson, 2001 Briand et al., 2004 Acheson and Acheson, 2010			

Dunn et al., 2011; Murray et al., 2000). At the extreme, some fisheries with active monitoring employ "real-time" closure systems in which fisheries in a given area are closed for a short time when the amount of unwanted bycatch exceeds a defined threshold (Catchpole and Gray, 2010).

While many HCMs can have negative expected socioeconomic outcomes in the short-term (discussed in the next section) depending on the context, size limits, bag limits, gear restrictions, and escapement thresholds can potentially increase product quality. For example, gear restrictions in a Kenyan artisanal fishery led to increased product quality and fisher incomes partially from the increased selectivity of the gear (McClanahan, 2010). Size limits can increase revenues if the value of the species is positively related to its size (e.g., in Alaskan crab fisheries, Donaldson and Donaldson, 1992). Importantly, these benefits can be short-lived if overfishing of larger-sized individuals affects stock age structure and abundance over time, and may be an artificial benefit if excessive and unmonitored discarding of individuals under the size limit is occurring and discard mortality is high. Wasteful discarding of this type occurred in the New England yellowtail flounder fishery before the institution of output controls (Alverson et al., 1994).

Common challenges of individual HCMs

Single HCMs can fail to accomplish many objectives and can even result in unintended, negative consequences (Table 3). Many HCMs, when applied individually, result in decreases in fishing efficiency and profits. An increase in bycatch is an additional major concern under catch limits, bag/trip limits, size limits, and sex-specific limits. Compliance with HCMs tends to be low and outcomes tend to be poor when governance is weak and incentives drive fishing inefficiency.

Both theoretical considerations and empirical studies of real fisheries suggest that negative biological, ecological, and social outcomes of HCMs arise because of fisher behavioral responses to HCM implementation (Salas and Gaertner, 2004; Branch and Hilborn, 2006; Fulton et al., 2011). Single HCMs often create incentives for fishers to race to fish (i.e., maximize short term catch at the expense of long-term

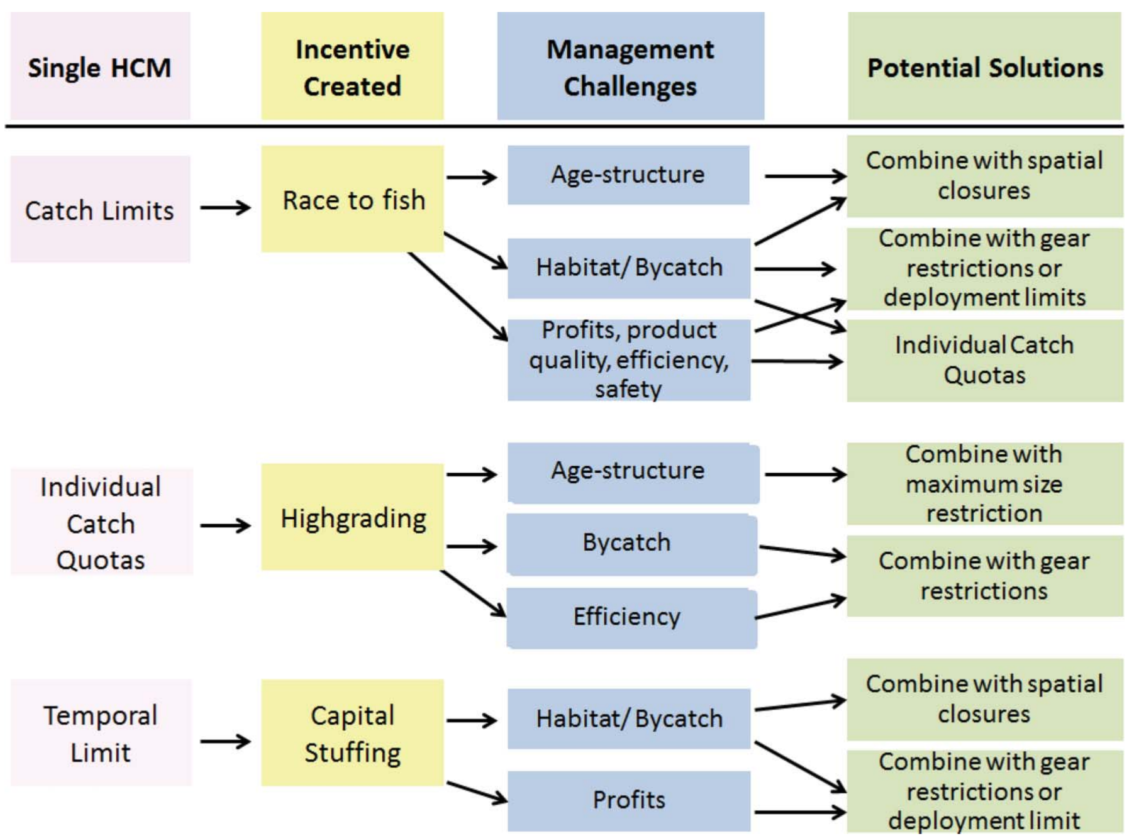

Figure 1. Examples of combining HCMs, rights-based management, and spatial management to tackle challenges associated with single HCMs. Single HCMs (pink boxes) can lead to incentives for fishers that create management challenges (yellow and blue boxes), but can be mitigated by combination with other strategies (green boxes). This figure shows just a few examples of combining HCMs, but others could utilize a similar framework.

sustainability) and to discard unwanted catch. Open access conditions in a fishery can exacerbate these negative outcomes (Pope, 2002). Finally, management and scientific uncertainty both contribute to and are a result of negative outcomes of HCMs. This section briefly describes how open access, uncertainty, the race to fish, and discarding can erode positive benefits of HCMs and create unexpected negative consequences.

Open access

When participants in a fishery and the access to its resources are not limited, HCMs often are unable to meet management objectives (Pope, 2002). Fishers generally apply high personal discount rates (although this varies within and among fisheries, Curtis, 2002; Richardson et al., 2005), have little incentive to operate in their long-term best interest, and instead act to capture as large a share of the resource as quickly as possible. Individual HCMs like bag or deployment limits may incentivize latent effort in a fishery, or encourage new entrants, thereby increasing total effort and fishing mortality (Sissenwine and Kirkley, 1980; Waters, 1991; Salas and Gaertner, 2004). For example, in the Maine lobster fishery, the institution of an individual trap limit resulted in some large-scale, efficient fishers being forced to reduce their number of traps, while at the same time, small-time lobstermen could increase their pot numbers (Acheson, 2001). The ultimate result was an increase in total lobster traps fished in the Gulf of Maine and a decrease in economic efficiency, the opposite of the intended result. This unintended outcome led lobstermen to lobby for further regulation and limited entry in the fishery. Limited entry can eliminate fisher competition with outsiders by removing uncontrolled entry of new participants, but limited entry alone does not necessarily lead to better outcomes, in part because of capital stuffing (next section, Metzner, 2005; Branch and Hilborn, 2006).

Co-management, or the devolving or sharing of regulatory power between governments, communities, and other stakeholder groups, is another strategy that has been successful at alleviating open access issues in some fisheries, notably in parts of Africa, Asia, and the European Union (reviewed by Wilson, Nielsen, and Degnbol, eds., 2003). Co-management is often most successful when fishing communities are small and local, and community cohesion is already strong (Pinkerton, ed., 1989), but there are challenges to co-management as well. Power struggles between stakeholder groups, membership conflicts, and divestment of important government resources are all obstacles to successful co-management (Njaya, 2007), and while it is a promising solution, co-management is not applicable in all fisheries.

The race to fish and capital stuffing

Under open access, catch limits and temporal limits can incentivize fishers to compete with one another to maximize their share of their catch. This race to fish compromises fishing efficiency, harvest rates, and product quality, and can lead to negative impacts on habitat and bycatch (Branch and Hilborn, 2006; Waters, 1991; Gillis et al., 1995). There are two strategies fishers can take to bolster their own harvest in response to restrictions: increase their fishing time or increase fishing power (by increasing their inputs of labor or capital). For example, if fishers are constrained by a catch or temporal limit, they are incentivized to make more trips in an attempt to approximate their previous level of catch before the implementation of the new regulation (Coleman et al., 2004; Ault et al., 2005). These extra trips even in poor conditions lead to the safety concerns associated with catch, bag/trip limits and temporal limits (Hanna and Smith, 1993; Kaplan and Kite-Powell, 2000).

Because HCMs tend to restrict only one or a few dimensions of fishing effort, the race to fish incentivizes excessive investment in unrestricted dimensions of fishing effort in reaction to regulation, a phenomenon termed “capital stuffing” (Branch and Hilborn, 2006). In the attempt to maximize catch rates, fishers can experience decreased profits. For example, if a management measure limits vessel length, fishers can increase vessel width to process more catch, which occurred in the Dutch beam trawl fleet (Rijnsdorp et al., 2008). If a temporal limit or vessel restriction constrains fishing, fishers are incentivized to invest in other capital inputs (e.g., extra gear on board, larger engines, etc.) to more quickly capture a share of the catch. In the Bristol Bay sockeye salmon fishery, vessel length restrictions led fishers to invest in increasing vessel width and engine power, which increased the cost of fishing and reduced overall profits (Metzner and Ward, 2002). In the Dutch beam trawl fishery, fishers have continued to modify their technology in order to fish harder, as a result of both competition with one another and as a response to management constraints (Rijnsdorp et al., 2008).

The incentive to discard

Some HCMs (when used individually) can increase discarding of bycatch (Gillis et al., 1995). Bycatch refers to any unintended catch, such as individuals of species other than the target species, or sizes or sexes of the target species that fishers are prohibited to land by regulation. Gillis et al. (1995) distinguish three forms of discarding, which can all have serious biological, ecological, and socioeconomic implications. The three forms of discarding are: exclusion, where unwanted species or sizes are simply removed from the catch; capacity-discarding, where fishers are forced by regulation or vessel capacity to discard species for which regulatory limits have been reached; and high-grading, where fishers preferentially discard otherwise marketable fish to preserve room for more valuable individuals (e.g., larger fish, Kristofersson and Rickertsen, 2009). Bycatch issues have arisen in small-scale to industrial fisheries, including a multi-species Kenyan reef fishery (Mangi and Roberts, 2006), the U.S. West Coast and New England groundfish fisheries (Branch, 2004), and the Icelandic cod fishery (Kristofersson and Rickertsen, 2009). Bycatch on a worldwide scale is difficult to estimate, but may be on the order of more than 7 million tons/year (Kelleher, 2005). In 2002 in the U.S., discarded biomass was approximately 25% of total catch (Harrington et al., 2005).

Catch limits, bag/trip limits, size limits, and sex-specific limits can all incentivize discarding behavior (Sissenwine and Kirkley, 1980; Branch and Hilborn, 2006). If there is not a substantial additional effort, cost, or penalty associated with being selective in which individuals and which species are retained by fishing, fishers will maximize their landings of the most valuable fish. Moreover, in the race to fish, fishermen generally have less leeway to fish carefully and will be less selective, potentially increasing discard rates (Harrington et al., 2005).

Discarding of all types has the direct effect of causing additional mortality outside of the landings of target species. Importantly, not all fish that are discarded survive. Discard mortality is difficult to measure and varies across fisheries and species, but can often be as high as 60–100% (Davis, 2002; Gillis et al., 1995). If discard mortality is indeed significant, then discarding will have other negative biological and ecological effects, and introduce mortality that is unaccounted for in many fisheries assessments (Crowder and Murawski, 1998). High-grading can exacerbate age structure truncation in the fished stock, as well as induce long-term, unnatural selective pressure for smaller, less productive individuals (Sampson, 1994; Branch and Hilborn, 2006). Bycatch, even of commercially worthless fish, is often comprised of important prey, predators, or competitors of targeted species whose abundance may influence the dynamics of valuable species (Gillis et al., 1995). Bycatch of protected species (e.g., marine mammals, sea turtles, sea birds) is another ecological concern (Crowder and Murawski, 1998).

Bycatch represents an economic loss for fishers. Regulations that force capacity-discarding or incentivize high-grading cause the complete loss of the value of fish that might otherwise be landed. For example, in southern New England, a size restriction on yellowtail flounder caused nearly 46.5 million fish to be discarded from 1987-1992, representing nearly 60% of the catch over that time period and valued at more than $50 million (Alverson et al., 1994).

Bycatch also creates issues for fishery monitoring by managers and regulatory agencies. Discarded fish are not included in catch accounting unless there is full accountability or 100% on-board observer coverage, or extrapolated estimates of discard rates are used (Turris, 2000). If discard mortality is high, discarding becomes a hidden source of fishing mortality, and can represent a large source of uncertainty in stock assessment (Davis, 2002).

Uncertainty

Unintended consequences of HCMs like the race to fish, capital stuffing, and discarding lead to scientific and management uncertainty (Rice and Richards, 1996). The entire process of fishery management, from stock assessment to implementation to monitoring and enforcement, is subject to uncertainty, which is a significant challenge in achieving the desired outcomes of HCMs (Fulton et al., 2011). Progress has been made in estimating some sources of scientific uncertainty and accounting for it in the setting of HCMs (Punt and Hilborn, 1997; Johnson et al., 2014; Hurtado-Ferro et al., 2015; Restrepo and Powers, 1999; Szuwalski and Punt, 2012). One type of uncertainty, the uncertainty in the response of fishers to management, has been less studied and has proven difficult to reduce, quantify, and incorporate effectively into management (Fulton et al., 2011). Although existing empirical studies suggest that the responses to HCMs described above (the race to fish, capital stuffing, and discarding) are fairly predictable, the magnitude of those effects and their impact on the resource is extremely difficult to predict and mitigate (e.g., Metzner and Ward, 2002; Rijnsdorp et al., 2008).

Mitigating limitations of individual HCMs

It is clear that many HCMs, considered alone, are either unable to overcome traditional challenges in fisheries management, such as open access and uncertainty, or exacerbate the problem by incentivizing unintended fisher behaviors. These hurdles make it challenging for managers to achieve their desired objectives (Branch and Hilborn, 2006; Fulton et al., 2011). To mitigate many of the negative impacts associated with the implementation of HCMs (Table 3), managers can strategically combine HCMs, incorporate rights-based management, and/or implement spatial management. Utilizing these strategies, harvest can be more effectively controlled, and multiple, even competing management objectives can be better achieved. Figure 1 summarizes some of the improved outcomes when these strategies are employed, which are discussed in detail in the following sections.

Combining harvest control methods

A limitation of many HCMs is that they are designed to manage a single species, leading to potentially unaccounted for effects on ecosystem dynamics (Holland, 2007). Management strategies that use a combination of appropriate HCMs can achieve multiple biological, ecological, and socioeconomic objectives.

Examples of successfully controlling harvest in a fishery and simultaneously achieving other objectives suggest that a HCM aimed at directly controlling fishing mortality should be combined with other HCMs designed to meet ecosystem-specific objectives such as reducing habitat damage or landings of bycatch (Graham et al., 2007; de Bruyn et al., 2013). Many of these effective combinations are evident from examining the columns of Table 3. For example, to limit habitat damage caused by the race to fish under a catch limit, the catch limit may be combined with gear/vessel restrictions or deployment limits (Figure 1). This has been an effective management strategy in the Oregon trawl fishery (Bellman et al., 2005). The combination of catch limits with gear restrictions can also limit the ability of fishers to increase fishing power by switching to potentially more destructive fishing practices (Branch and Hilborn, 2006).

Catch or bag limits that directly control mortality may be combined with temporal limits or gear restrictions to improve selectivity and reduce discards (e.g., Vestergaard, 1996; Gillis et al., 1995; Figure 1). In the Kuwait trawl fishery, combining catch limits with seasonal temporal restrictions and spatial closures significantly reduced bycatch (Ye et al., 2000). Successful management of walleye pollock in the eastern Bering Sea and the fishery's relatively low levels of bycatch are attributed to gear restrictions combined with catch limits (Graham et al., 2007).

Combining multiple HCMs is also effective for buffering against the multiple sources of uncertainty described in the previous section (Waters, 1991; Sutinen, 1999; Fulton et al., 2011). The magnitude and source of uncertainty associated with the implementation of an HCM will vary and largely depends on the type and quality of data used to estimate a sustainable fishing mortality rate, how well fishery inputs or outputs can be used to accurately estimate fishing mortality, and the capacity of management to effectively monitor and enforce regulations (Stefansson and Rosenberg, 2005; Yamazaki et al., 2009). When multiple HCMs are implemented, uncertainty can be reduced, reducing the risk of stock collapse (Dichmont et al., 2001; Hilborn et al., 2001; Stefansson and Rosenberg, 2005; da Rocha and Gutiérrez, 2012). For example, the International Pacific Halibut Commission uses both catch limits and size limits to manage the stock, which reduces uncertainty in assumptions about stock size structure and fisheries selectivity in assessment (Hilborn et al., 2001). The Australian Northern Prawn Fishery uses a deployment limit, gear restrictions, and a days-at-sea approach in part to reduce the uncertainty in the estimate of the relationship between fishing mortality and fishing effort (Dichmont et al., 2001).

Combining HCMs must be done strategically and is not always successful in achieving multiple objectives. The "band-aid" approach (Hilborn et al., 2004), where more and more regulations are implemented in a reactionary rather than an adaptive or strategic manner, can lead to inefficiency and negative outcomes. In the New England cod fishery before the introduction of rights-based management, management consisted of TACs, gear restrictions, size limits, trip limits, and spatial closures, all with little success in recovering an overfished stock (Hennessey and Healey, 2000). Therefore, simply employing multiple HCMs is not necessarily effective in itself, and instead, careful, strategic combination of HCMs is of greater importance.

Rights-based management

The socioeconomic and biological/ecological issues associated with open access fisheries and the race to fish can be overcome by creating incentives for fishers to behave in way that is aligned with management objectives (Fujita et al., 1998; Fulton et al., 2011). Rights-based management gives fishers exclusive rights to a fishery resource by allocating ownership of catch, effort, or area in a fishery to individuals or a community (Charles, 2002).

Individual quota-based catch limits

Individual, quota-based catch limits, or catch shares, allocate a fleet-wide catch limit to individual fishers or small groups of fishers (Hilborn et al., 2004; Fujita and Bonzon, 2005; Quentin Grafton et al., 2006; Beddington et al., 2007; Costello et al., 2008). An advantage of catch shares is that the race to fish is eliminated, alleviating many of the negative ecological and socioeconomic impacts that occur under catch limits (Costello et al., 2008; Branch, 2009; Essington et al., 2012; Figure 1). If they are constrained by only their own annual or seasonal share of the catch (often, a percentage of the TAC), fishers are now incentivized to maximize their personal efficiency in capturing their share, reducing costs to increase profits rather than racing to increase revenue through greater catch (Costello et al., 2008; Figure 1). Furthermore, if a fisher's catch share is a secure right (i.e., guaranteed for a long period), the fisher now has a long-term incentive to protect the sustainability of the stock as a whole, in order to increase the value of his or her proportional share over time as the fishery recovers (Squires et al., 1995).

It is important to note that individual catch limits alone do not necessarily reduce, and actually can exacerbate, fisher incentives to discard (Copes, 1986; Arnason, 1994; Poos et al, 2010), but there are additional incentives that can be created for fishers to avoid discarding (Branch, 2009; Gilman et al., 2014). For example, specific bycatch and habitat quotas can be implemented in addition to individual catch quotas for target species (Diamond, 2004; Holland and Schneir, 2006; Branch, 2009). Reducing the variation in market value between individuals or species landed in a fishery can also reduce discarding (Branch, 2009). In Iceland, discarding was reduced and scientists were able to more accurately estimate fishing mortality after management began implementing high fines for discarding and allowing a fixed percentage of bycatch in landings that does not count against individual quotas, thereby creating a low value market for bycatch species (Petter Johnsen and Eliasen, 2011). In Denmark, incentives to reduce discards were created by awarding higher catch quotas to vessels that are voluntarily equipped with video surveillance to monitor discard rates (Graham et al., 2007), simultaneously reducing scientific uncertainty and improving management outcomes.

Catch share performance is sensitive to design and implementation, so performance varies. For example, pressure from other related sectors with economic reliance on a fishery, such as fish processors, can still result in incentives for fishers to maximize catch on a specific schedule (e.g., Matulich et al., 1996). Nevertheless, empirical studies of catch share performance (usually, individual transferable quotas or ITQs) indicate that while catch shares do not always result in stock biomass increases, they do reduce the risk of fishery collapse, increase fishery profits, and improve compliance with catch limits (Costello et al., 2008; Essington et al., 2012; Melnychuk et al., 2012; Dewees, 1998; Branch, 2009). Efforts have been made to provide practical guidance on the appropriate design and implementation of catch shares (e.g., Bonzon et al., 2010).

Individual effort quotas

Individual/vessel effort quotas (IEQs) are a form of rights-based management that allocate fishing effort units to a restricted number of individuals or vessels (Metzner et al., 2005), instead of allocating shares of the catch as is the case with ITQs. Typically, IEQs are established in the form of number of fishing gears (e.g., number of traps) or number of fishing days (Charles, 2002). IEQs are more effective at controlling fishing mortality than days-at-sea or deployment limits alone because they eliminate uncertainty in how many fishers will participate in a fishery and total effort is capped (Pope, 2002). In some cases, IEQs may be preferred over individual catch limits to reduce the incentive of fishers to misreport landings, to limit the impact the fishery may have on habitat, and/or to reduce discards (Pope, 2002).

The ability of an IEQ to effectively limit fishing mortality and its potential impact on the cost of fishing is dependent on the potential of fishers to increase fishing effort through changes in unrestricted effort dimensions (Ulrich et al., 2002; Charles, 2002). In these cases, capital stuffing can still occur and the cost of fishing can dramatically increase (Pascoe and Robinson, 1998). Thus, the most successful examples of IEQs have occurred in fisheries where the potential for capital stuffing is limited due to fishing strategy, or when all dimensions of fishing effort are managed through gear restrictions or deployment limits (e.g., in the Bermuda and West Australian lobster fisheries, Ward and Luckhurst, 1991; Pope, 2002; Acheson and Acheson, 2010).

Territorial user rights in fisheries (TURFs)

Fishery managers can also grant spatial fishing rights to individuals or communities, commonly referred to as Territorial User Rights in Fisheries, or TURFs (Christy, 1982). Individuals belonging to a TURF establish HCMs that will apply inside their dedicated area. TURFs differ from previously discussed management strategies because rather than focusing on single-species, they encompass an entire area, a step towards a more ecosystem-based approach to management (Rieser, 1997). In some cases, TURFs have achieved levels of ecological outcomes comparable to marine reserves (Gelcich et al., 2012).

Like ITQs, TURFs can be successful in improving economic and incentive problems associated with other HCMs, often leading to better biological outcomes as well (Wilen et al. 2012; White and Costello, 2011). Like any other fishery management strategy, success of a TURF at achieving management objectives is dependent on design (e.g., the number of fishers and the geographical area included), conditions existing in a fishery (resource levels and socioeconomic context), and fisher behavior (Wilen et al., 2012). Fishers in a TURF do not always behave predictably, which can lead back to some of the same problems described above (unintended consequences), on a smaller scale (Waters, 1991; Gelcich et al., 2007). In other cases, community structure may not lend itself to the implementation of TURFs, resulting in high transaction costs relative to associated benefits (Pomeroy, 2012). Nevertheless, evidence shows that when infrastructure for fisheries management is lacking at the state, provincial and/or central government level, as in many developing nations and small-scale fisheries, TURFs can improve the management of a fishery by allowing fishers to monitor the fisheries and ecosystem in their designated area and adjust harvest strategies according to their own observations (Hilborn, 2007b). Granting a community ownership over their resources in a defined area can increase enforcement levels and decrease costs (Gutiérrez et al., 2011).

A limitation of TURFs is their inability to protect an entire stock from overharvesting if a species is highly mobile or the recruitment processes occur over a larger spatial scale than the TURF (Johannes, 2002; Hilborn et al., 2005; White and Costello, 2011). For example, TURFs have been identified as an ideal management strategy in the Australian abalone fishery because the spatial scale of the stock is small (Prince et al., 1998), but for species that have a wider spatial range, the reduced ability of the TURF to control the status of the stock will lead to a breakdown in the positive incentives they were implemented to create (White and Costello, 2011). Thus, in order for TURFs to be successful, effective HCMs need to be present over the total area of the stock, both within and outside of the TURF.

Spatial management

Finally, fisheries management can include spatial management strategies in addition to direct HCMs to tackle many common HCM problems. The most common and well-known type of spatial management is a marine reserve, a designated area where fishing is prohibited. Spatial management can also include spatio-temporal closures (e.g. closing an area with a known fish spawning aggregation for a limited time), the mandated spreading of allowable catch or effort across management zones, and areas that restrict certain fishing gears to protect habitat or certain fish life stages (Kritzer and Liu, 2013).

Spatial management in fisheries can improve biological and ecological fishery management outcomes (Figure 1). In the Alaska groundfish fishery, catch limits and strict spatial restrictions for fishing gear have been a successful management strategy for reducing bycatch, improving habitat, and maintaining a sustainable catch level, and their removal has been associated with increases in bycatch (Witherell et al., 2000; Dew and McConnaughey, 2005). Spatial management as a bycatch reduction technique has also been employed with success in the Kuwait shrimp fishery and the U.S. pelagic longline fishery (Ye et al., 2000; Goodyear, 1999). In Norway, seasonal spatial closures and area-specific catch limits have been used to control harvest and protect juvenile cod (Graham et al., 2007). In Bermuda, temporal closures of red hind spawning aggregation sites in combination with gear restrictions have been effective at controlling harvest and protecting spawning behavior (Luckhurst and Trott, 2008; Dean et al., 2012). These benefits are in addition to well-documented effects of the reserves themselves, including reducing the overall impact of the fishery on habitat (McClanahan and Arthur, 2001), improvements of ecological community structure, an increase in primary and secondary productivity (Babcock et al., 1999), and an increase in stock biomass and protection of stock age structure (Lester et al., 2009; Berkeley et al., 2004).

A major advantage to the implementation of marine reserves is their ability to completely remove fishery impact from the protected area (if well enforced), providing a buffer against uncertainty (Rice and Richards, 1996; Turner et al., 1999). A model by Stefansson and Rosenberg (2005) found that protecting a fraction of a fish stock in reserves reduces the risk of overfishing and the chance of stock collapse in the long term.

Similarly to single HCMs, reserves are not useful in all fisheries, and moreover, reserves in isolation can produce negative consequences as a result of fisher behavior (Hilborn et al., 2004). The creation of spatial restrictions on fishing often causes a shift in the spatial distribution of fishing effort, and can induce overexploitation outside reserve boundaries of the very stocks the reserve is designed to protect, or other vulnerable species (Turner et al., 1999; Coleman et al., 2004; Rijnsdorp et al., 1998; Stelzenmuller et al., 2008). Reserves can also lead to complications for stock assessment by potentially promoting a patchier spatial structure (Field et al., 2006).

Displaced fishing effort and unintended consequences resulting after implementation of a reserve can be mitigated when effective HCMs are in place outside of the reserve (Allison et al., 1998; Hilborn et al., 2006). When harvest levels are appropriately controlled a spillover of biomass from marine reserves to the adjacent fishery often occurs that can benefit fisheries (Roberts et al., 2001; Gell and Roberts, 2003). For example, in the Mediterranean Sea, the implementation of a marine protected area in combination with gear restrictions and effort limits was linked to a doubling of catch per unit effort within 4 years of management implementation (Guidetti and Claudet, 2009). The establishment of marine reserves in combination with effort controls in the Sourfriere Marine Management Area in St. Lucia resulted in a significant increase in reef fish landings after 5 years (Roberts et al., 2001). In New England, fishing mortality of depleted groundfish stocks was significantly reduced and other ecological resources were protected after the implementation of marine reserves, gear restrictions, and trip limits in the Georges' Bank groundfish fishery (Murawski et al., 2005). Thus, the combination of appropriate HCMs and spatial management can be mutually beneficial (Pauly et al., 2002; Yamazaki et al., 2015).

Conclusion

Selecting appropriate methods to achieve multiple fisheries objectives is a difficult task for fishery managers. HCMs can have both positive and negative effects on biological, ecological, and socioeconomic aspects of their fisheries, and these effects can run counter to some desired outcomes. Instituting regulations that control catch or limit aspects of the fishing operation can create strong incentives for fishers to adapt and increase other aspects of fishing effort in order to maintain catch, even at the expense of efficiency. Open access, the race to fish, incentives to discard unwanted catch, and uncertainty all create enormous challenges for fisheries management. Effective management therefore requires careful consideration of both the proposed objectives as well as the potential response of fishers to various methods of achieving those objectives. Any HCM in isolation carries the potential for both positive and unintended negative consequences. But, when combined strategically and with the inclusion of spatial and rights-based management, objectives can be effectively met, negative impacts can be minimized, and the overall uncertainty in management strategies can be reduced.

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