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Unintended consequences sneak in the back door: making wise use of regulations in fisheries management

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Abstract

In this chapter we discuss the potential failure of simple management models. Analysing components of a complex adaptive system in isolation is often misleading. The fundamental complexity of the social and natural environment has to be fully accounted for if unpleasant surprises are to be avoided. We examine a list of general management tools used in real-world fisheries, arguing that the success of a given instrument depends not only on its inherent properties but also on the way these instruments are administered. Similarly, we address how uncertainty and the biological complexity of the resource system may result in unintended consequences, including unanticipated costs. This demonstrates that for each resource system, the informational constraints have to be considered. Hence, interdisciplinary research is mandatory in order to reach adequate management decisions for social–ecological systems.

Introduction

Marine fish stocks are renewable natural resources. They have the potential to provide food, income, and other services to mankind on a sustainable basis (Smith *et al.* 2010). Yet in reality, overfishing – the wasteful

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exploitation of marine resources – is a widespread observable fact (Jackson *et al.* 2001, Hilborn *et al.* 2003, Myers and Worm 2003, Worm and Myers 2004). On the one hand, there is no doubt that globally fisheries are in crisis (Clark 2006). On the other hand, how we can manage to rebuild global fisheries is still under debate (Worm *et al.* 2009). Interestingly, there are few cases of environmental policy wherein the gap between actual and potential performance is as large as in fisheries (Heal 2007). The underlying cause of overfishing is most often thought to be the open access nature of many fisheries: each individual fisherman takes fish out of the ocean until the cost of catching one more fish exceeds the return of doing so. The fisherman has no incentive to leave fish as an investment for future harvesting; if the fisherman does not take the fish when they can be taken, another fisherman will. This problem is often described with the metaphor of the "tragedy of the commons" (Hardin 1968). Like most metaphors, it simplifies the true complexity of the problem. In this case it masks the two facets of overfishing that Munro and Scott (1985) defined as a "Class I problem" and a "Class II problem."

First, the Class I problem relates to excess fishing mortality; too many fish are harvested. Turned the other way around, too few fish are left in the oceans to reproduce. That is, future social and natural losses result from overstraining the replenishing potential of the resource. It resembles a "temporal trap" (Messick and McClelland 1983) as the concentration on today's gains squanders obtainable gains in the future.

Second, even when the government is aware of this problem and sets a Total Allowable Catch (TAC), too many boats will "race" to catch as much as possible until the TAC is reached. This is the Class II problem, where social and natural waste is the result of a perverse incentive structure brought about by the fact that fish can be appropriated only by the first fishermen to catch them, resembling a "social trap" (Messick and McClelland 1983). A symptom of this "rule of capture" (Boyce 1992) is the widespread overcapacity of fishing fleets.

Decision-makers today meet challenges not previously experienced in the era of unregulated open-access fisheries (Homans and Wilen 1997). On the one hand, today's decision-makers have more possibilities due to the increased level of knowledge. On the other hand, today's managers are expected to uphold both biological and economic sustainability in an increasingly complex world (Clark 2006). Not all management instruments work in the same way: while some solve the Class I problem, others overcome the Class II problem. Therefore, any management advice should specify whether it aims at solving a Class I problem, a Class II problem, or both.

An excellent framework for analysing such complex social–ecological systems for sustainable management is given by Ostrom *et al.* (2007) and Ostrom (2009). A social–ecological system consists of four subsystems: (i) resource

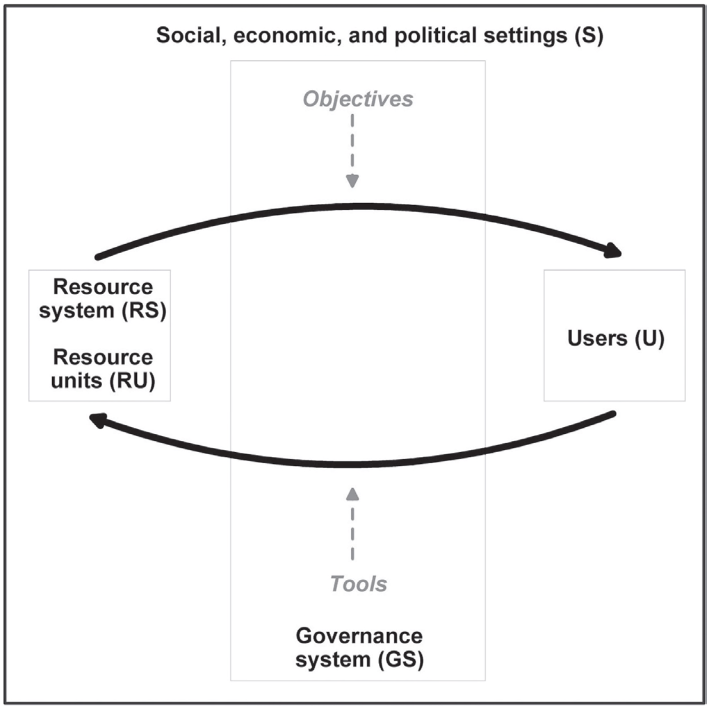


Fig. 5.1. An adaptation of Ostrom's (2009) framework of core subsystems for analyzing social–ecological systems in our marine fisheries context. Here, we emphasize the feedback loop between the resource system and resource units in their interactions with the users (black arrows) in the form of objectives, and the resulting management tools applied to the resource system and/or units (gray arrows).

system (e.g., a coastal fishery), (ii) resource units (e.g., fish stock), (iii) users (e.g., fishermen), and (iv) governance system (e.g., the specific laws and social norms in place) (Fig. 5.1). Within each subsystem, relevant variables can be identified to help map policy recommendations to specific system characteristics.

Panaceas for resource management typically fail (Ostrom *et al.* 2007). This can happen as a result of a variety of factors, often in combination. Frequently, this occurs because of overweighing, or, alternatively, simply ignoring the importance of one of the subsystems. For example, a solution that focuses on the protection of resource units (RU), like biomass of a certain species, may fail because it does not take into account how it is affected by the response of users (U) to these regulations (see Fig. 5.1). It is crucial to recognize the occurrence of feedback and, to the extent possible, to identify particular feedback structures within and between the specific systems (Berkes *et al.* 1998, 2003). If management strategies are based on results derived from analyzing one of the subsystems in isolation, the outcome may be very different than what the manager had in mind. These unintended consequences occur because overlooked or underemphasized issues will always find a way to sneak in through the back door. By this, we mean that models that look only at parts of the system lack important components that are present in reality. As a result, these models are inaccurate at best, but, often, they will also provide completely flawed results. It is conventional wisdom that every complex problem has an answer that is clear, simple, and wrong. We are obviously not the first ones who claim that this holds for fisheries as well. Wilson (1982), for instance, points out that objectives and forms of regulation would be very different from those proposed by the traditional economic view, when "complicating factors" were taken into account.

This chapter proceeds in three steps: In Part A, we highlight the fundamental complexity of the social and natural environment relevant for fisheries management. In Part B, we discuss a list of management tools with regards to their ability to alleviate Class I and Class II problems. We argue that this depends not only on the inherent properties of a given instrument but also on the way an instrument is administered. In Part C, which also serves as a summary, we broadly categorize different sets of social and natural complexity. By constructing four stylized examples, we highlight that the adequacy of a given instrument in a given case is contingent on the specific structure of the costs of implementation and the difficulty of obtaining all relevant information.

Part A: Complexity

Stakeholder participation and the social environment

Stakeholders can be defined as any member of society who has direct (primary stakeholders) or indirect (secondary stakeholders) interests, or stakes, in the actions of a fishery (Gray and Hatchard 2008). It is important to keep in mind that, in practice, managers and scientists often have hidden agendas themselves, in spite of their alleged neutrality (Jentoft and McCay 1995).

Stakeholder participation can be an effective way to reconcile conflicting objectives (Dankel 2009). Through an active and assisted dialog process, objectives can be cognitively broken down and made more compatible (Follett 1955). For example, the objective "highest sustainable yield" could in fact be a symbol for a more specific objective; say "operating within a 10% profit margin over the next 5 years." Likewise, the broad objective "ecosystem preservation" could be a symbol for a more specific objective like "a 50% decrease in the amount of trawling that has contact with bottom habitat." Additionally, participation may enhance the chances of reaching consensus and lead to better decisions due to the integration of the specific expertise these stakeholders have (Jentoft *et al.* 1998). Ideally, the outcome of such participation coincides with what would be best from society's point of view, especially regarding long-term sustainability. Unfortunately this is not necessarily the case. Far too often, the voice that shouts loudest is heard best (Hatchard 2005), especially when some stakeholders have far more resources (financial funds as well as knowledge) than others (Esteban and Ray 2006, Mikalsen *et al.* 2007). In many cases it is impossible to distinguish an active debate among stakeholders from lobbying. Often stakeholders are willing to spend a substantial amount of money and time on influencing political decisions. This form of "rent-seeking" activity (Krueger 1974, Johnson and Libecap 1982, Bergland *et al.* 2002) is, of course, highly undesirable from the society's point of view, but is often a well-established part of the political culture and therefore hard to eradicate. In spite of this, it would be naïve to conclude that all lobbying would cease if stakeholders were excluded from the present form of decision-making. This is especially true because the decision on whom to include and exclude is itself a political choice, making the process even less transparent (Mikalsen and Jentoft 2008). If primary stakeholders are involved in the decision-making process, they should therefore be made responsible and accountable (Berghöfer *et al.* 2008, Mikalsen and Jentoft 2008).

Managers, on their side, should also be accountable and bear the full responsibility of their decisions while as in current forms of management they have often lost neutrality (Jentoft and McCay 1995). In part, this is because sustainable long-term management use of marine resources requires planning over a time horizon that is longer than the duration of political offices. Such challenges are made even more difficult by the fact that, often, policy makers use fisheries management as a vehicle to solve other political issues (and if these involve other environmental issues, it introduces artificial connections and relationships among the various elements of ecological systems). Prominent examples are regional development, employment or simply redistribution of income. These are all legitimate political choices, but they do not necessarily fulfil the explicit management goals of a fishery.

When management objectives have been identified and prioritized, scientists may present management trade-offs based on current knowledge of the fish stocks. But scientists are often confronted with large degrees of uncertainty (especially in regard to trade-offs and consequences involving other components of an ecosystem) that, especially when not successfully communicated, can disillusion stakeholders (Rosenberg 2007) and breed distrust towards scientists and their methods. Therefore, an open dialog process (Follett 1955) is a pertinent first step where different stakeholders and scientists can meet to gain more knowledge of inherent trade-offs of the resource, data and modeling involved to support management transparency and trust-building.

In most cases, fisheries management is a top-down bureaucratic exercise with centralized control (Gray and Hatchard 2003, Prince 2003, Daw and Gray 2005); there is a tendency to disconnect the human system from the ecological system by not explicitly including the human component of ecosystems with all of its user groups. Since there are important feedbacks from the governance system to the users, including or excluding stakeholders will lead to institutional repercussions. Central intervention from authorities very often directly undermines existing norms of cooperation, lowers the willingness to obey these rules and weakens stewardship motives. The literature has identified many cases where external interventions, intended to stimulate certain behavior, in fact eroded any motivation to voluntarily behave as intended (Frey *et al.* 1996, Deci *et al.* 1999, 2001, Frey and Jegen 2001, Gintis *et al.* 2005, Ostrom 2005, Frey and Stutzer 2006, Bowles 2008, Vollan 2008, Richter and van Soest 2010). This phenomenon, often referred to as "crowding out," holds especially for external incentives in the form of direct payments, but also for external control that signals distrust to the individual. This happens because individuals base their decision not only on financial considerations, but are also often intrinsically motivated to be a good member of society. A fisherman may, for instance, feel responsible or morally obliged to use nets that minimize bycatch: he may want to signal to others that he is a trustworthy person, who has high moral standards. Standard economic models typically ignore how moral motivation is affected by financial incentives. Instead, it is assumed that financial incentives come on top of moral motivation and, when the two are consistent, one would expect that it can only strengthen the overall incentive. The literature on crowding-out (where one motivation replaces another), however, has established that this assumption is often invalid because moral incentives and financial incentives are interlinked and therefore non-separable (Bowles 2008): a financial incentive directly affects, and often crowds-out (i.e., replaces), the incentive coming from moral motivation. If a fisherman suddenly receives money for using bycatch-minimizing nets, this external reward may supplant his moral motivation to use them voluntarily. As a result, he may still use more of such nets (if the financial incentive is large enough), but, in principle, it is possible that he will use less of them if the incentive is perceived to be too small.

In principle, it is also possible for government policy to crowd-*in* (i.e., stimulate) good behavior through the decisions by fishermen other than simply to conform to policy. If banning nets that produce a lot of bycatch helps stigmatizing the use of them, a fisherman who is personally indifferent about the problem of bycatch may not be indifferent towards social pressure and may try to comply with the social norm. Therefore, governmental policy can also help by supporting and evoking social values and public-spirited motives (Bowles 2008).

Financial incentives are not alone in replacing voluntary actions; external control can do the same. In many cases, an individual obeys a certain social norm or law because he considers himself to be a good citizen, and not so much because he fear to be fined. Once the authorities start monitoring an individual frequently, he responds to this signal of distrust by non-compliance when he is not monitored. This can happen because he infers that he is simply not expected to comply by default, or he reciprocates this sign of mistrust by breaking the rules. In both cases, the individual sees the authorities as an opponent, rather than as a partner. This finding has been corroborated in economic experiments and distrust has aptly been called "the hidden cost of control" (Falk and Kosfeld 2006).

Policy makers should take into account that any external intervention may have feedbacks not predicted by simple standard economic models. Some fairly simple rules can be used to try to minimize the negative consequences (Ostrom 2005, Frey and Stutzer 2006, Richter and van Soest 2010). First, policies that are designed in a way that reveals distrust towards users will most likely destroy any voluntary compliance that may have been present before (Anderson and Lee 1986, Sutinen and Kuperan 1999, Hatcher *et al.* 2000, Bowles 2008) and certainly inhibit additional voluntary compliance.

Second, a law that is not perceived to be legitimate and fair, is less likely to be obeyed (Frey 1997, Ch. 6). A good example comes from Denmark, where "fishers feel they are taken hostage by an illegitimate management system, and thus feel it is morally correct not to comply" (Raakjær Nielsen and Mathiesen 2003). In South Africa the government tried to reduce illegal fish landings by establishing formal rights for the local fishermen. But some fishermen had the feeling that the process was not fair and expressed their discontent by "protest fishing" (Hauck 2008). Similarly, economic experiments in the laboratory have shown that individuals indeed feel less obliged to comply with regulation by an institution that is perceived to be unfair (Kosfeld *et al.* 2009).

Stakeholder participation can be an important way to achieve legitimacy (Jentoft *et al.* 1998, Hatcher *et al.* 2000, Dankel 2009). Such a participatory approach may build trust among users themselves, but it also contributes to trust between users and central authorities. Economic experiments have indeed shown that involving individuals in the process of institutional design leads to more efficient outcomes (Ostrom *et al.* 1992, 1994, Vyrastekova and van Soest 2003). On the other hand, if individuals fail to reach consensus, stakeholder involvement can be counterproductive; the outcome can be less cooperative than if the individuals had never been involved in designing the institution (Sutter and Weck-Hannemann 2004, Tyran and Feld 2006). These findings from controlled experiments indicate that stakeholder participation can replace opposition with motivated stewardship, and increased compliance. But this will only be the case if an actual consensus is reached and the institution is designed in a fair way.

Uncertainty and the biological environment

Worldwide marine fish stocks are declining (Worm and Myers 2004, Worm *et al.* 2006, 2009, FAO 2008,), leading to changes in ecosystem structure and functioning. After overexploitation of large predatory species, fishermen may switch to target smaller prey species, making "fishing down the food web" a predominant threat to overexploited marine systems (Pauly *et al.* 1998, 2002, Pauly and Palomares 2005). Habitat loss from trawl (the fishing net usually towed behind a fishing vessel) activity and bycatch (unintended mortality of non-targeted organisms caught in fishing gear) threatens populations of non-targeted species. This may be manifested as a reduction in species richness and ecosystem diversity (Armstrong and Falk-Petersen 2008).

Fishing may also be effectively size-selective where larger fish are more likely to get caught, leading to age-truncation where younger age classes dominate the population and spawning stock biomass (Marshall *et al.* 2006, Ottersen 2008). Such juvenation and loss of age diversity may negatively affect recruitment and make stocks less robust or resilient to climate change and variability (Hsieh *et al.* 2006, Marshall *et al.* 2006, Ottersen 2006). Pertinent questions arise. How does fishing and changes in the environment, like climate change, affect inter- and intra-species interactions? In turn, how do these impact food-web dynamics and ecosystems? For example, how do fisheries change stock vulnerability and resilience? Are there tipping points where, beyond a certain threshold, stock collapse is inevitable? And, if the stock collapses, what is the potential for recovery?

Fishing can change the basic dynamics of exploited populations; for example, exploitation can result in larger variability in fish abundance, which may potentially pave the way to systematic declines in stock levels (Anderson *et al.* 2008, Stenseth and Rouyer 2008). A recent study that summarized the magnitudes of phenotypic change in fish, ungulates, invertebrates, and plants found that harvesting may produce rates of evolution up to 300% greater than in natural systems (Darimont *et al.* 2009). In commercial fish populations, changes in life-history traits, exemplified by maturation at earlier ages and smaller size, are greater when exposed to strong fishing pressure (Sharpe and Hendry 2009). Such phenotypic changes may have a genetic component driven by the selection pressure caused by intense harvesting (Heino 1998, Heino *et al.* 2002, Heino and Godø 2002, Olsen *et al.* 2004, Dieckmann and Heino 2007, Marshall and McAdam 2007, Dunlop *et al.* 2009, Stenseth and Dunlop 2009). Potential effects of such genetic changes include the erosion of genetic and phenotypic diversity (Jørgensen *et al.* 2007). Therefore, fisheries-induced evolution is of special concern because genetic changes may be difficult to reverse (Law and Grey 1989, Conover *et al.* 2009, Enberg *et al.* 2009). The extent to which fisheries-induced evolution occurs and how important it is compared with other factors are being debated (Hilborn 2006, Conover and Munch 2007, Jørgensen *et al.* 2007, Browman *et al.* 2008, Andersen and Brander 2009, Ozgul *et al.* 2009). However, addressing the genetic impact in such phenotypic changes is important if management is to be precautionary. Otherwise negative socio-economic and biological consequences from unnoticed fisheries-induced evolution (including coevolutionary effects on other species) could sneak in the back door.

The identification and, where possible, the quantification of uncertainty in all the steps from data collection to model implementation is crucial to derive reliable projections for decision-making. In fisheries, the first level where uncertainty enters is in survey data and catch statistics, with cascading effects into models and model choice. Therefore, stock assessment (quantification of the number of fish in the sea) is a challenging, but crucial field of research. Models are continuously being improved or replaced. For example, survey estimates used in population models are not always consistent, and are difficult to reconcile with commercial catch statistics. To meet these challenges, as they involve uncertainty in marine science, state-space modeling, a statistical modeling framework, has become popular for use on data for many fish stocks (Millar and Methot 2002, Millar and Meyer 2002, Aanes *et al.* 2007, Bogaards *et al.* 2009, Lindegren *et al.* 2009, Swain *et al.* 2009, Eikeset *et al.* 2010).

Choosing the level of model complexity is another challenging task: management has often focused on single-species populations, especially historically. However, it is progressively being recognized that single-species applications are inadequate for management decision-making when they exclude important multi-species feedbacks like predator–prey relationships within an ecosystem (Hjermann *et al.* 2007, Lindegren *et al.* 2009, Morissette *et al.* 2009).

All of these factors contribute to the overarching principle of biological complexity of ecosystems. This principle contributes to the understanding of how fishing can create substantial change in ecosystems, to result in altered structure or function (e.g., lower biodiversity). Some of the changes may result in lower yield from the targeted fish; some changes may be hard or impossible to reverse even if fishing ceases (Casini *et al.* 2009, Enberg *et al.* 2009, Lindegren *et al.* 2009). To meet the goals of adaptive management, models need to integrate the natural and social system as early as possible in order to provide knowledge and develop specific operational objectives for the resource.

Part B: Fisheries management

Many different tools for fishery management are available and have been applied and analyzed over the past decades. It is clear that what works well in one setting may lead to management failure in a different context (Brock and Carpenter 2007, Ostrom *et al.* 2007). Therefore, a key message is that a single best management instrument does not exist (Grafton 2000, Caddy and Seijo 2005, Degnbol *et al.* 2006, Jentoft 2006, Dankel *et al.* 2008, Ostrom 2008). Successful policy is not so much a question of inventing a new and magic strategy, but of adequately applying existing instruments to the specific situation at hand. However, this has proven to be difficult in the past.

Management responsibility

An often overlooked question is not only *what* to manage, but *how* to manage. For example, a regulation on the total allowable catch for one fishery may have very different effects, depending on whether it is agreed upon communally or administrated by a central government. A key ingredient of any successful management strategy is to provide the users with the right incentives. We will therefore take the question of how management is brought about as our principal characterization when portraying the management tools below. Afterwards, we will discuss specific management tools in more detail.

Centralized management

The vast majority of industrialized fisheries are managed by a central authority (government) which stipulates laws and regulations that are legally binding. If users are caught violating these regulations, they face a penalty. This seems to be a straightforward bureaucratic approach, as the government by its very nature is equipped with the power to set up, monitor, and enforce a given set of rules. The costs of doing so can, however, be extremely high, and there is a real danger that users will be alienated. As a result, informal arrangements between the users may be crowded-out (i.e., replaced), and so may any willingness to comply with these laws. The “hidden cost of control” (Falk and Kosfeld 2006) in the form of distrust can be substantial. As a general rule, successful central management requires strong enforcement and monitoring. Therefore, even if a certain law or regulation can be easily formulated, it can be extremely difficult to implement and enforce it.

There is also the danger that unintended consequences of economic incentives will sneak in through the back door. If it is forbidden to land a species that is threatened and the fines for doing so are high, the users may throw it overboard when it comes on deck as bycatch. This may mask the overall effects on fishing on this particular species as conventional catch data used for stock assessments will not reflect bycatch discards.

Co-management

In contrast to centralized management, co-management relies on a broader sharing of management responsibilities between governing systems (i.e., the State), research institutions and stakeholder groups. In fisheries discourse, co-management is presented as an alternative model which is reliant on stakeholder dialog and participation for cooperative management decisions between the State and other co-managers. A good review of the various forms of co-management is provided in Carlsson and Berkes (2005) and a review of implementation of fisheries co-management in developing countries can be found in Chuenpagdee and Jentoft (2007). In the context of fisheries, most research regarding co-management identifies legitimacy and stakeholder empowerment as important success factors of such a governance regime (Jentoft *et al.* 1998, Jentoft 2000a, Jentoft 2000b, Jentoft and Mikalsen 2004, Jentoft 2005, Chuenpagdee and Jentoft 2007, Jentoft 2007, Pinkerton and John 2008, Armitage *et al.* 2009, Jentoft *et al.* 2009).

Community-based management

Community-based management takes the co-management model a bit further from the top-down model and closer to a bottom-up management paradigm. The idea of community-based management is that the fishing community itself, separate from the state, decides on a harvesting strategy that is sustainable and profitable. This implies that the government deliberately steps down and relies on the community to develop management decisions. Actions may be legally non-binding, but still not purely voluntary, as they are based on social norms that may be enforced by fellow community members (Ostrom *et al.* 1992). Therefore, rule-compliance may be mandatory for members of the community and heavily sanctioned according to rules developed locally or at higher levels. This form of community-based management can be powerful, especially when users have close social ties and share the same norms and values. The government may, however, take a supportive role in giving scientific advice, by facilitating community meetings, or by encouraging desired behavior, such as promoting the use of nets that minimize bycatch.

Many examples show that local users are able to agree on management decisions if certain conditions are met (McCay and Acheson 1987, Ostrom

1990, Baland and Platteau 1996, Ostrom *et al.* 2002). While community-based management aims at upholding a harvesting strategy by social norms of cooperation, the actual harvesting strategy may take the form of a regulation of the mesh size (gear regulation), the number of days at sea (effort regulation), or of any other variable that defines the fishing process. Hence, the way a harvesting strategy is put into practice is not necessarily specific to the community-based approach. However, what is specific to community-based management is the explicit involvement of users in the process of deriving and implementing rules (Jentoft 2000b) (for example, via structured group consultations).

It is worth pointing out that social norms often solve the "social trap" (Class II problem), but not necessarily the "temporal trap" (Class I problem). Fishermen may, for instance, take turns getting the best fishing spots (rather than competing for them), but may strongly resist joining a cooperative to achieve long-term sustainability (Taylor 1987). Norms of cooperation may even aggravate the Class I problem of overexploitation. This may happen, for example, when norms are not aimed at sustainable management, but, instead, at lowering costs of exploitation (e.g., through sharing information about the location of the fishing grounds; Holm *et al.* 2000).

In spite of this, community governance can be very effective and efficient, in particular when the users are able to pool their risks or when cooperative management helps lower costs of harvesting (Swallow *et al.* 1997). The literature on this topic includes several key variables that can be linked to the self-organizing capacity of a community and the sustainability of common-property regimes. A good synthesis is given by Ostrom (2009), who identifies a common-property regime to be successful when: (i) the size of the resource system is moderate, (ii) the resource is neither too abundant, nor already exhausted, (iii) the system dynamics are predictable, (iv) the resource unit mobility is low, (v) the number of users is small, (vi) some users act as leaders, (vii) users hold common social norms and values, (viii) users have common knowledge about the system, (ix) the resource is very important to the users (in terms of livelihood or cultural value), and (x) the users have full autonomy for crafting collective-choice rules. By these standards, the chances for success of self-organized management for marine ecosystems are mixed (McClanahan *et al.* 2009). Some coastal (typically bay) fisheries can be successfully managed by a small community (Ostrom 1990, Schlager and Ostrom 1992, Schlager *et al.* 1994, Baland and Platteau 1996, Agrawal 2001, Ostrom *et al.* 2002, Ostrom *et al.* 2007), but when fish species are highly migratory and foreign fishermen are difficult to exclude, the prospects for community governance are rather bleak.

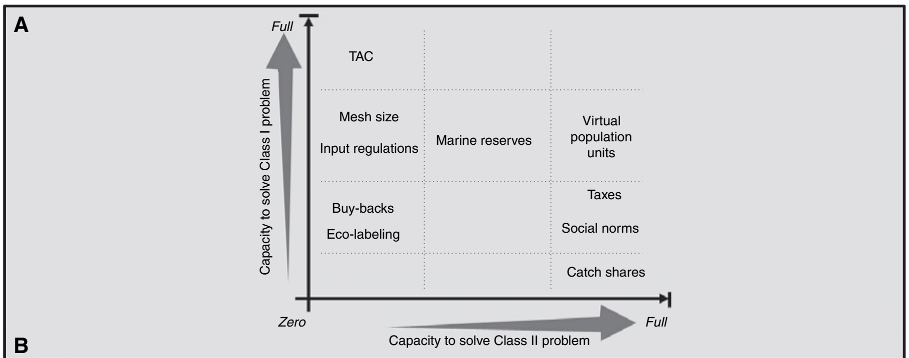
Management tools

Fisheries management can rely on a variety of tools (see Fig. 5.2 for an exposition of the tools we discuss). Good overviews can be found in Rettig (1995), Kahn (2005, Ch. 10), and van Kooten and Bulte (2000, p. 94). We will distinguish between tools that are based on a command and control approach, such as fines for catching fish below a certain size limit, and tools that are based on financial incentives, such as imposing a tax on landings. Finally, we attempt to give an overview of the ongoing debate on tradable permits. These are a special class of tools based on financial incentives in that they aim to exploit the efficiency of decentralized competition by creating a market for harvesting rights.

Management tools can be described and analyzed along several dimensions: one may ask whether a given class of policies aims to avoid the social or natural waste brought about by excessive harvesting (Class I problem), or changes the prevailing "rule of capture" (solve the Class II problem, see Fig. 5.2A). Alternatively one may ask whether a given instrument is robust to social and biological complexities, i.e., is it likely that it leads to unintended behavior from the fishermen, or is it likely that this instrument will lead to unintended changes in the resource or consequences for the ecosystem (Fig. 5.2B)? A management tool that targets the Class I problem should limit what is taken out of the water to ensure a sustainable stock for the future. This can be done by setting, for example, a Total Allowable Catch (TAC). A different angle is taken through input-centered instruments, which essentially control the way fish are taken from the ocean. An example would be to manage fishing capacity through controlling days at sea. Whether input or output controls perform better depends on many factors (Yamazaki *et al.* 2009); not all input controls are equally able to solve Class I or II problems and some of them are more likely to lead to unintended consequences than others. We will address this issue in the next section.

Command and control approach

Let us first take a closer look at the tools that are used to control what is taken out of the water (output controls), before turning to controls that regulate the way of harvesting (input controls). The prime example and most ubiquitous output-centred instrument is a cap on the TAC (Clark 2006). That is, all harvesting of a given fish species is prohibited once the total allowable volume has been landed. While this may effectively protect the resource stock and, in principle, solve the Class I problem of overfishing, a TAC does not necessarily lead to an efficient use of the resource (Class II problem). Quite to the contrary, each fisherman has an incentive to catch as much as possible before the



B

Tool	Biggest advantages	Biggest disadvantages	Challenges from the biological system	Challenges from the social system
TAC	Sets an effective limit on how much to be harvested	Requires substantial information about the ecosystem, and the realized catch	Uncertainty in stock assessment may overestimate stock size	Race to fish; fishing down the food web induced by TAC on top predator, fishermen substituting to other (unregulated) species
Social norms	No central regulation needed; no crowding out of voluntary actions	Scale-dependent, depends on self-organizing capacity of a community	Adaptation to sudden changes in the natural environment can be slow	Adaptation to sudden changes in the social environment (e.g., new and more efficient harvesting technology) can be slow
Minimum size limit	Administratively cheap and easy to monitor		May lead to unintended size-selective response in the harvested population	
Marine reserves	Protect habitat and species	Users who bear the costs of foregone fishing are often not the ones who benefit	Migration pattern and spatial distribution may change	May have strong distributional consequences for some users and, hence, lead to political opposition
Input regulations	Administratively cheap and easy to monitor	Only an indirect effect on the catch		Fishermen substitute unregulated inputs for the regulated ones (effort creep)
Taxes	Highly efficient, effective and create revenues for governments	A lot of information on the fishing sector is needed to set a tax that leads to an intended catch		Strong resistance form incumbent fishermen
Buy-backs		Very costly and often ineffective		In anticipation of a buy-back program users may increase capacity, leading to a situation that is worse than open access
Eco-labeling	Create consumer awareness and may provide positive spillovers to non-labeled fish	Can only have a complementary character		Price increase can lead to increased effort, proliferation of self-serving labels
Virtual populations units	In theory, ensures perfect management	Unfeasible; requires perfect information at all levels; involves heavily virtual accounting; unclear scaling issues		
Catch shares	A transferable quota solves the problem of overcapacity efficiently without central intervention	Effects on ecosystem depends fully on corresponding TAC; costly to administer and high monitory requirements; duration and issuance of catch shares has distributional consequences	Characteristics depend on TAC	Problem of bycatch, discarded catch and substitution to non-controlled species; auctioned quotas evoke resistance from incumbent fishermen; anticipated windfall profits from the introduction of ITQs may lead to capacity increase, leading to a situation worse than open access; ambiguous whether catch shares crowd stewardship motives in or out

Fig. 5.2. A classification of management tools and their characteristics. Panel A: Management tools and their capacity to solve Class I (excess fishing mortality) and Class II (overcapacity) problems. Panel B: Management tools and their general advantages, disadvantages, and challenges regarding biological and social complexity. TAC, Total Available Catch.

TAC is filled and the fishery is closed for the rest of the season, leading to the infamous “race to fish” (Grafton *et al.* 2006). In the extreme case, this kind of derby fishery can lead to the complete dissipation of profits as price and quality of the landed fish deteriorate while harvesting costs are increasing (Homans and Wilen 1997). Moreover, a significantly shortened season often places serious strain on fishermen, gear, and environment. One of the most infamous examples is probably the North Pacific halibut fishery, in the 1980s, when the year’s catch was taken in 3–5 days after opening of the season, regardless of weather conditions (Homans and Wilen 2005).

An additional problem with how TACs have been used is that they target individual species without consistent consideration of other species. Once the quota for one species is fulfilled, fishermen may shift to another one. The extreme case occurs when fishermen are “fishing down the marine food web” (Pauly *et al.* 1998, 2002, Pauly and Palomares 2005).

It is not desirable to set the TAC every year on an ad hoc basis, because this leads to substantial economic uncertainty for the fishermen. It also requires time-consuming negotiations between countries (for shared stocks), or within any individual State’s governing system, which can be an obstacle when the stock has declined and a collapse needs to be prevented by prompt emergency actions. Therefore it is helpful for managers to have an adaptive management plan for how stocks should be exploited. One increasingly popular management tool with the mission of a sustainable exploitation pattern is the implementation of harvest control rules (HCRs). In this approach, the TAC is established through specific input variables, especially the size of the spawning stock biomass. An HCR is a feedback control rule that links a harvest scenario and a stock size (Sandal and Steinshamn 1997, 2001, Arnason *et al.* 2004). An HCR framework can be built on the precautionary principle by including reference points that are quantified and set to prevent overexploitation and secure future stock recruitment by ensuring spawning stock biomass, or other selected indicators, to be above a defined precautionary limit (Beddington 2007). However, most HCRs in practice today retain the inadequacies of single-species approaches, with little, if any consideration of unintended consequences to other species and the ecosystem.

Although often overlooked in the literature, it is important to acknowledge that the “race to fish” also is an influential factor in determining a fished stock’s age composition (Turvey 1964, Wilson 1982). Given that a fisherman deems that his own action has little influence on the overall outcome, he will have no incentive to avoid targeting young fish; he cannot be assured that he will have the benefit of gains from the investment of leaving a fish in the ocean so that it can grow, reproduce, and be harvested at a later time. Many fisheries are indeed managed with *minimum size limits* that prohibit harvesting fish that are too young or too small. However, these size limits are almost always administered on an ad hoc basis and rarely take biological or economic criteria into account (Froese *et al.* 2008). Simulations from the Barents Sea cod fishery indicate that the profits could be more than doubled, simply by changing the mesh size (Diekert *et al.* 2009a).

Another output-centered management approach that has sparked considerable interest this decade is the use of *marine reserves*. The aim is to provide a spatial or temporal refuge to particularly vulnerable or valuable life stages of a population. Examples could be a no-take zone around a highly productive and diverse coral reef, or a seasonal closure of the fishery during spawning. Sumaila *et al.* (2007) found that closing 20% of the high seas to fishing may have a relatively small decrease in the global reported marine fisheries catch (1.8%), while the gain from reserves would be maintenance of marine diversity and benefits for current and future generations. In principle, marine reserves can be very effective in preserving biodiversity (Sumaila and Alder 2001, Lubchenco *et al.* 2003), particularly in warm water ecosystems (shallow water coral reefs) compared with temperate and cold open-water systems (Kaiser 2005). In spite of this, they can be quite inefficient, because adaptive behavior of fishermen harvesting outside the reserve may override the gains from protection (Hannesson 1998, Sanchirico and Wilen 1999). Alternatively, fish may migrate from densely populated, protected areas to less densely populated areas where they are harvested inefficiently. A large literature on marine reserves exists with considerable disagreement on the effectiveness of these methods (for overviews of this approach see Sanchirico *et al.* 2006 and Kaiser 2005).

One has to take into account that users may have the incentive to undermine the establishment of a marine reserve that has the purpose of protecting an endangered species. Marine reserves therefore perform particularly poorly if they are not effectively controlled and clash with existing community customs. While it is important to analyze the ideal design of marine reserves, it is even more important to build community support for them (Kareiva 2006). Hence, one may conclude that marine reserves work best embedded in successful community-based management or co-management. It is pertinent to note that identifying and quantifying long-term consequences of an extinction of a species to an ecosystem and its related economic consequences is extremely difficult (Van Kooten and Bulte 2000, ch. 8 and 9).

In general, the informational needs of output-centered instruments are demanding. The sustainability of a stock can only be ensured when its current size is accurately known, the total harvest can only be limited when the landings can be controlled, the fishing mortality can only be limited when it is known which fish are targeted by the fishing gear, and special components of the stock can only be protected when their attributes are known. The advantage of output-centered management tools is of course that they directly target the defining characteristics of the system (i.e., how many and which fish to harvest, how many and which fish to leave in the ocean).

In contrast, input-centered instruments essentially control what is used to take fish out of the ocean. Typical aspects of fishing that are managed by this class of instruments are days-at-sea, vessel length/width/tonnage, and gear restrictions. Although the number of active boats is just another dimension of inputs from the perspective of fish, it has the implication of turning regulated open access into regulated limited entry. "Closing the commons" (Hersoug 2005) may have considerable social side effects on employment, settlement, and the cultural landscape in general. Input regulations almost invariably lead to "effort creep" where fishermen substitute uncontrolled for controlled input. In the words of Wilen (1979, pp. 855–856) "we cannot necessarily simply limit 'effort' (a multidimensional notion) by, say, limiting tonnage or vessel numbers, or numbers of fishermen. With flexibility fishermen have the option to, and may, in fact, simply readjust other factors in their control to expand effort and subvert any imposed restrictions." This is also referred to as "capital stuffing" (Clark 2006), which is indeed a widespread empirical observation. On the other hand, as Crutchfield (1979, p. 746) notes: "The vessel is, after all, only a platform that carries harvesting equipment. There are obvious limitations on the extent to which additional capital investment … can increase catching power if key proxies for increased fishing power such as tonnage and length are constrained."

In spite of these limitations and drawbacks, input controls are often the easiest way to set an upper bound on what actually can be harvested. The informational needs for input-centered management are only moderate and this class of regulations provides flexible tools that can be adjusted to local circumstances. This makes them often the most practical management tools, especially in complex multi-species fisheries where the necessary information on biology and fleet structure is difficult to obtain. For example, the optimal harvest levels in a tropical multi-species fishery are often immensely hard to define and even harder to monitor (due to bounds on biological knowledge, technical ability, and institutional capacity). In contrast, a fisherman's mesh size and length of his boat is fairly easy to observe. At the same time, however, these instruments have only an indirect impact on the actual resource stock. They are therefore not able to directly protect the resource stock (Class I problem), and they (by themselves) also do not change the perverse incentive structure (Class II problem).

Tools based on financial incentives

Taxes increase the cost of catching a fish and essentially determine the point where taking out another fish from the sea is no longer worthwhile. Managing a fishery via taxes works therefore only indirectly, as it requires extensive information about the economic components of the system. These extensive informational requirements are definitely a disadvantage (Arnason 1990). Yet if there is sparse information about the biological components of the system, Weitzman (2002) has argued that managing by prices (i.e., taxes) may actually be preferable to managing by quantities (i.e., quotas). It seems counterintuitive to use a tax instead of, for example, a TAC if the state of the stock is unknown. However, this result is based on the assumption that taxes can dampen harvesting activity effectively by making it more expensive. Albeit, there is – to the best of our knowledge – not one fishery which is managed by taxes as a specific instrument to solve Class I and Class II problems. The reason seems that taxes are often deemed to be politically infeasible (Scott 1979, Johnson and Libecap 1982, Brown 2000). In the words of Munro and Scott (1985, p. 662): "Fishermen are not noted for their reticence in using any and all political power at their command."

Another class of tools that draws on financial incentives is buy-back programs. Calls for measures to reduce overcapacity are often heard in relation to the observation that harvesting capacity in global industrial fisheries grew at a rate eight times greater than the rate of growth of landings over the two decades 1970–1990 (Greboval and Munro 1999). Buy-back programs ensure that boat owners are paid to take their boat permanently out of the fishery. Although these programs may be favored by the industry, their potential to perform in practice is limited, to say the least (Holland *et al.* 1999). First of all, it will most likely be the oldest and least efficient vessels that will be decommissioned initially. Therefore, efficiency is likely to be enhanced (Class II), but effects on overexploitation will only be marginal (Class I). Second, owners will not withdraw unless sufficiently compensated, and in a limited entry fishery this implies granting boat-owners payments far above original vessel costs (Clark 2006). Both these arguments hint that an effective reduction of fishing capacity via buy-back is likely to be very expensive. But to make matters worse, such a program could actually lead to extremes in capacity build-up if it is anticipated by the fishermen (Clark *et al.* 2005). And finally, buy-back programs may be next to useless in a global perspective if vessels that are taken out of one fishery are simply sold to be used in another fishery, touching on the "flags of convenience" phenomenon that is known to support illegal, unreported, and unregulated fishing.

Finally, a fairly recent market-based approach is eco-labeling. Based on the widely successful introduction of "dolphin free" labels that signaled the use of tuna-catching gear that avoided mammal bycatch (Teisl *et al.* 2002), the goal is to improve the harvesting pattern by changing the structure of the demand side. Non-governmental organizations such as the Marine Stewardship Council (MSC) award their labels to fisheries that fulfill an in-depth set of criteria for sustainable fishing. However, to be successful, this approach necessitates a substantive product demand (Gardiner and Viswanathan 2004); when only 1–2% of the consumers are receptive to such a label, its impact will most likely remain negligible. Moreover, it is prone to the proliferation of self-serving labels that are issued by the industry itself after adhering to significantly lower standards (Jacquet and Pauly 2007). Finally, if the label is not tied to the specific use of harvesting techniques, the label might be perceived by fishermen as a price premium, which could lead to increased effort (Gudmundsson and Wessells 2000). Hence, eco-labels will not be very effective if not embedded in a broader management plan. Nevertheless, they may have a complementary character, not the least of which would be raising awareness about the issue of sustainable fisheries.

Tradable permits

Tradable permits are a special case of market instruments. Individuals are endowed with harvesting rights, such as a catch quota, that they own as property. These permits can be sold or bought from other holders. The existence of a market for harvesting rights is appealing for at least two reasons. First, most people are very sensitive to financial incentives, making market instruments very effective. Second, in the absence of market failures, any market will allocate resources most efficiently without any central intervention and informational requirements. The central idea is that the externality at the root of the "tragedy of the commons" should be overcome by giving clear and well-defined property rights to those that harvest (Hannesson 2004, Grafton *et al.* 2006). Establishing a market for these rights would then effectively separate the individual harvesting decision from the development of the fish stock (Arnason 1990). However, whether tradable permits can in fact achieve their promise is actively debated. In the remaining part of the section we will give an overview of the main arguments assessing whether individual transferable quotas (ITQs) will eradicate overcapacity and the low profits obtained in the fishing sector (Class II problem), and at the same time, decrease the pressure on overexploited fish stock (Class I problem).

While the notion of "clear" or "well-defined" property rights sounds good in theory, the practice is often much messier, making careful analysis necessary (Wilson 1982, Grafton 2000). In fact, property rights have several relevant dimensions, as pointed out by Schlager and Ostrom (1992, 1999). First, one may have the rights to enter a certain physical space, and extract resources. Second, one may hold the right to make management decisions, such as deciding to catch only fish above a certain size. Third, one may be able to exercise the right to enforce property rights by excluding others. Fourth, one may be able to transfer these property rights to a third party. Traditionally, economists favor an approach that ensures all of these rights, because this will maximize economic profits. The first three points make sure that the holder of the rights maximizes long-term benefits, while the last point ensures that the most efficient user will end up holding the rights. It is actually very difficult to come up with a policy tool that fulfills these criteria, since a necessary condition is that users take all consequences of harvesting into account so that the price for which one permit is traded in the market reflects the full value of the resource.

To this end, it has been proposed that fishermen be provided with a right to manage their own part of the stock, bearing the full consequences of their own exploitation decision. This idea – under the names "population stewardship right" (Gavaris 1996), "transferable dynamic stock rights" (Townsend 1995), or "virtual population units" (Lee and Gates 2007) – is indeed very appealing. However, as each fisherman would have to keep track of his own virtual stock, and the impact of his harvesting would have to attribute to the real overall stock development and recruitment, such a management tool is only feasible when there is full knowledge of the social and biological complexities. It is therefore unlikely this idea will become an available workhorse for managers reasonably soon.

A much simpler and already widely used management tool is the use of ITQs or "catch shares." These quotas give the exclusive right to harvest a certain amount of fish, but there is a wide variation in the actual implementation of this idea. In some fisheries, quotas are allocated to individuals by means of an annual auction. In others, the quota is tied to the fishing vessel, but the vessel may be bought or sold. Sometimes these quotas are issued in absolute values, but in most cases they are issued as a fraction of the total allowable catch.

Empirically, the track record of overcoming the race to fish by ITQs is indeed impressive (Grafton *et al.* 2006). For example, after ITQs were introduced in the North Pacific halibut fishery, the short season was lengthened to the whole year, with the effect that fresh fish was available for longer periods which resulted in considerable beneficial side effects (including much safer working conditions for the fishermen) in addition to more cost-effective harvesting (Homans and

Wilen 2005). Pinkerton and Edwards (2009), however, questioned the persistence of efficiency gains, mostly due to asymmetric information, imperfect capital markets and other market distortions.

Sometimes the fear is expressed that transferable quotas will end up in the hands of a few highly industrialized fishers and small, traditional boats will be driven out of the market. This is indeed likely to happen and it is important to understand that this is not a negative side effect of an ITQ, but the whole point of a transferable quota. Economic theory predicts that ITQs will most likely end up in the hands of the most efficient users and overcapacity will be reduced. In general, efficiency gains from ITQs will be higher compared with non-transferable quotas if there is more heterogeneity among fishing techniques and boats. But this can cause unintended consequences since the most economically efficient user may be the one whose harvesting efforts are most detrimental to the environment.

Moreover, the reallocation of fishing activity may create devastating effects on fishing communities and considerable political tensions (Helgason and Pálsson 1998, Pinkerton 2009). If society attaches cultural value to community life and small-scale family-owned fishing boats, the welfare losses could, in principle, be higher than any gains in efficiency. Another source of political tension relates to the duration of the quota. If the right to harvest is perpetual, the question of who exactly is the beneficiary becomes very important (Jentoft 2006). Selling quotas through auctions seems efficient and fair, but resistance from established fishermen can be expected to be very high. Individual transferable quotas that are given for free to incumbent, i.e., established fisherman will most likely be welcomed by the recipients. But it seems unfair to transfer perpetually to a small number of people, wealth that, in principle, belongs to the whole society (Bromley 2009). Dividing the pie today can also be unfair to future generations. Giving the quotas away for free may create additional perverse incentives, especially if it is based on current capacity, an often-heard suggestion. In anticipation of an ITQ system, fishermen may be willing to incur losses to increase their capacity now, given that they may be rewarded with a valuable quota. In Iceland, anticipated free ITQs based on catch history may have led to increased fishing in the period before the quotas were actually distributed (Haraldsson 2008). This undermines not only progress toward solving the Class II problem but also aggravates the Class I problem.

How do ITQs, in general, fare with respect to solving the Class I problem? Evidence seems to indicate that establishing ITQs indeed positively affects the long-term status of a stock (Grafton *et al.* 2006). Statistical analyses of 11 000 fisheries have indicated that the establishment of catch shares has reduced the probability of stock collapse (Costello *et al.* 2008, Heal and Schlenker 2008). It is, however, notoriously difficult to disentangle institutional and economic reactions and performance. It is not unlikely that a general awareness among stakeholders has led to a management change (establishing ITQs) and the reduced stock collapse is the direct result of the same awareness rather than the management change.

Also, the overall effect of ITQs on marine ecosystems is not unequivocal (Branch 2009). This may be due to a number of caveats: first, catch shares will not achieve efficiency when there are externalities (e.g., congestion of fishing spots) in the production process (Boyce 1992) or if the resource is of heterogeneous quality (Costello and Deacon 2007). Second, an incomplete coverage in terms of the principal target species may lead to a substitution of uncontrolled species for controlled ones (Grafton and McIlgorm 2009). Also, the related bycatch and discarding problem (Herrera 2005) may be substantial.

On a more profound level, the allocation of catch shares alone could, of course, only overcome the problem of overfishing if, and only if, the TAC is set correctly. Someone who holds the right to harvest a fixed amount of fish, or a fixed fraction of a TAC simply has no incentive to withdraw from that right. Sometimes the hope is expressed that ITQs will induce an expanded sense of stewardship on the part of the users (Grafton *et al.* 2006, Costello *et al.* 2008). The argument here is that an ITQ is a secure asset (like a share of a company) and if the fisheries collapse, the quota would be worthless. Therefore, ITQ owners will start caring about the state of the stock (their asset) and jointly agree on a lower quota. This view is probably overoptimistic, because the failure to reach consensus on what would be best for everyone (and especially, if the well-being of the ecosystem is a consideration) is exactly why most fisheries pose a social dilemma and are, hence, in crisis.

From a theoretical perspective, the fact that ITQs may reduce the number of users, because less efficient users leave the industry, may help crowding-*in* stewardship motives. It is likely that a smaller number of users will find it easier to reach consensus on reducing exploitation. But, as established in the previous section, material incentives often crowd-*out* voluntary stewardship motives. Thus, it is essentially an empirical question of whether or not the lower number of users outweighs this crowding-out effect. Individual transferable quotas may be especially detrimental because they give fishermen an unambiguous enforceable right to harvest a certain amount of fish. One may even argue that buying "rights to destroy nature" are akin to medieval indulgences (Robert 1994) and therefore quite the opposite of progress toward stewardship based on social motivation.

Summing up, it is clear that catch shares form an interesting group of management tools: they require regulatory activity in setting the overall harvesting limit. Organizing and distributing the individual rights occurs at a central level, while the trading and changing of incentive structure happens at the individual level. However, the natural conditions that allow for ITQ management (high level of predictability) seem to be fulfilled only in a narrow set of circumstances in marine fisheries. Given that evidence to support the contention that ITQs do indeed induce stewardship motives is sparse, it seems wise to not take any irreversible steps. This links particularly back to the question of "how" a specific fishery is managed; even if the natural pre-conditions for successful ITQ management are present, it is important not to destroy effective informal arrangements. In general, establishing ITQs will not be cheap, as any catch share management implies considerable management costs. At times these may be prohibitively high (Grafton and McIlgorm 2009). Moreover, as an efficient catch-share system is expected to generate considerable profits, the distribution of catch shares may cause considerable political tensions (Hannesson 2004, Clark 2006). Last, but not least, it is clear that catch shares will be no global solution: roughly 50% of the world's value from fisheries is taken from waters where either no single country has sufficient control to exclude other countries or where the country in question does not have the ability to institutionalize such a management scheme (Diekert *et al.* 2009b).

Part C: Policy recommendations for four stylized examples

Overfishing cannot be stopped with simple technical fixes (Degnbol *et al.* 2006). Neither Class I problems (the social and natural waste stemming from overstraining the replenishing potential of the resource), nor Class II problems (the social and natural waste which is the result of a perverse incentive structure brought about by the fact that fish can be turned into money only by the first person who catches it) will be solved by one instrument (see Fig. 5.2). Solving both simultaneously is even more complicated, if possible at all with current options. Remedies for overexploitation require first, and foremost, agreement on what a given ecosystem is capable of delivering, thus the need for an explicit management objective (Dankel *et al.* 2008). This objective and the tools intended for its attainment will only be perceived as legitimate and fair when all stakeholders have the possibility to influence the decision process. In particular, external "incentives that appeal to self-interest may fail when they undermine the moral values that lead people to act altruistically or in other public-spirited ways" (Bowles 2008). However, not only the social subsystem, but also the resource subsystem is of fundamental complexity. To achieve true ecosystem-based management the larger context within which these subsystems occur must be taken into account. Not only direct human-induced changes from resource use, but also natural changes to the resource's environment and its qualitative properties will have a profound impact on the resource dynamics and its variability.

The actual success of a given set of policies is thoroughly contingent on the specific circumstances (Sen 2009). Nevertheless, it is possible to broadly categorize different classes of biological and social settings that result in particular combinations of informational needs and transaction costs, and ultimately lead to different sets of policies that are recommendable.

The first example is a hypothetical small-scale coastal fishery where fishermen know each other and have social ties on several levels outside of their professional activity (e.g., religious or community organizations, etc.). Fishing is a way of life and is done mostly by traditional means. The fishery mainly targets an autonomous stock which is not systematically affected by factors outside the fishery. Such a fishery would lend itself to informal management as many communal ties already are firmly established and little formal interaction would be needed to secure sustainable fishing. Indeed, outside intervention in a top-down manner (e.g., in the form of official government controls) could be viewed as an illegitimate intervention and could lead to a crowding-out of stewardship incentives. However, applied measures that are easily observed and enforced by the community itself, such as gear restrictions or minimum market sizes, could signal best practice and help to maintain a cooperative equilibrium.

The second hypothetical example is a coastal fishery where fishermen may know each other but closer ties are confined to the professional level. Fishing is a way to make money and is pursued in a technologically advanced and industrial manner. The fishery is largely an autonomous stock which is not systematically affected by factors outside the fishery. Here, community management would be less effective, and such an industrialized fishery would lend itself better to market-based approaches such as ITQs. In fact, the technical efficacy of the fleet might make it necessary to externally control the amount of harvest in order to curb the Class I problem. Nonetheless it would still be instrumental to include fishermen and other stakeholders in management decisions, as this would significantly enhance the legitimacy of the overall TAC and other regulations. The latter would complement the ITQ system in order to minimize negative externalities.

The third example would also be a coastal fishery where fishermen may know each other but ties are again confined to the professional level. As in the previous example, fishing is a way to make money and is pursued in a technologically advanced and industrial manner. However, the fishery consists of many different fish species that can replace each other in the market but that are complementary in the water, constituting a complex ecosystem. In contrast to the second example, ITQs will be very costly in such a setting as they would have to involve most or all target species (to avoid substitution to uncontrolled species). To cope with the Class I problem, some form of limits on the volume of landing or on the amount of employed effort would still be needed. In addition, a temporal or spatial restriction on harvesting would be needed to protect the most vulnerable or productive parts of the system. Given the complexity of the resource(s), there would be a strong need for in-depth biological research. Again, stakeholder involvement in all stages of management and research would be crucial in order to enhance understanding and a sense of "ownership," thereby stimulating joint responsibility for the fishery.

The fourth example is a high-seas fishery, where individual fishermen do not know each other and fishing is highly industrialized and pursued internationally at a corporate level. The fishery consists of mainly one species, which is, however, highly migratory. Direct stakeholder participation will be very difficult in such a setting due to the distance separating them. At the same time, top-down management will be nearly possible as there is no single central enforcing agency for the high seas. On the other hand, international agreements on the most proximate and easily observable measures (such as gear restrictions) might be possible and protect the sustainability of the fishery (albeit at an inefficient level). Additionally, pressure from consumers (e.g., mediated via eco-labeling) might provide further incentives to fishermen to harvest in a sound manner.

In conclusion, sustainable fisheries management necessitates carefully identifying and disentangling all levels of biological and social complexity (Ostrom 2009). Management should be designed to avoid hidden assumptions and overlooked issues that result in unintended ramifications that sneak in through the back door. Moreover, it is crucial that the specific tools that are applied remain flexible and adaptable. It is, therefore, also very important to consider not only what is managed, but also how it is managed. It is mandatory that we account for how regulation is perceived and how it affects existing behavior based on incentives, social norms, or customs. The ideal would be a governance system where the objectives and tools are the result of a democratic involvement of all stakeholders. Yet, the fundamental challenge would be first, to set up the institutions necessary to keep such a system in place, and second, to make such a system robust to slow or sudden changes in the socio-economic (e.g., dominance by one interest group) or natural environment (e.g., climatic change).

It remains paramount to recognize that we cannot wait for all uncertainties to resolve before action is taken. Rather, we need to apply the appropriate available measures, by taking the salient biological features into account, bringing stakeholders on board, and then adapt management as the future unfolds.

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