The coevolution of renewable resources and institutions – Implications for policy design

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Abstract

This PhD thesis studies how natural renewable resources and institutions governing those resources mutually influence each other. Theoretical models are developed in which members of a small community have joint access to a common pool resource. We analyze under which circumstances social norms of cooperation evolve that effectively regulate resource exploitation, but also when those social norms break down, identifying obstacles for community governance. Furthermore, in the light of biological and social complexity this thesis analyzes how governmental policy should be designed if self-governance is not sufficient to protect the resource stock. The insights obtained are applied to the case of Arcto-Norwegian cod. An optimal management plan is developed that can be adapted to several policy objectives concerning the utilization of the fleet. In addition, management advice is given for the case that harvesting may trigger an evolutionary response of the fish stock.

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Introduction

1.1 Setting the stage

When I spent one summer in Oslo to write parts of this thesis, I was living in a shared house together with Anette, a musician, and Cecilia, a young student from Sweden who worked during summer vacations in Oslo. One evening I came home and saw Anette in the kitchen – she was furious: "Andries, I am so angry! Cecilia hasn't paid rent, she doesn't clean her dishes, and now I discovered that she also steals my food! I hate that Partysvensker[1]!" Seeing Anette so upset, it occurred to me that it was definitely not the right moment to tell her that I had finished all her coffee, even though I had already bought new one. If I told her now – in this context – she could have easily thought that I had misunderstood her story and thought that she was also mad at me. So, next day I told Anette very occasionally that I had bought new coffee. Anette looked at me and said "Andries, please don't think that I told you that story about Cecilia to make you stop taking my coffee. I didn't mean it in such a way." And I replied "Anette, that is exactly why I did not tell you yesterday. I did not want you to think that I would think that you are angry at me. Because I know you are not."

What can we learn from this little anecdote? It shows us that sharing a common resource is not always easy. In fact it can be quite difficult for several reasons. First, some individuals may use the resource at the expense of others and think only about their own short-term benefits. Especially if the individuals sharing a resource (i) do not really know each other, (ii) do not really care about each other, (iii) do spend only a short time sharing the resource, cooperative-minded individuals may be completely powerless against selfish individuals.

[1] Many young Swedish students spend their summers in Norway to work hard and earn good money, only to party even harder and spend all the money in excessive binge drinking. These so-called "Partysvensker" are often blamed for their disruptive effect on the courteous Norwegian society – in spite of the fact that Norwegian students do basically the same.

Second, it is not always immediately obvious what kind of behavior is appropriate. Why is drinking Anette's coffee OK, while eating Annette's pasta (which she had planned to eat for dinner) is not? Apparently what is acceptable and what not depends to a large extent on the characteristic of the resource. It makes a lot of sense to share coffee (as it tastes fresh the best) or bulk stuff that lasts long and does not cost much, salt for example. In contrast, one may want to keep the good bottle of red wine that one has always at home (just in case an unexpected visitor comes) or the portion of fresh pasta serving one person for dinner for oneself.

Third, the anecdote shows that social interactions are inherently complex. We observe and judge behavior of other people around us, and also reflect on our own behavior. But what makes human interactions really complicated is the fact that we also consider how our actions are perceived by others. This is one of the main points of the theory of the moral sentiments of Adam Smith (1759, section 3.1):

> *We suppose ourselves the spectators of our own behaviour, and endeavour to imagine what effect it would, in this light, produce upon us. This is the only looking-glass by which we can, in some measure, with the eyes of other people, scrutinize the propriety of our own conduct.*

One consequence of this social complexity is a high context-dependency of our actions. What may be perfectly acceptable to us in one social setting, may be unacceptable, or even embarrassing, in a different one. Most economic models tend to ignore this social complexity, mostly for convenience. The example of me drinking Anette's coffee was rather innocent, but there may be situations where the stakes are very high. For example, if many individuals have access to the same fishing grounds, severe overexploitation – or even extinction may occur. This thesis argues that social complexity should be taken into account when resource management and government policy is analyzed and designed. After all, any form of resource management is also about managing people, with the explicit or implicit aim to change the incentive structure of those people. This can only be done successfully, if the incentive structure is understood.

The next section gives a brief overview about examples of social dilemmas, as they may occur when a natural renewable resource, such as a fish stock, is jointly managed. Section 1.3 gives a very brief overview on the nature of different types of common pool resources. Section 1.4 introduces the concepts of social norms and social preferences, while section 1.5 introduces an evolutionary approach to social norms. Section 1.6 introduces us to the fascinating world of fisheries, a topic that is addressed towards the end of the thesis, and section 1.7 gives an overview of the different chapters in this thesis.

1.2 Social dilemmas in common pool resources

A social dilemma occurs if private interests do not coincide with collective interests. Typically, the actions of one agent give rise to an – unintended – externality that affects the payoff of other individuals. If agents do not take this externality into account, the outcome of their actions will be socially sub-optimal. Social dilemmas are ubiquitous in daily life and especially visible when a common pool resource is harvested. Very often simple metaphors, such as the Tragedy of the Commons (Hardin, 1968) are used to describe the anatomy of a social dilemma, in spite of the fact that the range of potential problems may be very broad, and fundamentally different (Kollock, 1998). In the case of a renewable resource, users face often more than one social dilemma, making the need to be precise even more important.

First, there is an intertemporal or dynamic externality associated with resource harvesting. This happens when the effects of current exploitation cause a decline of resource availability in the future (Wilson, 1982).

Second, there is very often a static externality as well associated with resource harvesting. One example is a situation where exploitation costs depend positively on the extraction efforts of all agents. This is sometimes called a "crowding cost" and may happen for several reasons. In fisheries, nets or vessels may congest (Clark, 1990, p.223), or it becomes increasingly costly to search for a good fishing spot.

Third, environmental costs are often not only born by all agents engaging in the same activity, but by all individuals living nearby or even far away. In agriculture, using a lake to support farming or cattle may cause lake eutrophication, putting a cost on all individuals who use the lake (Carpenter et al., 1999; Iwasa et al., 2007; Scheffer et al., 2000; Suzuki and Iwasa, 2009a). Cutting parts of a forest may lead to a direct cost by reducing the amount of ecosystem services provided by the deforested area (Cárdenas et al., 2000; Satake and Iwasa, 2006; Satake et al., 2007).

Most economic models assume environmental costs to be a positive strictly convex function of the level of economic activity or harvesting. This implies that average costs increase when economic activity is getting more severe. This assumption seems to be realistic for most situations. A good overview on these issues can be found in Burrows (1995) and Hanley et al. (1997, Ch. 2). It is not uncommon that both dynamic and static externalities are present at the same time. An intertemporal externality is particularly hard to overcome, because the future is uncertain, individuals are often slightly myopic and hold different time and risk preferences. Therefore, one can often observe that communities are able to reach a consensus on how to overcome the direct externality (e.g. taking turns in getting the best fishing spots rather than competing for them), but no consensus may be reached on the amount of fish that is extracted; see for instance Taylor (1987).

1.3 Common property regimes and common pool resources

In the economics literature, the terms common pool resources and common property regimes are often used interchangeably – possibly also because of the common abbreviation (CPR). In this thesis, a common pool resource will be defined as a type of good that delivers benefits to several individuals, where individuals cannot be excluded easily (otherwise the good would be private) and the average benefits decrease as the number of users increase (otherwise it would be a public good); see Ostrom (2003). A common property regime will be defined as a set of institutional arrangements, that defines the condition of access to and control over benefits, arising from a common pool resource; see Swallow (1995). Common property regimes are often implemented by small communities at the village level.

There is ample empirical evidence that self-regulation of communities can be effective in reducing overextraction of resources (Baland and Platteau, 1996; Bowles and Gintis, 2002; Casari and Plott, 2003; Coleman and Steed, 2009; Cordell and McKean, 1992; McCay and Acheson, 1987; Ostrom, 1990; Ostrom et al., 2002). Additionally, laboratory experiments have helped understanding under which conditions self-governance fosters and what help users overcoming a social dilemma (Carpenter, 2007; Castillo and Saysel, 2005; Fehr and Gächter, 2000; Janssen et al., 2010; Masclet et al., 2003; Milinski et al., 2006; Ostrom et al., 1994; 2009). Whether self-regulation is effective depends essentially on two factors. First, do the necessary social norms and rules exist that can – if followed by all users – preserve the natural resource? Second, if these social norms exist, are they obeyed and enforced by the users? The first question requires understanding the nature of the resource at stake. What may be a sustainable exploitation pattern for one resource may be very destructive for another. Furthermore, the likelihood that social norms for resource preservation evolve also depends on the resource characteristics. Schlager et al. (1994) identify stationarity and storage possibility as attributes that facilitate self-governance based on social norms. Both characteristics assure that benefits from delaying harvesting can be reaped at a later point. If resources are very mobile and cannot be stored (e.g. a highly migratory fish stock) the government should set some rules, as the local users are incapable of doing so. According to Swallow (1995), the most important motivations for successful common property regimes are risk pooling, economies of scale in exploiting the resource, equity in the distribution of the benefits and high transaction costs for central enforcement. In these cases, privatization – if possible – would solve the social dilemma, but the outcome may still be inferior compared to a common property regime. A good synthesis is given by Ostrom (2009), who identifies a common property regime to be more successful when: (i) the size of the resource system is moderate, (ii) the resource is neither too abundant, nor already exhausted, (iii) the resource 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.

1.4 Social norms and social preferences

A social norm is a customary rule of behavior that is self-reinforcing (Young, 2003). When following a norm leads to a strictly higher payoff than not doing so, there is no need for enforcement. When this is not the case, social norms are enforced through two mechanisms. The first mechanism can be summarized as social sanctions. Instruments that have been explored in the literature include peer–to–peer punishments (Fehr and Gächter, 2000; Fehr and Gächter, 2002; Gächter et al., 2008), peer–to–peer rewards (Vyrastekova and van Soest, 2008), verbal expressions of disagreement and discontent (Masclet et al., 2003), but also excluding individuals from profitable economic exchange (Milinski et al., 2002; Milinski et al., 2006), and direct ostracism (Vyrastekova and van Soest, 2007). In many cases these mechanisms are combined, and the mere threat of using them is often sufficient to induce cooperative behavior (Andreoni et al., 2003; Ostrom et al., 1994; Ostrom et al., 1992; Rockenbach and Milinski, 2006).

The second enforcement mechanism can be best described by a process of norm internalization (Young, 2008). A social norm is internalized when an individual feels obliged to obey it, even when not monitored. In many cases, it is the combination of sanctions and norm internalization that works hand in hand: an individual who has internalized a norm may be willing to bear significant costs to punish norm violators (Boyd et al., 2003; Hauert et al., 2007; Henrich and Boyd, 2001). This happens because an agent who has internalized a social norm does not only feel obliged to act in a certain way herself, but she expects others to follow that strategy as well. If these expectations are not met, certain emotions are triggered (Bicchieri, 2006). If other agents violate a norm, typical feelings are anger, indignation or disdain. Shame, guilt or embarrassments are widely experienced after own inappropriate behavior. According to Dolan (2002) these emotions provide “the principal currency in human relationships”. This has been pointed out earlier by Trivers (1971), who stated that “friendship, dislike, moralistic aggression, gratitude, sympathy, trust, suspicion, trustworthiness, aspects of guilt, and some form of dishonesty and hypocrisy can be explained as important adaptations to regulate the altruistic system.”

One of the social norms studied most is the notion of fairness. A large number of studies have shown that individuals make monetary sacrifices to improve the situation of agents that are less well-off. In such a case one could also speak about agents holding an equity preference, since individuals seem to prefer giving up some of their own payoff to obtain a more equitable outcome (Fehr and Fischbacher, 2002; Fehr and Schmidt, 2003). Again, agents usually expect that a sense of fairness is shared by other individuals, making such a preference conditional or reciprocal. This implies that agents do not have a preference for an equal outcome as such, but rather for fair play. In principle there is no difference between having a reciprocal or conditional preference for fairness and having internalized a fairness norm, since the consequences will be the same. This ambiguity is mainly due to the interdisciplinary nature of this issue. While the term internalized social norms (Scott, 1971) is very common in sociology and psychology, economists tend to work with preferences. An important difference is, however, a non-conditional social preference. In that case, an agent always prefers a more equitable outcome, irrespective of whether this is shared by other agents. Another semantic caveat is the fact that in the economic science, social norms are sometimes interpreted as equilibrium outcomes of a collective decision-making process. This overshadows a bit the notion that social norms are inherently dynamic and therefore an adaptive process (Ostrom, 2005b). In this thesis, social preferences are by definition held by single individuals, while the emerging patterns (preferences and decisions) at the group level are defined as social norms. In this micro-macro transition (Coleman, 1990), norms are part of a feedback process, as individual actions shape the norm, which in return influences individual choices.

Once a certain norm is established in a population, individuals that do not have internalized the norm tend to conform in order to avoid punishment or disapproval that may lead to a loss of social status (Bernheim, 1994). Thus, norm compliance helps explaining why social norms of cooperation are followed even if non-cooperation seems the more profitable choice. However, norm compliance alone does neither answer the question which social norms evolve, nor explain how beneficial such an evolving norm is (Boyd and Richerson, 2001). Only because a social norm is enforced does not automatically imply that the social norm is beneficial and makes the group better off. To understand the nature and dynamics of social norms, it is not enough to analyze whether they are stable once they are in place, but also how they got established – we need to understand the evolutionary process.

1.5 The evolution of social norms

Humans face in general internal and external constraints when making decisions. Herbert Simon (1955) pointed out that humans are not optimizers, but human choices can be better described as a process of adaptation and learning. Assuming choices to be optimal is, however, a good proxy if individuals learn quickly, problems are not too complicated and situations change only slowly (Simon, 1959). In fast changing, complex, or unfamiliar environments behavior is typically far from equilibrium and is, hence, not optimal. In those cases it is appropriate to take learning processes into account. Adding learning and evolution to classical game theory is a fruitful way for explaining human behavior (Goeree and Holt, 1999). Also social preferences (Sethi and Somanathan, 2001; Weibull and Salomonsson, 2006) can be understood as an evolutionary process. Preference formation is typically influenced by learning, culture and natural selection (Rogers, 1994). The outcome of this process is not so clear, because the environment is complex, and adaptation is not always quick. In such a case, as already noted by Simon in 1959, prediction about the behavior of an organism requires information about the speed with which it "adapted" to the environment and "moved towards its goal".

The emerging pattern of decisions and preferences determine the social norms that will be in place. According to Coleman (1990, p. 241) norms do not emerge randomly, but evolve for a reason. A good example is the case where a social norm helps solving a coordination or externality problem (Arrow, 1971). This point is not so clear-cut, since certain norms seem to make everybody worse off (Elster, 1989). Additionally, many norms that may look futile now may have been appropriate at the time they evolved. Likewise, a social norm that is unfavorable for most individuals, while beneficial for some may spread if the social consequences of conformity are strong enough. A social norm will always coevolve with its environment, which means that the social norm will be shaped by the environment and – in return – shape the environment. Environmental factors include the institutional environment, such as the overall culture or government policy, but also natural factors, such as natural resource characteristics or climate.

1.5.1 Evolutionary models

This section does not attempt to give a comprehensive overview of evolutionary models, but will only give the reader a hunch about which evolutionary models are used in this thesis. For a good overview on classical game theory, one may consult Fudenberg and Tirole (1991). Gintis (2000) has written a great and entertaining book that covers classical and evolutionary game theory. Probably the best introductory reading on evolutionary game theory is given by Nowak (2006). Also Weibull (1997) and Hofbauer and Sigmund (1998) are good and thorough readings in evolutionary game theory. Finally, a very nice book by Sigmund (2010) on the evolution of cooperation has recently appeared.

In economics, one of the most common ways to model evolutionary processes is through a set of coupled differential equations. In particular, the so-called replicator equation (Taylor and Jonker, 1978) is often used to model how the frequency of a fixed number of strategies changes over time. Its main advantage is the fact that it does not require any additional assumptions except that different strategies grow exponentially, where the growth rate depends on their payoffs, while the overall size of the community is fixed. This can be best illustrated with an example, which is based on Nowak (2006). In a community, individuals are either cooperative or they are not – let us call these non-cooperators defectors. The community comprises N individuals in total, of which C are cooperators and D are defectors. As a starting point it is assumed that the community size is not fixed, and both strategies simply grow exponentially dependent on their payoff π. Then the change in numbers of cooperators over time is given by $dC/dt = \pi_C C,$ while the number of defectors changes by $dD/dt = \pi_D D.$ If payoffs are positive, both groups would grow ad infinitum, which is not very plausible. Therefore, usually the restriction is imposed that the community is closed. Individuals can switch from one mode of behavior to the other, but the overall community size is fixed, which implies $N = C + D.$ In such a case the two strategies do not grow independently anymore, but their growth is coupled by an interaction term ϕ. The growth of the two strategies is now given by $dC / dt = C(\pi_C - \phi)$ and $dD / dt = D(\pi_D - \phi).$ The condition $N = C + D$ is only met if ϕ is equal to the average payoff in the population $\bar{\pi} = \pi_C C / N + \pi_D D / N;$ see Nowak (2006, p.15). Therefore, the growth of the two strategies can be expressed as $dC / dt = C(\pi_C - \bar{\pi})$ and $dD / dt = D(\pi_D - \bar{\pi}).$ This gives us the replicator equation as, for example, used in Noailly et al. (2003) and Sethi and Somanathan (1996). The main advantage of the replicator equation is the fact that the framework is fairly generic and differential equations can be analyzed very conveniently. Besides, it can be modified easily to allow for extensions. For example, it has often been suggested that not only diseases, but also habits, customs, social norms, beliefs, and even feelings are contagious and can be modeled through epidemiological models (Bettencourt et al., 2006; Cavalli-Sforza and Feldman, 1981). Since these models also make use of differential equations, they can be easily combined with the replicator equation. In chapter 3 we use a model, in which cooperation is contagious and spreads through personal interaction, but agents are also driven by payoff considerations, as is the case in the replicator equation.

While the simplicity of the replicator equation framework is certainly appealing, it has some important limitations. First of all, it requires a set of discrete strategies, which have to be imposed by the researcher. In the previous example it was assumed that the community comprises only cooperators and defectors, but we ruled out any other strategy, such as enforcers (who punish defectors). Mutations are usually not considered, and therefore new strategies do not arise by itself. Second, it is also problematic to allow individuals to choose mixed strategies, such as cooperate only half the time. An evolutionary stable mixed strategy, as defined by the Bishop-Cannings theorem (Bishop and Cannings, 1978) is a property at the population level, which implies that it is irrelevant whether a certain fraction of the population follows two pure strategies, or all individuals follow the same mixed strategies. This implies that it is impossible to distinguish a community, where everybody is a half-hearted cooperator from a community where half the members are full-hearted cooperators, while the other half consists of cold-blooded defectors. The fact that it is impossible to distinguish the population from the individual level is more than a slight inconvenience: it may also cause mixed strategies to be structurally unstable (Dieckmann and Metz, 2006).

A good modeling alternative is adaptive dynamics (Dieckmann and Law, 1996; Geritz et al., 1998; Hofbauer and Sigmund, 1990; Nowak and Sigmund, 1990), an analytical modeling tool that can conveniently analyze continuous strategies. While adaptive dynamics is widely used in biology, it has not fully reached the field of economics yet; for exceptions see Dercole et al. (2008) and Horan et al. (2008). Adaptive dynamics makes use of the assumption that different processes operate at different time scales. We will use this modeling framework in chapter 4 to analyze how social norms for resource harvesting evolve.

Akin to adaptive dynamics is what is known as the indirect evolutionary approach (Güth and Kliemt, 1998; Huck and Oechssler, 1999). Very much like adaptive dynamics it makes use of the assumption that economic decisions are changing quicker than preferences. Usually, it is assumed that choices are based on preferences, while only the material outcomes determine the evolutionary survival of a strategy. This is an interesting case, as it seems like a handshake between the self-interested homo economicus and his behavioral counterpart, who consistently makes wrong decisions – not because he is irrational, but because he is human (Morrissey and Marr, 1995). Nonetheless, assuming the success of a strategy to be solely determined by material payoff seems too restrictive and does not find empirical support (Güth et al., 2007). But we should not dismiss the indirect evolutionary framework as such. In principle, it is straightforward to modify the indirect evolutionary survival in such a way that status considerations (or any other form of non-material incentives) are part of the payoff function and determine whether a strategy is successful or not. Compared to adaptive dynamics, the biggest difference is that the indirect evolution approach usually looks only at equilibrium outcomes and not at the trajectories towards that equilibrium. Also, adaptive dynamics can distinguish whether a population converges towards one homogenous equilibrium (say half-hearted cooperators), or splits into a population that comprises several different strategies (say full defectors and full cooperators).

An additional interesting approach is evolution in finite populations (Hauert et al., 2007; Nowak et al., 2004; Taylor et al., 2004). It is not used in this thesis, but it will be briefly mentioned as it takes specifically into account the stochastic component of evolutionary processes. In small community sizes stochasticity plays a bigger role and inferior strategies may get established by accident.

Finally, agent-based simulations will be used (BenDor et al., 2009; Bousquet and Le Page, 2004; Grimm and Railsback, 2005; Tesfatsion and Judd, 2006). They lack analytical tractability, which is a disadvantage, but allow almost infinite complexity. Since they are based on rules and probabilities, they are ideal for complementing insights from analytical equation-based models. Agent-based (or individual-based) models are used in chapter 4 and chapter 6 of this thesis.

1.6 Fisheries

Globally, 80 percent of marine fish stocks are overexploited or maximally exploited (FAO, 2008). Symptoms of this overexploitation are declining fish stocks, overcapacity of fishing fleets, and consequently low profits. Hence, there is no doubt that the global fisheries are in crisis (Clark, 2006; Hilborn et al., 2003; Jackson et al., 2001; Worm et al., 2009). At the same time, it has been widely recognized that successful resource management must be case-specific (Beddington et al., 2007; Degnbol et al., 2006; Jentoft, 2006). While small communities have a key role in managing common pool resources sustainably, most large marine fisheries do not fit in this picture: The number of resource users is large, users come from different regions, or even countries, making it unlikely that a management regime will be set up without an active role of governments. This is especially the case for highly migratory fish stock, where those who restrain fishing effort will not necessarily be the ones who benefit from these protective actions (Berkes, 2006). Therefore, fisheries are often seen as a manifestation of Hardin's Tragedy of the Commons. This makes some form of government intervention often necessary.

In this thesis, particular attention will be given to Northeast Arctic cod, which is currently the world's largest cod stock and the main target species of the Norwegian fishing fleet. Chapter 5 investigates how Northeast Arctic cod could be optimally managed – taking into account several policy objectives. Chapter 6 looks at a particular interesting case of coevolution between institutions and renewable resources, as harvesting causes a genetic response in the exploited species. A recent study that summarized the magnitudes of phenotypic change in fish, ungulates, invertebrates, and plants found that harvesting may produce rates of evolution that is up to 300% larger 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 can be caused by intense harvesting (Jørgensen et al., 2007; Olsen et al., 2004 ; Stenseth and Dunlop, 2009).

Unlike in terrestrial ecosystems, it is almost impossible to define clear boundaries that separate different users in marine ecosystems. This leads to strong interactions between fishermen with different family or community backgrounds, and also diverse boat and gear types. As a result, the prospects for self-governance are often low, while the arising social complexities in fisheries are typically very high. A direct consequence of this social complexity is that individuals are embedded in a social context and decisions are context dependent (Granovetter, 1985). This implies that these decisions cannot be necessarily described by models assuming that agents are only self-interested (Jager et al., 2000; Ostrom, 2010). Instead, agents typically care about how their actions are perceived by others and may also be intrinsically motivated to keep up a certain self-image. This is the case in developing countries, as well as in developed countries, where fishing is often perceived to be an important element of community culture and identity (Ginkel, 2009; Henrich et al., 2001). As a consequence, fishermen are not always acting like profit maximizing entrepreneurs, but decisions about fishing techniques, such as choice of gear, equipment, boats, but also fishing locations and fishing periods are highly influenced by the social environment (Jentoft et al., 1998; Salas and Gaertner, 2004; Wilen et al., 2002). These social complexities have profound impacts on the success of any management regime and these factors will be discussed in chapter 7.

1.7 Overview of this thesis

We know that small communities have a salient role in managing renewable resources, but there are still important gaps of knowledge that this thesis addresses. First of all, even if we understand that humans do indeed act much more cooperatively than predicted by models with individuals motivated exclusively by monetary outcomes we are not quite sure about the exact mechanisms. Chapter 2 forms a bridge between the literature on the evolution of cooperation, resource management and design of environmental policy. That chapter also addresses how government policy influences any voluntary motivation to act as a good citizen. How can government policy make individuals feel more socially responsible? And when does government intervention make individuals feel decoupled from society? Chapter 3 and 4 both zoom in at the community level and try to unravel the mechanisms of social sanctions. In both chapters a small community jointly harvests a common pool resource. In chapter 3 cooperators try to convince selfish individuals that obeying a cooperative harvesting strategy is in everyone's benefit. Therefore, the cooperative spirit is contagious – at the same time individuals feel the rising temptation to defect as more individuals cooperate, because of the excessive profits that could be grabbed. While we expected these two forces to stabilize each other, we found something different –something very surprising. Chapter 4 does not a priori assume individuals to be either cooperators or defectors. Instead we do not impose any moral labels from the outside. Instead, a social preference to sanction peers evolves from a continuum of strategies, and we explicitly allow also for anti-social punishment. This form of punishment is targeted at individuals who are acting in society's interest and is therefore theoretically very difficult to explain. This model builds on the observation that institutions, such as a moral value system, change slower than economic decision. The results shed interesting lights on the question under which conditions a sanctioning mechanism evolves towards a socially optimal level. Along the way, we touch upon one of the biggest puzzles of Philosophy: The roots of morality. Chapter 5 is the first chapter that explicitly deals with fisheries. In that chapter we move away from informal institutions towards formal institutions. The question we try to answer how an optimal management plan for Northeast Arctic cod can be adapted to several policy choices. These choices include harvesting cod at minimal costs, using only a specific type of boat to minimize an adverse impact on the ecosystem, or having a diverse fleet for cultural considerations. Chapter 6 turns the concept of coevolution around. While we analyzed in chapters 3 and 4 how informal institutions evolve with the resource, here, it is finally time for the resource to strike back. Under which conditions does harvesting pressure lead to genetic change in the exploited fish stock? If it does, is such a change good or bad? What are the management implications for such evolving fish stock? Chapter 7 synthesizes the findings obtained so far and bridges social and biological complexity with policy design for the case of fisheries. We provide specific policy recommendations that can be readily implemented in many fisheries. Finally, chapter 8 provides a discussion on the methods used in this thesis and contrasts them with the literature. Besides, it is briefly addressed why political, but also scientific constraints make fisheries management extremely difficult.

2

Global environmental problems, voluntary action and government intervention

Abstract

The global community currently faces several very pressing environmental challenges. Governments are in the process of designing environmental policies to address these problems unilaterally, but also collectively (in the form of international agreements). Meanwhile, private citizens and firms are observed to take protective action voluntarily. Whereas standard game theory would predict that formal government intervention can only provide an extra stimulus for protective action, there are many examples of external interventions decreasing agents' propensity to undertake socially desired activities. This chapter provides an overview of the literature on the circumstances under which formal interventions can crowd out voluntary contributions to the common good. Furthermore, it is discussed how the effectiveness of government intervention may be improved by preserving the agents' intrinsic motivation to contribute to the common good.

This chapter is a slightly modified version of: Richter, A.P. and D.P. van Soest, 2011. *Global environmental problems, voluntary action and government intervention.* In Governing Global Environmental Commons: Institutions, Markets, Social Preferences and Political Games. E. Brousseau, T. Dedeurwaerdere, P.A. Jouvet and M. Willinger (eds). Oxford University Press, in press.

2.1 Introduction

Climate change, the depletion of (high sea) fisheries, and biodiversity loss rank high on the list of environmental problems the global community is confronted with. Over the past decade these problems have been addressed extensively in the media (at least in developed countries), and as a result firms and consumers are very much aware of their existence and also – albeit to a varying degree – of the necessity to take mitigating action.

Even if individual agents are well aware of these problems, it is not necessarily the case that they do take protective action. Environmental quality depends on the aggregate behavior of all agents on this planet, while each individual agent's actions have a negligible impact on any of the above global environmental problems. Indeed, all these problems are classic examples of the so-called "Tragedy of the Commons" as the benefits of protective actions are enjoyed by everyone, while the costs of taking them are private.

This analysis suggests that without government intervention, the prospects for mitigating or preventing climate change, fisheries depletion and biodiversity loss are bleak. However, casual observation of the behavior of people around us reveals that many people take at least some preventive action. When buying a new electric appliance people do take into account the energy consumption of the various competing brands; energy efficiency and amounts of CO_2 emitted per kilometer driven are among the criteria on the basis of which people choose a new car, and many households voluntarily separate their waste flows for recycling purposes. Also within the business community the concept of corporate social responsibility receives increasing attention.

These examples of private mitigation activities suggest that people are motivated to take protective environmental action even if extrinsic incentives – such as environmental taxes or quota – are absent. This raises two issues. The first is whether the amount of action taken is equal to the socially optimal level. And second, if more action is needed, how should government intervention be designed to bring preventive action to its socially optimal level?

The answer to the first question is easily given – for most environmental problems we cannot rely on the voluntary actions of consumers and firms alone. If we could, these environmental problems would have been solved a long time ago. But the second question is much more relevant, because of the following reason. In the standard game-theoretic framework – that is, assuming that all agents are exclusively self-interested – government intervention is always welfare enhancing: own-profit maximizing agents do not voluntarily contribute to the common good (if there are costs involved in doing so). However, because we observe that at least some people voluntarily undertake protective actions, the possibility arises that government intervention may be counter-effective, because it may result in crowding out of the agent's intrinsic motivation to contribute. And this is what this chapter aims to analyze; under what circumstances does government intervention strengthen ("crowd in") or weaken ("crowd out") the regulated agents' intrinsic motivation to act pro-socially, and under what circumstances is the change in behavior permanent, and when is it reversible?

The set-up of this chapter is as follows. In section 2.2 we provide an overview of the economic and psychological literature on the interaction between formal government intervention and the regulated agents' intrinsic motivation to contribute, resulting either in crowding in or out. In section 2.3 we turn to examples of crowding out in (real-world) environmental problems. Having established that crowding out is observed to occur in the real world, we analyze how this is related to the institutional setting (section 2.4). In section 2.5 we indicate the factors that may lead to crowding out. Section 2.6 shows how these factors effect formal and informal institutions. The corresponding policy recommendations are provided in section 2.7, and finally, section 2.8 concludes.

2.2 Intrinsic motivation and crowding out

One is said to be intrinsically motivated to perform an activity when one receives no apparent reward except the activity itself (Deci, 1971). Employees in firms may not be just motivated to work hard to capture incentive payments; they also take pride in their work. In a similar vein, people are willing to abstain from certain activities that are harmful to their environment, even if the private returns to these activities are positive (think of voluntary reducing the amount of waste produced by one's household, which requires effort to search for environmentally friendly products that are, in many cases, more expensive than other products).

The reason why the introduction of extrinsic incentives does not always result in increased provision of public goods is because the extrinsic reward may negatively affect an individual's intrinsic motivation 'to do the good'. This is best illustrated using Figure 2.1. In this figure, an agent's contribution to a public good is measured along the horizontal axis, while the vertical axis reflects the incentives provided by the government to induce the agent to act cooperatively (a subsidy per unit of contribution to the public good, or a tax per unit of pollution generated). If the agent is intrinsically motivated to contribute to the public good, her contributions will be non-zero even if there are no incentives provided by the government to do so. Suppose that this level is equal to point A in Figure 2.1. Now the standard line of reasoning in economics would be: "we do not know why the agent provides a non-zero contribution to the public good, but if the government provides incentives (in the form of a subsidy or a tax), the stimulus for the agent to contribute is larger and hence contributions will go up." Assuming that the agent's supply function is stable, the policy maker would expect that contributions would be increased to point B if the incentive (on the vertical axis) is set equal to, for example, s*.

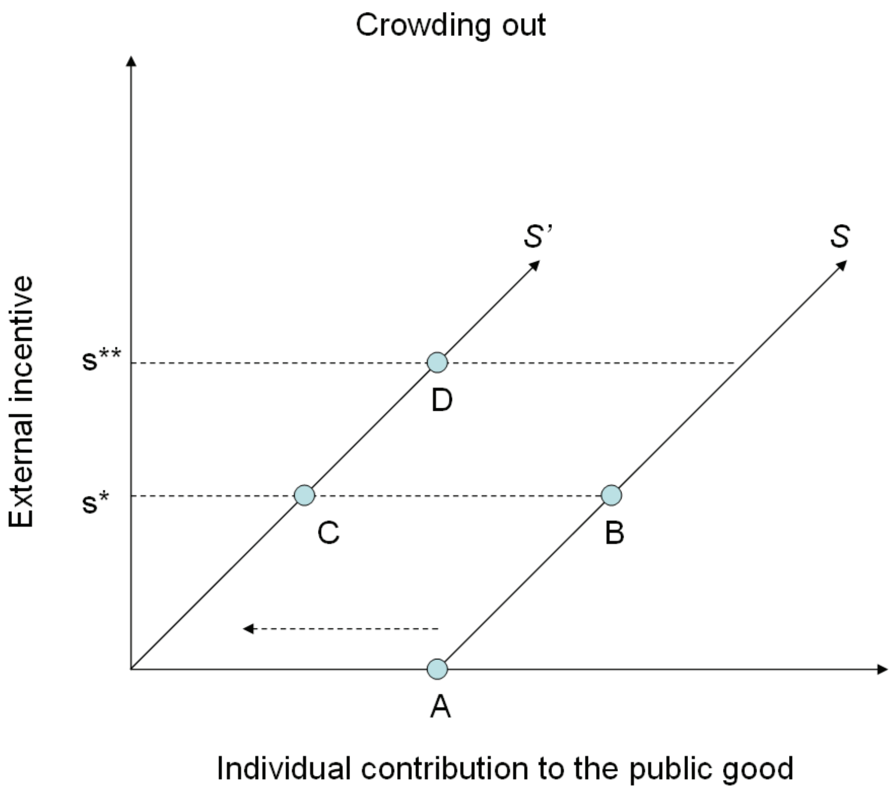

Figure 2.1. Crowding out occurs if the supply curve of an individual shifts to the left and the individual contributes less to the public good for a given external incentive

However, the assumption of a stable supply function is often not met in practice. External intervention may crowd out the intrinsic motivation to contribute, resulting in a leftward shift of the individual's supply function from *S* to, for example, *S'*. If so, setting the incentive equal to s* would result in contributions ending up in point C rather than in point B. The monetary incentive itself increases the marginal cost of shirking and/or increases the marginal benefit of good provision, but this does not only result in a movement along the original supply function (from A to B), but also in a leftward shift to point C. Hence, crowding out occurs because external incentives are not separable from intrinsic motivation (Bowles, 2008).

This simple graph suggests two things. First, government intervention is effective if the incentive is sufficiently strong, but it may be counter-effective if the external incentive is too small (see also Frey (1997, Ch. 11), and Nyborg and Rege, (2003)). In our example introducing s* results in contributions falling from A to C, but if s ≥ s**, contributions end up in D, or better. The second conclusion is that it is probably cheaper to use policy instruments that preserve the agent's intrinsic motivation to contribute; when preserving the agent's intrinsic motivation, the same level of contributions can be achieved at much lower incentives. This is because (i) the loss of intrinsic motivation constitutes a direct decrease in utility, and (ii) the enforcement costs are likely to be an increasing function of the level of stimulus provided.[2] And clearly government intervention would be even more effective if it were able to crowd *in* the agent's intrinsic motivation to contribute, inducing her supply curve *S* to move to the right. Hence, governments should be interested in finding out which policy instruments are likely to result in crowding out, and which are able to increase contributions by strengthening the agent's intrinsic motivation to contribute.

According to psychologists, crowding out is likely to occur if external intervention is perceived to reduce people's self-determination and/or their self-esteem because they feel that their involvement and competence is not appreciated (Frey and Jegen, 2001, p.8). But the extent to which crowding out occurs in actual practice, is hard to establish because of the many confounding factors. Kreps (1997) gives the example of the difficulty of identifying the existence of intrinsic motivation in the workplace. Employees may work long hours because they take pride in their work, but also because they respond to fuzzy extrinsic motivators including fear of discharge, scrutiny by fellow employees or even the desire for their coworkers' esteem.

Whereas strong empirical support of the stylized fact is hard to find, anecdotal evidence abounds. Children are willing to mow the lawn, but after they have been paid once to do it, they are only willing to do it again if they receive monetary compensation (Deci, 1971). The pharmaceutical company Merck decided to invest in developing an unprofitable drug against river blind-

[2] Note that this is even the case if the stimulus is provided in the form of taxes. For the same level of monitoring, higher taxes make tax evasion more profitable and hence enforcement needs to be increased. This is a welfare cost while the tax revenues themselves are a transfer from the agent to the government, and hence do not constitute a welfare gain.

ness in order to increase its workforce's intrinsic motivation for research (Murdock, 2002). Imposing a fine on parents arriving late to collect their children at day care increased the number of late-coming parents (Gneezy and Rustichini, 2000a). Small honoraria for seminar speakers may increase the probability of declining the invitation (Gneezy and Rustichini, 2000b). Unit pricing of unsorted waste does not necessarily result in a substantial increase in the amount of material offered for recycling (Ackerman, 1997; Berglund, 2006). Survey results suggest that people may actually decrease their contribution to voluntary work if a fee for non-participation is introduced (Brekke et al., 2003). Opposition against the implementation of "not in my backyard" (NIMBY) projects may increase when monetary compensation is offered to the local community. An example is the case with a mid-level radioactive nuclear waste repository in Switzerland (Frey and Oberholzer-Gee, 1997; Frey et al., 1996).

2.3 Crowding out in environmental problems

As Kreps (1997) argues, "abundant smoke signifies a fire, and the assertion is too strongly rooted in folk wisdom to be entirely hot air." While monetary incentives do not crowd out intrinsic motivation per se (Cameron and Pierce, 1994), it is by now well-established it can happen for a wide range of conditions (Bowles, 2008; Deci et al., 1999, 2001; Frey and Jegen, 2001; Frey and Stutzer, 2006; Gintis et al., 2005; Ostrom, 2005a; Vollan, 2008).

However, this list of instances of crowding out raises two questions. The first question concerns the circumstances under which crowding out is observed to take place. The examples presented are of a parent who wants his lawn mowed by his child, a boss of a firm who wants a project to be implemented, an owner of a day care centre who wants to induce the parents to pick up their kids on time, etc. These are all examples of so-called principal-agent problems where the principal has certain objectives for which he is dependent on the efforts of someone else (the agent) to have them achieved. In these situations the interests of the principal and the agent do not completely coincide. So how does this relate to environmental problems where there is not just one principal to be "served", but a larger group – or even society? Indeed, failing to take protective action in environmental problems results in damages accruing to a large and diverse group of agents (citizens, firms, or consumers) as is the case with, for example, overharvesting fisheries and failing to invest in water capture in arid regions. These environmental problems have in common that all agents involved would be better off if they collectively undertook protective action, but given that all others do so, it is in each agent's private interest not to do it.

The second question regarding the examples presented refers to the temporal nature of the change in motivation: if crowding out occurs, is this change permanent and (largely) irreversible, or is it just transitory?

Before addressing the issue of whether crowding out may occur in (global) environmental problems too, we investigate whether social interaction in multiplayer social dilemma situations is properly described by the neoclassical assumption of selfish actors. Here, social dilemmas are situations in which actions that maximize the sum of payoffs of all stakeholders do not coincide with the actions that maximize the payoffs of an individual agent, and vice versa. An example of such a situation is the fishery. After every fishing trip, the agent pockets the sales revenues of his harvests, but he bears only parts of the costs. When catching a fish one does not only remove that fish from the pond or sea, but also all offspring generated by that fish. If all fishermen decided to catch fewer fish, more offspring would be produced and all fishermen would be better off. However, given that all other fishermen restrained their fishing effort, each individual fisherman would profit from increasing his fishing effort. And similar considerations apply to other environmental problems such as global warming and biodiversity loss.

Whether or not agents are willing to contribute voluntarily to a public good is often difficult to observe in practice. Figures on individual catch, for example, are hard to obtain, and also it is difficult to establish what baseline level to compare actual catch with. One more easily observable act of voluntary cooperation is agents' propensity to discipline their peers. For example, Brazilian fishermen in the Bahia region destroy the nets of fellow fishermen who do not respect the catch quotas (Cordell and McKean, 1992). Sanctioning one's peers is an example of public good provision because there are often – hidden – costs associated with imposing punishments (for example because one exposes oneself to retaliation) while – if effective – the benefits of the punished individual restraining his fishing effort level accrues to all individuals having access to the fishery.

Because it is hard to establish to what extent observed pro-social behavior is the result of intrinsic motivation or because of other considerations (including the threat of social punishment or the presence of hidden rewards), economists and psychologists alike have studied this behavior by means of controlled experiments. In social laboratory experiments subjects are confronted with a (computerized) game that captures the essence of the decision problem the researcher wants to analyze (such as the fishery problem described above). Subjects are induced to think carefully about how to behave as decisions have actual financial consequences. The propensity of humans to act cooperatively has, among others, been studied in Public Goods (PG) games and in Common Pool Resource (CPR) games. In both games the costs of contributing to the common good is larger than the private benefits but smaller than the resulting increase in aggregate payoffs (i.e., the sum of payoffs of all group members). In the PG game the decision to be made is how much to contribute to a public good, where the associated benefits accrue to all members of the group. In the CPR game subjects need to decide whether they are willing to put in less than the privately optimal amount of harvesting effort in order to reduce the negative consequences of their catch on the payoffs of their peers. It is easy to see that homo economicus would act non-cooperatively, deciding not to contribute to the public good in the PG game and choosing the privately optimal extraction effort level in the CPR game. This prediction also holds in case of multiple rounds of interaction as long as the number of periods the game is played is finite (and subjects are informed about this). In such a finitely repeated setting, there is no reason to "invest" in trying to maintain cooperation in the last round (because there are no future decisions to be affected), and hence also not in the round before that (because there will be zero cooperation in the last round anyway). But that means that there is no reason to invest in maintaining cooperation in the second-but-last round either, and hence, on the basis of backward induction, the conclusion is that it does not pay to invest in even the first round. The typical pattern that emerges when PG games and CPR games are played over multiple rounds is that cooperation declines quite steeply over time (see for example Fehr and Gächter, (2000), and Ostrom et al., (1992)). This holds when university students are used as subjects, but it holds for a wide variety of other subject pools too. This suggests that humans are willing to act cooperatively, but that defection by others results in cooperation declining over time.

Interestingly, both Fehr and Gächter (2000) and Ostrom et al. (1992) also implemented treatments in which subjects can punish their peers for acting non-cooperatively. In these treatments, each round consists of two stages. The first is the "social dilemma stage" as described above (i.e., modeled in the form of either a PG game or a CPR game), and the second is the sanctioning stage. In the latter stage subjects can decrease the payoffs of any of their peers at positive costs to themselves. Game theory predicts that sanctioning would never occur because of a backward induction argument very similar to the one above. In the last period there is no reason to sanction because the punisher incurs costs while it is impossible to affect the punished individual's behavior because the game ends. Hence there is no incentive to act cooperatively in the first stage (the social dilemma stage) of the last round either. That means that there is no reason to punish one's peers in the second stage (the punishment stage) of the one-but-last round, which implies that there is also no incentive to act cooperatively in the first stage (the social dilemma stage) of that one-but-last round. Continuing reasoning backwards, the standard assumption of purely selfish players result in the prediction that there will be zero contributions to the common good and also no sanctioning in any of the rounds the game lasts.

The experimental evidence gathered by Ostrom et al. (1992) and Fehr and Gächter (2000) refute these predictions. Punishments do take place, and the threat of being sanctioned raises the level of cooperation almost to the level that maximizes group payoff. And similar results are found when self-regulation is by means of rewards (as opposed to punishments), although the results tend to be a little less strong (Vyrastekova and van Soest, 2008).

Given that humans (fishermen in the real world, student subjects in controlled economic experiments) act more cooperatively than predicted by standard economic theory, the question is to what extent formal government intervention can be counterproductive by crowding out the regulated agents' propensity to cooperate voluntarily. In the environmental economics literature several cases have been documented of crowding out occurring in environmental problems. One of the earliest examples is by Anderson and Lee (1986, p.690) who observe that "the suggestion that policies be implemented assuming that people will not comply with them has the potential for eroding social capital which depends on respect for the law". This claim has been corroborated by, among others, Hatcher et al. (2000) and Sutinen et al. (1999).

A second example of formal rules crowding out informal norms is provided by Cárdenas et al. (2000). In this study, experiments were run with people in rural Colombia who are confronted with a common pool problem in their daily life. The game used by Cárdenas et al. was a CPR game in which subjects were asked to decide how much timber to extract from a forest. The scenario presented was that harvesting had an adverse effect on water quality (as is actually the case in the study region), posing a cost on everyone in the group. The game was played first without any regulations, while at a later stage an extraction norm was introduced that was enforced by a mild probabilistic fine. Cárdenas et al. (2000) find that subjects reduce their extraction level after the regulation is introduced, but start extracting more aggressively after realizing that consequences are rather mild. Strikingly, in the last rounds, extraction levels were higher with the regulation than without. As a result, payoffs are significantly lower when individuals are confronted with a formal rule than in its absence; the weak official rule interacted with the internal norms of the subjects and crowded out their intrinsic motivation to cooperate. Therefore, it can be seen as a warning towards indiscriminately introducing regulatory intervention without a proper understanding of how it might undermine norms already operating in the field.

These examples suggest that crowding out can occur in multiperson environmental problems too, but the studies presented above do not answer the question whether crowding out is likely to be permanent or just transitory. Some evidence on the issue of the irreversibility of crowding out is available from Bouma et al. (2008). This study addresses the issue of farmers' propensity to invest in the construction (as well as the maintenance) of soil and water conservation structures. The study sites were five villages in three different watersheds in (predominantly arid) rural India. Investments in soil and water conservation structures (as well as in their maintenance) provide private and public benefits, and one can hypothesize that the same set of factors determines whether households invest in the structures themselves, or whether they put effort into maintaining them. In practice there is a crucial difference between the two activities, though. The investments themselves are highly subsidized (either by government organizations or by NGOs), whereas the maintenance activities are not. As expected, the households' propensity to invest depends on the extent to which they are dependent on income from agriculture, on the size of their land holdings, etc. Interestingly, the authors found that the decision to invest also depends on the community's amount of social capital. To measure social capital the authors use several proxies including social homogeneity (as given by the relative size of the largest caste in the community), but social capital was also quantified by having villagers participate in a simple game, a so-called Trust game. This game is played by two players and is set up as follows. The player that moves first, the investor, has to decide how to allocate an amount of money between him/herself and the second mover, the trustee. The investor can keep all money, give the total amount to the trustee, or anything in between. The amount of money sent – if any – is tripled by the experimenter, and given to the trustee. Then the trustee has to decide how to allocate the amount received between him/herself and the first mover, the investor. The trustee can keep all money, give the total amount to the trustee, or anything in between.

Behavior in the Trust game is expected to reflect social capital because the standard game theoretic prediction is that the investor will not send any money. The reason is that the trustee will not give back any money if he/she is purely selfish, and hence the investor will pocket the investment fund him/herself. Hence the Trust game provides two measures of social capital – trust/altruism as measured by the amount sent by the investor, and altruism/reciprocity as measured by the share returned by the trustee; see also Cox (2004).

Bouma et al. (2008) used straightforward regression analysis to analyze the villagers' propensity to invest in soil and water structures themselves, as well as in their maintenance. Explanatory variables included the various measures of social capital (social homogeneity, amount sent, and share returned) as well as a large vector of subject-specific and village-specific control variables. In the results of the regression explaining the villagers' propensity to invest in the structures themselves, only variables that reflect private stakes are significant (per capita land holdings, household size) and none of the "social capital" variables. In contrast, the "social capital" ones are the most significant variables explaining the propensity to undertake maintenance activities. This is salient because the investment activities are subsidized whereas the maintenance activities are not. Interestingly, the probability of a household contributing decreases substantially if there are maintenance funds that support material costs but that do not compensate individual effort.

These results reflect two things. First, it seems that the formal intervention by the government organizations and NGOs crowded out the households' propensity to voluntarily contribute to a public good – given that social capital indeed matters, as evidenced by its role in the maintenance activities. Second, it is noteworthy that this crowding out in the investment phase did not spill over to maintenance activities, suggesting that crowding out may be highly context-specific, but also does not result in permanent crowding out (as the maintenance activities obviously took place after the investments in the structures had been made).

2.4 Crowding out and the design of environmental policies

Having established that indeed formal intervention may be counterproductive in terms of resource conservation, the question arises whether government policies can be designed such that formal and informal institutions are mutually reinforcing. It seems that three institutional characteristics are especially important: (i) the extent to which the external intervention is perceived to be legitimate and adequate (or proportional), (ii) the extent to which participation is voluntary, and (iii) to what extent the institution is perceived to be supportive (rather than restrictive).

When an institution is perceived to be legitimate and fair, participants are much more inclined to obey the rules (Frey, 1997, Ch.6). A striking example supporting this finding comes from Danish fisheries, 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). Somanathan (1991) describes how state intervention in Central Himalaya "directly weakened villagers' incentives to allow regeneration and conserve forests". As a result, a well-functioning informal system based on social arrangements was crowded-out. Similar phenomena can be observed in many African societies (Vatn, 2007). This raises the question how legitimacy can be achieved. Sometimes it seems to be enough to convince the individuals about the usefulness of the rule and that obeying it is in everyone's interest (Rodriguez-Sickert et al., 2008). Reeson and Tisdell (2008), however, found that moral suasion does indeed promote cooperation, but only in the very short-term.

One way to achieve legitimacy is involving stakeholders in the process of designing formal institutions (Dankel, 2009; Hatcher et al., 2000; Jentoft et al., 1998). Such a participatory approach may build trust between users themselves, but also between users and central authorities. This may crowd in stewardship motives, and increase compliance. This form of co-management has the additional advantage that stakeholders possess important knowledge which may help crafting better institutions (Jentoft et al., 1998). An active dialogue between stakeholders and decision makers can also help identifying and overcoming potential conflicts of objectives and stakeholders (Dankel, 2009). Many economists are somewhat skeptical about involving stakeholders too closely in the process of designing institutions, as it gives them the possibility to seek rents (Bergland et al., 2002; Johnson and Libecap, 1982). This is indeed problematic, especially when certain stakeholders have a lot of political influence or resources to lobby for their interests. In many cases, the voices that shout loudest are most heard (Hatchard, 2005). One could overcome this by making stakeholders more responsible and accountable (Mikalsen and Jentoft, 2008). This is especially necessary when objectives of local users and the whole society are not congruent. For example, local users may be interested in having a well-functioning ecosystem (which ensures income in the future), while they do not necessarily care about biodiversity as such.

Regarding this, several economic experiments have been conducted to test whether user participation does indeed increase the effectiveness of the institution under consideration. In a laboratory setting, this can be tested by allowing regulated subjects to vote on the details of the enforcement institution's design. Voting serves a dual purpose. First, the voting outcome (for example based on a majority voting rule) affects the design of the institution, and hence its direct effectiveness. But voting outcomes also provide information about the intentions and preferences of the community's majority to effectively protect the resource and to maximize group payoff (as opposed to trying to non-cooperatively maximize one's individual payoff). Therefore, in voting experiments we can observe whether a group of people is able to find consensus on designing effective institutions, but also whether voting itself affects the compliance of the institution that has been agreed upon.

Sutter and Weck-Hannemann (2004) provide an example where a failure to obtain majority agreement for the socially optimal action is detrimental to social welfare. In their experimental study, subjects have the possibility to vote on a minimum contribution level to a public good, upon which they make their decisions about how much to contribute. When the group fails to achieve a majority vote in favor of the rule, contributions are significantly lower than in the treatment without. This makes intuitive sense because even though in both cases there are no binding rules, a failure to reach consensus reveals information about the lack of cooperativeness of the co-players.

Obviously, the consequences of not achieving a majority vote are even more detrimental if the voting outcome results in the abolishment of formal institutions, as is uncovered by Tyran and Feld (2006). In this study, subjects can vote on the level of a (deterministic) sanction in a public goods environment. As is the case in Sutter and Weck-Hannemann (2004), subjects tend to contribute significantly less (more) when the majority vote was against (in favor of) the presence of an enforcement institution empowered to impose fines on those who contribute less than a certain level.

Having established that introducing voting with respect to details of the enforcement institution's design can either improve or reduce welfare (and conservation) depending on the voting outcome, the question arises what factors determine voting behavior. Vyrastekova and van Soest try to answer this question in two related papers (Vyrastekova and van Soest (2003), and van Soest and Vyrastekova (2008)). In these two papers, subjects are allowed to vote on whether the enforcement institution should be provided with sufficient incentives to actively sanction excessive extraction, or not. More specifically, one subject was assigned to take the role as policy enforcer. The other subjects voted on whether or not the subject representing the enforcement institution is allowed to keep the fine revenues. If a majority votes against this, any collected fines are removed from the game. In this case, the enforcer is not expected to actively impose fines when observing violations of the formal rule because there are fixed costs associated with punishing. In this setting the weakly dominant strategy is to vote in favor of the enforcer receiving the fine revenues. In Vyrastekova and van Soest (2003) two treatments were compared. In the first treatment, the policy enforcer always receives the revenues of her sanctioning activity (i.e., the fines imposed on those resource users who extract more than is prescribed by a rule). In the other treatment, the enforcer is only allowed to keep the fines if the majority votes in favor of this, as described above. Vyrastekova and van Soest find support for the hypothesis that voting actually improves efficiency of resource use as compared to the treatment in which incentives are assigned exogenously. Casting their vote serves as a means for resource users to communicate their stance with respect to the need for reduced aggregate extraction. Conditional on a majority having voted in favor of implementing an appropriate incentive structure, the extraction behavior was significantly more cooperative in the voting treatment than in the treatment where the enforcer is always allowed to pocket the fine revenues.

In a companion paper, van Soest and Vyrastekova (2008) analyze to what extent actual voting outcomes depend on the characteristics of the enforcement institution. The specific characteristic they focus on is the probability that when engaging in enforcement, the institution is indeed able to successfully impose fines. Keeping the expected fine constant, they compared the impact of a 50% chance of conviction (and a specific fine level) on voting behavior to that in case of a 90% chance of conviction (and a lower fine level). In both cases, the weakly dominant strategy is to always vote in favor of the enforcer receiving the fine revenues, because of the arguments given above. Van Soest and Vyrastekova actually find marked differences between the 50% and 90% probability treatments. Whereas in the latter treatment resource users almost always vote in favor of the enforcer receiving the fines, a favorable majority voting outcome is achieved in less than 40% of the cases in the former treatment. These results are striking as they imply that trying to save on enforcement costs by reducing the probability of conviction (with a concomitant increase in the fine level such that the expected fine is kept constant) is hazardous if the enforcement institution's effectiveness is at least to some extent dependent on the support of the regulated individuals. If the intervention is insufficiently effective, intrinsic motivation to contribute to the public good is reduced and the regulated agents decide to vote against the government regulation. These findings suggest that individuals will not support an institution that is perceived to be unfair. A very similar study has been undertaken by Kosfeld et al. (2009), where individuals could choose to become member of a sanctioning institutions. The authors show formally that a likely equilibrium outcome will be that such an institution will be formed and efficiency will be enhanced. These findings have been corroborated in an experimental setting. This study showed that institution formation can be an effective tool for solving a social dilemma, but fairness issues can be serious obstacles, confirming the results obtained by van Soest and Vyrastekova (2008).

An interesting case arises when individuals can communicate with their peers regarding what they perceive to be appropriate behavior. An experimental regularity is that communication alone is often sufficient to promote cooperation, even if any agreements made are non-binding. In many instances the social pressure arising from "cheap talk" corrects behavior more successfully than a fine that could serve as a price. Even more surprising is the fact that voluntary participation can foster cooperation even without social pressure. Di Falco and van Rensburg (2008) analyze the effect of governmental subsidies on livestock farmers in Ireland. Farmers receive livestock premia based on the number of cattle, but they can choose to sign up for a rural environmental protection scheme (REPS) as well. The authors analyze the effect of both payments on cooperation, but also on conservation effort. While the livestock premia have no effect on cooperation and a negative effect on conservation, the payments from the REPS lead to more cooperation and higher conservation effort. This is remarkable as encouraging cooperation is not an explicit aim of the REPS. An open question remains whether the voluntary nature of the program makes users more cooperative, or just attracts users that have more cooperative attitudes. While it is well established that there are important feedbacks between institutions, preferences, and economic outcomes, further research is needed to identify the causal relationships between those elements.

2.5 The theoretical foundations of crowding out

In the previous section we have documented that crowding out is highly related to (i) the legitimacy of the institution and the level of involvement of the individuals, (ii) the voluntary nature of it, (iii) and the enforcement structure. While these are all properties of an institution, the mechanisms behind crowding out must be identified at the individual level. Microeconomic models that assume agents to be exclusively motivated by material interests are undoubtedly very useful, but they are not necessarily capable of describing behavior of the average person, who is concerned about his identity, embedded in social structures, and equipped with a moral compass. Even worse, "policies designed for self-interested citizens may undermine the moral sentiments", as Samuel Bowles (2008) has pointed out. Therefore, formal models of moral motivation help us to understand the interactions between extrinsic and intrinsic motivations, while taking into account the corresponding feedbacks between the individual and the institution.

The fact that voluntary contributions to public goods are so omnipresent suggests that individuals derive some benefit from it. This raises the question whether people care about the public good itself or whether they enjoy the act of giving. The first is sometimes referred to as "pure" or output-oriented altruism, while the second is referred to as "impure" or action-oriented altruism (Francois and Vlassopoulos, 2008). This form of altruism is "impure" because it is not the result that makes people happy, but the act of giving itself. Whether this makes the deed less altruistic is part of a lively ongoing debate[3], and may explain why some people find the term "altruistic" misleading and name it therefore pro-social or other-regarding behavior.

Behavior is always the result of preferences and beliefs, embedded in certain institutions (Bowles, 2003). One way to account for pro-social behavior is to assume that agents have "social preferences", such as inequity aversion or care about the payoff of other people in general (Fehr and Fischbacher, 2002). Some authors have criticized that explaining social behavior with social preferences is a tautology ; see for example Baland and Platteau (1996, Ch. 6). This is certainly a valid concern for any model with a limited strategy space and does of course also apply to models of moral constraints – if the researcher imposes them, it is no surprise that model outcomes reflect "moral behavior". One could overcome this problem by developing very flexible models that allow for a whole array of strategies, such as pro- or anti-social preferences. Nyborg and Rege (2003) analyze how different models of moral motivation, based on altruism, social norms, fairness considerations and conditional cooperation can explain crowding out. In the literature several mechanisms have been suggested that give rise to crowding out; cf. Bowles (2008). First, a loss of self-determination triggers some loss of motivation. We hypothesize that this is linked to the fact that humans may undertake voluntary action to signal their pro-social stance and trustworthiness. Second, incentives convey information which changes the beliefs, and hence, the choices of an agent. And finally, incentives change the context frame of a decision or trigger a complete preference change. Let us discuss each of them in more detail.

2.5.1 Crowding out and costly signals

Costly signaling theory suggests that behavior which seemingly fails the cost-benefit test, occurs because such behavior conveys reliable information from the sender to the receiver. Contributing to the 'common good' can improve one's reputation in the community (or one's social status), which may yield future benefits. In this view acting pro-socially is hence an investment in one's

[3] This is nicely illustrated in the American sitcom "Friends", in the following conversation between Joey and Phoebe. Joey: Look, there's no unselfish good deeds, sorry. Phoebe: Yes there are! There are totally good deeds that are selfless. Joey: Well, may I ask for one example? Phoebe: Yeah, it's... Y'know there's...no you may not! (Friends, Season 5, Episode 4)

reputation or self-image, which can be profitable because it gives cooperative individuals the possibility to identify each other in social interactions, thus avoiding being exploited by non-cooperators. Trust plays a crucial role in economic exchange (Fehr, 2009), and one's contributions to public goods can be interpreted as a signal that the person is likely to be trustworthy in bilateral exchange situations too. Dynamic models have been developed that show that investing in one's reputation or self-image can enhance one's long-term payoff and hence contributing to the common good can be rational after all (Brandt and Sigmund, 2005; Gintis et al., 2001; Nowak and Sigmund, 1998).

Costly signals have been attributed to crowding out before, though often implicitly as impaired expression possibility (Frey, 1997), but also explicitly (Posner, 2000a, 2000b; Smith and Bird, 2005). When this signal gets blurred one may as well stop investing in it. This can be illustrated with a simple example. Consider a population that consists of pure altruists, strategic altruists and selfish individuals that all contribute to a public good. In society, being an altruist is perceived to be a good thing, and it leads to a good reputation or high social status (at least within the group of the altruistically-minded individuals). The pure altruists care only about the result, i.e. the public good, but not about the social consequences, while selfish individuals are purely financially motivated. Strategic altruists contribute to the public good when it leads to higher social status and reputation. When no material incentives are attached to the provision of a public good, the selfish would free ride, while the pure and strategic altruists would contribute. When a material incentive is introduced, the selfish increase their contributions. This implies that altruists can no longer be distinguished from selfish individuals. This could be one reason for the strategic altruists, who are concerned about their reputation, to stop signaling their good intentions. The same would occur if, more realistically, individual preferences are determined by material interest, altruistic motivations and reputational or self-image concerns. Bénabou and Tirole (2006) have shown formally how such a model can explain several aspects of crowding out. These theoretical predictions have been confirmed not only in laboratory settings, but also in a field experiment a financial incentive increases the willingness to contribute to a good cause in private (i.e. when nobody is watching), while it actually decreases contributions in public when it is an observable signal (Ariely et al., 2009).

In reality, the benefits from a good self-image are not constant, but depend on the composition of the population. Janssen and Mendys-Kamphorst (2004) modeled a situation where individuals are either altruists or egoists and choose whether to contribute to a public good. Social rewards depend positively on the number of contributing cooperators and negatively on the number of selfish people. A financial incentive for contributing induces more selfish people to contribute and hence lowers the social reward for altruists. As a result, altruists may cease contributing and aggregate provision may decrease. Ideally, models are flexible enough to identify the factors that affect the change in the various individual preferences, but that can also explain changes in the composition of the population ; see chapter 4 of this thesis for such a model.

2.5.2 Crowding out and beliefs

The information content of an incentive is related to beliefs, which have often been attributed to the crowding out phenomenon. This is especially the case in traditional principal-agent settings, where contracts are usually incomplete, and an extrinsic reward or control could change the perceived nature of the task. One reason for this to occur is that the reward reveals that the task requires much more effort or is much less fun than previously thought (Bénabou and Tirole, 2003). Therefore, beliefs play a big role in explaining crowding out in bilateral interactions, and it is possible to extend this line of reasoning to the delivery of public goods. There is a lot of debate on the scope of civic duties and therefore people may update regularly what they are expected to do as committed citizens. When one is getting paid for donating blood one may infer that it is something one is not expected to do by default. Scientists, politicians or celebrities may accept to give a talk without any compensation, but after having received a honorarium a couple of times, they may think twice whether or not to give a free lecture. Brekke et al. (2003) have developed a model in which utility depends on leisure, the consumption of a private good, the consumption of a public good, and a self-image as a socially responsible person (given by how actions deviate from some socially desired effort level.) A sufficient extrinsic incentive will make individuals feel morally no longer obliged to contribute, and hence, contributions may go down when the unit value of leisure is higher than the unit value of private consumption.

In certain cases, beliefs are linked to the pro-social signals described in section 2.5.1. If people are conditionally cooperative, not trusting reveals information about the expected share of selfish individuals in the population, making conditional cooperators not cooperate (Sliwka, 2007). In a similar vein, Ellingsen and Johannesson (2008) show how crowding out can easily occur in bilateral interactions, where players are either altruistic or selfish. Utility depends on material payoffs and on the warm glow from giving, but also on how actions are perceived by the others. Actions depend on one's own type, on one's beliefs about the type of the people one is matched with and on their observed prior actions. A key mechanism is that one obtains a higher utility by being nice to a good person (i.e. an altruist), and hence signaling to be an altruist may pay back.

2.5.3 Crowding out and context-dependent preferences

In many situations behavior is observed to be context-dependent and takes place in a "decision frame" that "is controlled partly by the formulation of the problem and partly by the norms, habits, and personal characteristics of the decision maker" (Tversky and Kahneman, 1981). Evidence from social experiments and the field suggests that many social preferences are conditional (Cox et al., 2008; Fischbacher et al., 2001). For this reason many people play fair only as long as the opponent reciprocates. More generally, preferences depend on the situation the agent faces and also on the process that has led to the situation (Bowles, 2003). Situation-dependent preferences are not unique for social preferences, as choices are always the result of given preferences in a certain environment. Process-regarding preferences, however, are special in the sense that they do not depend on the outcome only, but also on the chain of events that led to this outcome (Ben-Ner and Putterman, 1998). People may be reluctant to help someone who took some foolish decisions that brought him into trouble while they do help someone who was just extremely unlucky or unfortunate. Thus, one may conclude that preferences are higher-dimensional.

While many studies indicate the multi-dimensional nature of preferences verbally, they are hardly used in formal analyses, as the results may be rather complex. Assuming preferences to be higher-dimensional implies that the corresponding equilibria are higher-dimensional as well. When an individual stops cooperating, we may be inclined to detect a preference change. This may be the correct inference in some cases but not in all. An alternative explanation is that preferences are stable but the environmental context has changed, thus resulting in the agent changing his/her behavior. That means that people may decrease their contributions to the public good in response to the introduction of formal government intervention if they interpret the regulation as reflecting a lack of trust (Fehr and Fischbacher, 2005).

Laboratory experiments are useful for unraveling many of these situations. One example is the fact that players make different choices when they face a human opponent or a computer. Another frequent observation is that cooperative individuals cease cooperating after having been exploited by defectors: Once bitten, twice shy. What looks like a true preference change may just be a change in behavior given that a subject finds herself in a less cooperative environment than expected or believed.

This implies that it is very difficult to identify a true preference change. This is illustrated in Figure 2.2, which plots an individual's contribution to the public good on the horizontal axis against the sum of contributions of his/her fellow group members on the vertical axis. In this stylized example, contributing to public goods is conditional, i.e. it depends on the number of people who do so as well.

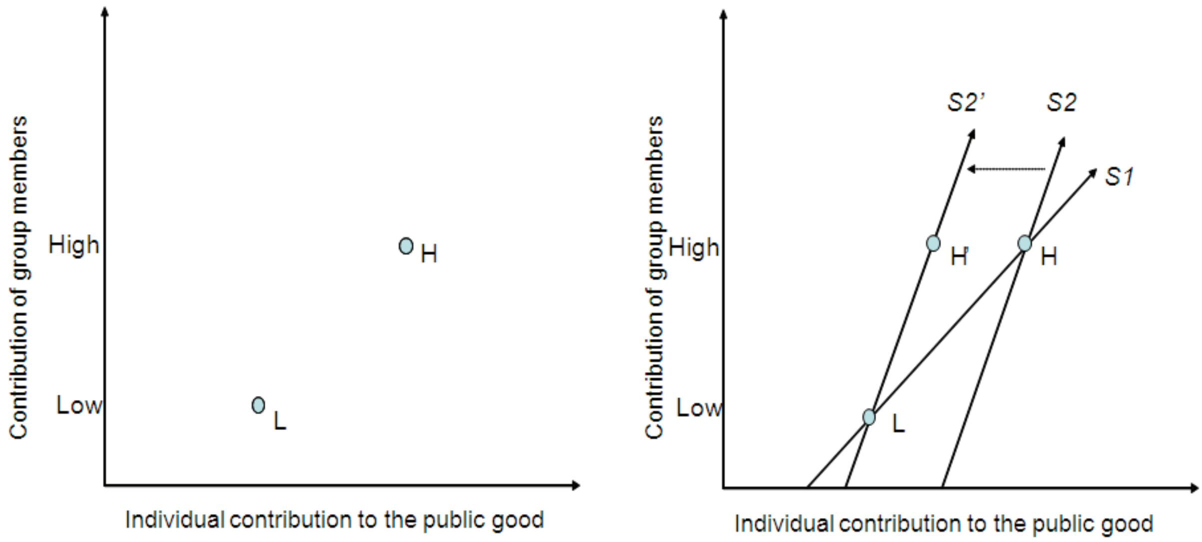

Figure 2.2. Reversible and irreversible changes in behavior. The left panel shows the observed contribution to a public good. The right panel illustrates that this information is not sufficient to distinguish a shift in the supply curve from a contribution change along a stable supply curve.

Suppose that an experimenter observes a high level of cooperation by both the individual and by the rest of the group (say point H) in the early stage of a game; see the left-hand side panel of Figure 2.2. During the game, the sum of contributions by the rest of the group (on the vertical axis) decreases and the individual agent reduces his/her cooperation too, resulting in a move to point L. The experimenter thus observes the shift from point H to point L, but she cannot determine whether this is change in behavior is only a response to a change in the environment, or whether the individual is so frustrated that his willingness to contribute in general has changed; see the right-hand side panel in Figure 2.2. The first situation is represented by a shift along the supply curve *S1*, where one could speak of a reversible change. If the number of cooperators returns to the initial high level, individual contribution will also be high again. An irreversible change would occur if the supply curve shifts to the left, from *S2* to *S2'*. This implies that a return to the initial level of cooperation in the group will not be sufficient to restore the initial level of contribution, because the new level will be at H'. Note that observing a drop in cooperation, followed by lower individual contribution is not sufficient to recognize which situation applies, because one does not know the slope of the supply function. To be able to do so, one would need sufficiently detailed data for single individuals, and that is typically not available. In real-world behavior it is even more difficult to distinguish temporary effect from a long-lasting preference change.

Another way of capturing the context-dependency of decisions is to assume that individuals hold multiple preferences that are weighted according to the situation. A good example is the model by Nyborg (2000) where individuals hold a preference as consumers, but also as good citizens. Individuals apply different preferences in different contexts. Therefore, choices depend highly on the situation. It is a priori unclear how consumers weigh these different preferences. In any case, an external intervention may lead to a shifting of weights. This is especially relevant when intervention influences the perception of a social dilemma. If an individual faces several co-players that are "in the same boat" she may be more inclined to infer some form of Kantian moral imperative: "if everyone cooperates, we will all be better off." If, however, a government regulator is imposing some law, the agent probably views the game very differently. The stage is now completely set, so her actions will most likely neither affect her future payoff, nor the rules set by the regulator. Therefore she would probably try to get the best out of the situation, given the presence of the regulator.

2.6 Repercussions on institutional constraints

In this section we will map these microeconomic fundamentals to the experimental results and try to understand its implications for institutional design. As mentioned before, we have identified three characteristics of an institution that seem to matter: Legitimacy, voluntariness, and the enforcement structure. Legitimacy is highly linked to context-dependency, but also to the endogeneity of preferences. When the institution lacks legitimacy, individuals may infer that being cooperative does not pay off, or even lose all confidence in cooperative behavior altogether. Note that it matters whether the institution is perceived to be legitimate, not whether it is legitimate by objective standards. Raakjær Nielsen and Mathiesen (2003) present a case from Danish fisheries, where rule-compliance went down after the fishermen felt decoupled from the decision-making process.

Fairness issues should be taken seriously, as they may not only influence the distribution of rents, but also the success of the institution, and therefore, its efficiency. In South Africa the government intended to reduce illegal fish landings by establishing de jure rights for the local fishermen. Some fishermen had the feeling that the process was not very fair and expressed their discontent by "protest fishing" (Hauck, 2008).

The voluntary nature of an institution is important for three reasons. First, it reveals important information about the intentions of other individuals. Second, it gives individuals the possibility to signal their social attitude or build up a good image or reputation. These can only work when individuals have actually the choice to do so, otherwise it would be impossible to distinguish true signals or images from forced ones. Because of this loss of information, individuals may as well cease investing in a social image at all. The third reason why a voluntary institution may perform better than a compulsory one, has been pointed out by Hauert et al. (2007). When individuals have the possibility to withdraw from some joint activity, it is much harder for others to take advantage of them. Therefore, self-determination is a route to escape being exploited by defectors. An example where a new law crowded out existing norms of reciprocity is presented by Borges and Irlenbusch (2007). When the German government introduced a law that made it possible to return any product just bought (for example via the internet), the number of products returned upon purchase increased sharply. This happened in spite of the fact that most sellers offered the same refund possibility even before the introduction of the law.

Concerning the importance of the enforcement structure, it is difficult to pinpoint the exact micro-foundations, as many mechanisms are at work, making the interaction between incentives and enforcement a complex one. Chhatre and Agrawal (2008) analyzed 152 forests that were common property in 9 countries. They found, as expected, that the more valuable forests were depleted faster when local enforcement is absent. Interestingly, in the presence of enforcement the opposite holds, as regeneration is higher when forests are more valuable.

It seems to be important to distinguish enforcement mechanisms that are centralized from decentralized ones. Most experiments focus on the role of punishments and rewards in decentralized peer-to-peer enforcement. While most studies show that peer-to-peer enforcement is very effective in inducing cooperation, its impact on welfare is more ambiguous (Egas and Riedl, 2008). Dreber (2008) found that costly punishment raises cooperation, but the cost may be so high that the community as a whole may be worse off and therefore "winners don't punish". This finding has been challenged by Gächter (2008) who found that punishment pays off, but only in the long run. Nikiforakis (2008) finds that adding a second stage of peer-enforcement may reduce all efficiency gains. This happens, because given the possibility of counter-punishments, most players do not punish in the first place, because they fear retaliation. The same breakdown can be observed when rewards are used as enforcement mechanism, albeit for different reasons. A second stage of rewarding gives defectors the chance to build a profitable rewarding network (Stoop et al., 2008). Therefore, both mechanisms seem to have their drawbacks, as the desire to punish non-cooperators can be very resource consuming, while rewarding or reputation does not always help sanctioning the bad guys. Rockenbach and Milinski (2006) have looked at this issue in detail, and concluded that it is the combination of punishment and rewarding based on reputation performs best in an experimental setting. Ohtsuki et al. (2009) have analyzed a formal model where two players of the population randomly meet and can either cooperate, defect, or punish. Individuals have a good reputation, or a bad one. There is, however, the chance that someone mistakenly identifies a partner as good when he is bad or the other way round. They find that when the probability to correctly identify someone is high, defecting with bad guys is the best strategy, while one should cooperate with the good ones. When the probability of correctly identifying someone is low, always defecting is superior. When the probability is in a very narrow parameters space between these two cases, punishing defectors and cooperating with cooperators is the winning strategy. This may indicate that increasing information (either through monitoring or gossip) makes punishment redundant, provided that one can actually refuse to interact with bad people. If one cannot, as typically the case in a common pool problem, the answer is less straightforward. In such a case, an appropriate defense mechanism against defectors is needed, be it either punishment or ostracism.

2.7 Recommendations for policy design

When a central authority sets an external incentive, crowding out may occur, with potentially very costly consequences. Several fairly simple recommendations can be made to avoid this unintended loss of social capital.

First, evidence suggests that individuals are able to form institutions and enforce them successfully. Communication and monitoring are important mechanisms for enforcement, and often gossip suffices as an enforcement instrument. If this is not the case, and more drastic punishments are needed, the welfare effects are more ambiguous. Therefore, lack of communication or monitoring possibilities can form an obstacle to the evolution of effective social norms. In such a case, formal institutions may be more efficient. When a central authority steps in, lack of involvement, but also fairness considerations can be an important reason why individuals actively try to undermine the institution. Including individuals in the rule designing process is often a solution, but may have drawbacks sometimes. This is especially the case when some individuals perceive the institution to be unfair, no consensus is reached or individuals who were opposing the rule feel not committed to follow it. Sometimes the social norms in place are maladaptive, because the environment or the technology has changed, while the social norms have not (Posner, 1996). In such a case, government intervention may be necessary, but should be done very carefully. Any attempt to manage or regulate social norms may backfire, because the authority is perceived to be part of the "game" (Posner, 1998). In the same vein, any attempt made by a central authority to strengthen existing social norms or "invest in social capital" may be well intended, but may have unpredictable consequences.

Financial incentives are problematic because individuals take into account that this will affect their (self)-image. Incentives in the form of public goods could be a solution, because individuals can then signal that they have a pro-social attitude. Indeed, provision of compensation in the form of publicly provided goods are more effective in increasing support for "not in my backyard" projects, such as a noisy road, than money (Mansfield et al., 2002).

Once the government decides to impose an external incentive, it is important to identify the nature of the product/task that is targeted. Subsidizing a good that has signaling character, like a hybrid car, may be counter-effective as pointed out by Ariely et al. (2009). If a small symbolic tax tries to underline the fact that a certain behavior is unacceptable, this may work especially well with goods that have signaling character, as the following example shows. After introducing a small tax on plastic bags in Ireland, using them became highly stigmatized and usage dropped by 94% (Bowles, 2008).

In general, it is crucial how the external intervention is perceived. In the famous child care study by Gneezy and Rustichini (2000a), a small fine was perceived as a price that parents were more than happy to pay. Therefore, the conventional economic wisdom of "getting the incentives right" may only work as intended if governments send an unambiguous message. In that sense, tradable emission rights, for instance, may be especially susceptible to crowding out, because the owner holds the "right" to do something, taking away the negative connotation from polluting (Frey and Stutzer, 2006). This seems to be relevant as well in the recent discussion on the success of individual transferable quotas (ITQ) in fisheries (Costello et al., 2008). When social issues are important, as is typically the case in small-scale fisheries, ITQs may undermine local stewardship (Ban et al., 2009). In the worst case, ITQ regimes crowd out ecosystem responsibility of the ITQ holders (who hold the "right" to fish), while leading to "protest fishing" among the non-holders, see also Chapter 7 of this thesis.

2.8 Conclusions

We can neither rely on external regulation, nor on voluntary actions alone to solve many environmental problems. While there is not one standard recipe for solving diverse social dilemmas, several regularities that determine success or failure of policy design have been identified. Experimental and theoretical work has shown that decentralized arrangements, based on voluntary action, communication, peer control, and reputation can be very effective, but also highly fragile. As a warning, it must be emphasized that imposing external interventions may not strengthen these arrangements, but replace them, leading to crowding out and a loss of social capital. Even when a certain regulation is established to formalize a certain right that de facto already exists, surprises may occur. In any case, governments need to be aware that institutional changes imposed may lead to unintended consequences that will be very difficult to reverse.

3

Contagious cooperation, temptation and ecosystem collapse

Abstract

Real world observations suggest that social norms of cooperation can be effective in overcoming a social dilemma, as it may occur in the joint management of a common pool resource, but also that they can be subject to slow erosion and even to sudden collapse. Using differential equations, we model a small community harvesting a renewable natural resource. The diffusion of cooperative harvesting norms takes place via interpersonal relations, while individual agents face the temptation of higher profits by overexploiting the resource. We show that a collapse of the social-ecological system easily occurs if agents are endowed with a finite amount of time that can be maximally used for resource harvesting. We explore the underlying mechanisms by analyzing how a catastrophic transition from cooperation to norm violation resulting in a resource collapse is affected by changes in key parameters, including the severity of negative externalities from resource harvesting, the size of the community, and the rate of technological progress in resource harvesting.

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3.1 Introduction

The history of mankind is one of gradual change in environmental quality and natural resource abundance, punctuated with sudden collapses of populations, species, ecosystems, and sometimes even of entire civilizations (Diamond, 2005; Scheffer, 2009; Taylor, 2009). Examples include the fall of the Maya empire and the collapse of the human population on Easter Island following the depletion of forest resources (Bahn and Flenley, 1992; Brander and Taylor, 1998; Diamond, 2005; Lentz and Hockaday, 2009; Scheffer, 2009). While system collapse is often the result of the interplay between natural resource regeneration and the social-economic system driving resource use, most of the research aimed at explaining its underlying mechanisms has focused on the former, with special emphasis on the existence of non-linear relationships in the dynamics of renewable natural resources. Examples of systems characterized by non-linear dynamics are those which feature a minimum population size below which extinction is inevitable (because of genetic degeneration, or because of increased difficulties to find a potential mate (Berck, 1979; Gould, 1972; Van Kooten and Bulte, 2000, Ch. 7)), but also those with complex interactions between the various components of the ecological system as is the case in, for example, shallow lakes (Mäler et al., 2003; Peterson et al., 2003; Scheffer, 1998) and grazing systems in semi-arid ecosystems (Anderies et al., 2002; Janssen et al., 2004; Kéfi et al., 2007). The non-linearities in the regeneration functions typically give rise to the prediction that continued overharvesting of the resource results in a gradual demise of the resource until a threshold – or tipping point – in the ecological system is reached, beyond which collapse is inevitable and where subsequent system restoration is very costly – if not impossible; cf. Scheffer et al. (2001).

In this chapter we contribute to the literature on tipping points in social-ecological systems by analyzing how social interactions between resource users affect a system's resilience. Following the literature on economic cooperation in social dilemmas (Bischi et al., 2004; Bulte and Horan, 2010; Noailly et al., 2003; Osés-Eraso and Viladrich-Grau, 2007; Sethi and Somanathan, 1996) we develop a model in which a finite number of community members have access to a commonly owned renewable resource. As is the case in the real world, we assume that the common property regime is such that community members are allowed to access and harvest the resource. However, it not allowed to hire non-community members to engage in resource harvesting. Therefore, each agent is endowed with a binding time constraint. Next, the natural regeneration function of the resource is modeled as a standard logistic growth function (Verhulst, 1838), and community members can decide to act cooperatively in harvesting the resource by limiting extraction, or not. Agents are tempted to act non-cooperatively, also referred to as defecting, because of the associated higher profits, but we also allow for the possibility that whenever a cooperator and a defector meet, the cooperator may convince the defector of the social desirability of acting cooperatively. The diffusion of social norms regarding harvesting is thus assumed to take place via interpersonal relations, with cooperation being "contagious" with a certain probability: when an encounter takes place, the cooperator makes a moral appeal to the defector to start acting in the community's interest, and the appeal may or may not be successful. This modeling approach is consistent with the effectiveness of verbal expressions of discontent in inducing and sustaining cooperation in social dilemma situations as observed in laboratory experiments (Masclet et al., 2003), but the mechanism can also reflect the use of self-regulatory instruments like peer-to-peer sanctions or rewards (Fehr and Gächter, 2000; Fehr and Gächter, 2002; Gächter et al., 2008; Janssen et al., 2010; Rand et al., 2009). Our model thus combines the literature on the evolution of social norms for resource harvesting (Ostrom, 2000; Richerson et al., 2003) with insights from that on contagious behavior (Cavalli-Sforza and Feldman, 1981; Dodds and Watts, 2005; Heal and Kunreuther, 2010; Lopez-Pintado and Watts, 2008; Young, 2009).

Unlike most studies of this kind, we do not assume any non-linearities in the dynamics of the ecological system or the social-economic system that, by themselves, give rise to multiple equilibria. If, for whatever reason, the number of cooperators increases, the social pressure on defectors to become cooperators increases, but the temptation to defect increases as well because the benefits of free-riding are larger too. And the opposite holds if there is a sudden, exogenous decrease in the number of cooperators. The social pressure on non-cooperators decreases, but the rents from defecting are dissipating too.

However, our system can still generate positive feedbacks between the two systems, giving rise to multiple equilibria, where the "good equilibrium" can be very resilient to exogenous shocks or external developments (such as exogenous technological progress in harvesting), while the same small shocks or developments cause the social-ecological system to collapse if the parameters are close enough to a critical threshold. This is a result of the positive feedbacks emerging because of the interaction of the social-economic and the ecological systems – these feedbacks are driven by the fact that labor time is scarce as community members are not allowed to hire outsiders to assist in harvesting. If the resource stock declines due to an exogenous development, cooperators restrain themselves, while defectors still harvest as much as they can. This increases the wedge between profits of defectors and cooperators, raising the temptation to defect. As a result, more cooperators decide to defect, putting even more pressure on the resource stock, which then increases temptation even more. This leads to a spiral, and eventually, the system flips to the "bad" equilibrium. The societal consequences of such a flip can be substantial because the system exhibits hysteresis: upon system collapse, moving back to the "good equilibrium" can be difficult and costly – if it is feasible at all.

To our knowledge, this chapter is the first to explain that collapse can be caused by interpersonal interactions and economic constraints, rather than by the presence of non-linear functions describing either resource growth or social pressure through user interaction. Our focus on the social-economic subsystem is especially relevant because of the prominent role social capital and community governance play in managing common pool resources like fish, forests, or grazing lands (Baland and Platteau, 1996; Ostrom, 1990, 2008, 2009; Ostrom and Nagendra, 2006). Having said that, it is also true that our chapter is not the first in noting that coupled social-ecological systems can be inherently complex (Barkley Rosser, 2001; Liu et al., 2007; Rammel et al., 2007). Research on tipping points in the social-economic system includes Schelling's model of segregation, in which agents prefer having neighbors with the same cultural background, giving rise to positive feedbacks depending on whether there are relatively few or relatively many 'kindred spirits' in a region (Card et al., 2008; Schelling, 1969). Similar tipping points associated with social reinforcement have been identified in extreme situations such as riots, or simply because agents face substantial uncertainty about the possible consequences of their actions (Ball, 2004; Bikhchandani et al., 1992; Gladwell, 2000; Granovetter, 1978; Noelle-Neumann, 1974; Scheffer et al., 2003). Regarding work about social interaction on renewable resource use, Iwasa et al. (2007) have analyzed a system in which agents are more inclined to undertake pollution-mitigating activities when the environment is in a poor state, and also when social pressure is high. In their model, alternative stable states occur when social pressure increases strongly with the fraction of cooperators in the community. This framework has been extended to incorporate non-linear resource dynamics as well, leading to even richer dynamics (Suzuki and Iwasa, 2009b). Finally, Taylor (2009) developed a model in which a resource has a certain minimum viable size and resource extraction has a negative effect on the profitability of a competing sector, making resource exploitation even more attractive. Taylor finds that this positive feedback leads to alternative stable states: either the resource is fully depleted, or in relatively healthy shape. This chapter is complementary to this research because we do not a priori assume any functional forms giving rise, by themselves, to tipping points, while our results stem from the fact individuals' time endowments are not infinite.

The setup of the paper is as follows. In section 3.2 we present the model, focusing on the mechanisms driving changes in the size of the resource stock and on those affecting the number of cooperating individuals in the community. To provide a benchmark against which to evaluate the outcome of the coevolution of the ecological and social-economic systems, we derive the standard non-cooperative equilibrium as well as the socially optimal allocation of effort. In sections 3.3 and 3.4 we analyze how the social-economic and ecological systems coevolve in response to changes in key drivers of change including population growth and technological progress in harvesting. The analysis is complicated but we are able to analytically identify the system's tipping points in case (i) there are no sources of income other than resource harvesting and (ii) the agents' effective time endowments system increase over time (for example because of technological progress in domestic activities). Having identified the mechanisms giving rise to tipping points in this simplest (and not most realistic) case in section 3.3, we resort to a numerical analysis in section 3.4 in which we relax the assumption of no alternative sources of income and where we analyze the system's resilience in the face of more important drivers of change such as population growth, technological progress in harvesting, or changes in the strength of moral persuasion. Section 3.5 concludes.

3.2 The model

We take the Gordon-Schaefer renewable resource model as starting point (Clark, 1990), and assume that there are $N > 1$ agents in a community who have access to a commonly-owned natural resource. The right to extract is exclusively associated with community membership; community members are not allowed to employ outsiders to assist in harvesting. The size of the resource stock at time t is denoted by $X(t)$. Each agent is endowed with a fixed effort rate $\hat{e}$ which she can allocate to harvesting the common pool resource, or to an alternative economic activity. The rate of effort agent i $(i = 1...N)$ allocates to resource harvesting at time t is denoted by $e_i(t)$, and hence $\hat{e} - e_i(t)$ is the effort rate she allocates to the alternative activity. We assume that the return to effort in the alternative economic activity is constant and equal to w, so that the income agent i derives from this activity is equal to $w(\hat{e} - e_i(t))$.

The relationship between harvesting effort and the quantity of resource goods harvested is given by the Schaefer production function, $h_i(t) = qX(t)e_i(t),$ where $h_i(t)$ denotes the harvest rate of individual i at time t and q is a technology parameter reflecting what fraction of the resource stock can be harvested per unit of effort allocated to harvesting (the so-called catchability coefficient). That means that the total harvest rate by the N agents at time t equals

$\sum_{i=1}^{N} h_i(t) = qX(t)\sum_{i=1}^{N} e_i(t)$.

Regarding the resource dynamics, this harvesting activity reduces the remaining stock, but there is also natural regeneration. The change in the size of the resource stock at time t, dX/dt, is equal to the net natural growth resulting from reproduction and mortality, $G(X(t))$, minus the sum of the individual harvesting rates of all the N agents having access to the resource. We assume that resource regenerates according to the standard logistic growth function, $G(X(t)) = rX(t)(1 - X(t)/K)$ where $r > 0$ is the intrinsic growth rate and $K > 0$ is the carrying capacity – the maximum stock size the resource would eventually reach if no harvesting took place. Without loss of generality we rescale the resource stock with the carrying capacity by setting $K = 1$. The variable X can now be thought of as the size of the natural resource stock as a fraction of its maximum value, and hence the size of the resource stock changes over time as follows:

$$dX/dt = rX(t)\left(1 - X(t)\right) - qX(t)\sum_{i=1}^{N} e_i(t). \tag{3.1}$$

Regarding the returns to harvesting effort, we assume harvests can be sold at the time-invariant unit price P (so that sales revenues are equal to $Ph_i(t) = PqX(t)e_i(t)),$ but we also assume that resource harvesting gives rise to an instantaneous negative externality: the returns any agent receives on her effort negatively depends on the total effort put in by the $N-1$ agents. More specifically, we assume that the net income generated by resource harvesting at time t is equal to $(PqX(t) - vE(t))e_i(t),$ where $E(t) = \sum_{j=1}^{N} e_j(t)$ denotes the community's aggregate effort in resource harvesting, and v reflects the extent to which the net marginal benefits of resource harvesting of an individual agent are reduced if the community's aggregate effort increases by one unit. Here, v can be thought of as the costs of congestion (for example because agents interfere with each other or have to compete for the bests spots), but also as any other negative interaction between contemporaneous harvesting activities (Cárdenas et al., 2000; Clark, 1990, p.223; Iwasa et al., 2007; Satake et al., 2007; Scheffer et al., 2000; Suzuki and Iwasa, 2009a; Wilson, 1982). Adding up the net revenues of harvesting and those of the alternative economic activity, total income earned by agent i at time t is:

$$\pi_i = PqX(t)e_i(t) + w(\hat{e} - e_i(t)) - vE(t)e_i(t), \text{where } 0 \le e_i(t) \le \hat{e}. \text{ [4]} \tag{3.2}$$

Note that this setup implies that there are two negative externalities, an instantaneous one and an intertemporal one. The intertemporal externality arises because current resource extraction reduces the amount of resources available in the future – thus reducing the marginal productivity of harvesting effort (cf. (3.1) and (3.2)). The instantaneous externality, vE, arises because of crowding effects (cf. (3.2)). These intertemporal and instantaneous externalities give rise to so-called Class I and Class II problems, respectively (Munro and Scott, 1985), and we assume that community members are concerned about the instantaneous crowding externality (the Class II problem), but not about the intertemporal externality (the Class I problem) – because they are not fully informed about the dynamics of resource regeneration, or simply because they are myopic. This setup is consistent with the real-world observation that many communities are able to reach a consensus on how to overcome the direct externality of the Class II problem (e.g. taking turns in getting the best fishing spots rather than competing for them), but not on the amount of resource to be extracted, which corresponds to the Class I problem; see for instance Taylor (1987).

[4] From here onwards we omit time indicators, unless omitting them may cause confusion.

3.2.1 Privately and socially optimal resource use

Before analyzing the interaction of agents when some (but not necessarily all) adhere to a social norm, we first determine the socially optimal and privately optimal use of the renewable resource to obtain benchmarks for the more complex interactions between cooperators and defectors as presented in sections 3.3 and 3.4.

Harvesting decision of cooperators

Because agents do not take the intertemporal Class I problem into account, the relevant benchmark for cooperation is the aggregate effort that maximizes the community's instantaneous aggregate income to overcome the Class II problem. We refer to this benchmark as the social optimum.[5] Using superscript SO to denote socially optimal values and taking into account that $0 \leq e_i(t) \leq \hat{e}.$ for all t, the aggregate effort that maximizes instantaneous social welfare $E^{SO}(X)$ is defined as follows:

$$E^{SO}(X) = \max_{E} \left\{ PXqE + w(N\hat{e} - E) - vE^2 \middle| \ 0 \leq E \leq N\hat{e} \right\}. \tag{3.3}$$

Solving (3.3), the symmetric individual socially optimal extraction effort rate $(e^{SO} = E^{SO} / N)$ is equal to:

$$e^{\text{SO}} = \begin{cases} \hat{e} & \text{if } X \geq (w + 2vN\hat{e}) / (Pq), \\ \dfrac{PqX - w}{2vN} & \text{if } w / (Pq) \leq X < (w + 2vN\hat{e}) / (Pq), \\ 0 & \text{if } 0 \leq X < w / (Pq). \end{cases} \tag{3.4}$$

From (3.4), we see that it is socially optimal to allocate all available effort $\hat{e}$ to resource harvesting if $X \geq (w + 2vN\hat{e}) / (Pq)$ and hence there is no social dilemma as long as putting in $\hat{e}$ cannot draw down the resource stock below this level – even if all N agents commit maximum effort. Calculating dX / dt (see equation 3.1) when $E = N\hat{e}$ and $X = (w + 2vN\hat{e}) / (Pq)$, we find that

$$\hat{e} > \frac{r\left(Pq - w\right) / N}{\left(2vr + Pq^2\right)}$$

is a necessary and sufficient condition to have a social dilemma. Let us now determine the socially optimal steady state resource stock. The resource is in equilibrium when $dX / dt = 0$ and the ag-

[5] The socially optimal level of the resource stock as defined here is below the "true" socially optimal level as it only solves the instantaneous Class II problem whereas the true social optimum would require agents taking into account the intertemporal (= Class I) problem as well.

gregate harvests equal the natural growth. Using (3.1) and (3.4), the socially optimal steady state resource stock is thus equal to:

$$X^{SO}=\begin{cases}(r\text{ - }\hat{e}Nq)/r & \text{if} \quad \hat{e}\le\dfrac{r(Pq-w)/N}{2rv+Pq^2},\\ \dfrac{2vr+wq}{2vr+Pq^2} & \text{if} \quad \hat{e}>\dfrac{r(Pq-w)/N}{2rv+Pq^2}.\end{cases} \tag{3.5}$$

Harvesting decision of defectors

Next, let us analyze what happens if all agents try to maximize their own private welfare without taking into account the negative consequences of their extraction effort on the welfare of all other agents in the community. We refer to this outcome as the non-cooperative equilibrium (using superscript NC). The effort rate that maximizes instantaneous private welfare is given by

$$e(X,E_{-i})=\max_{e_i}\left\{PXqe_i+w\left(\hat{e}-e_i\right)-v\left(E_{-i}+e_i\right)e_i\,\middle|\,0<e_i\le\hat{e}\right\}, \tag{3.6}$$

where $E_{-i}\equiv\sum_{j\neq i}e_j$. The best response function for selfish agents to the aggregate effort of the $N-1$ other agents in the community is given by

$$e^{\text{BR}}(X,E_{-i})=\min\left\{\frac{PXq-w}{2v}-\frac{1}{2}E_{-i},\hat{e}\right\}, \tag{3.7}$$

where superscript BR stands for best response. In the symmetric non-cooperative equilibrium we have $E_{-i}=(N-1)e^{BR}$, and hence the symmetric non-cooperative equilibrium effort rate equals

$$e^{NC}(X)=\begin{cases}\hat{e} & \text{if } X\ge\left(w+v\hat{e}(N+1)\right)/(Pq),\\ \dfrac{PqX-w}{v(N+1)} & \text{if } w/\left(Pq\right)\le X<\left(w+v\hat{e}(N+1)\right)/(Pq),\\ 0 & \text{if } 0\le X<w/\left(Pq\right).\end{cases} \tag{3.8}$$

Next, we determine the non-cooperative equilibrium steady state resource stock. Substituting $\sum_{i=1}^{N}e_i(t)=N\hat{e}$ and $\sum_{i=1}^{N}e_i=N\left(PqX-w\right)/\left(v(N+1)\right)$ – cf. (3.8) – into equation (3.1) and setting $dX/dt=0$, we find that the non-cooperative steady state resource stock is equal to

$$X^{NC}=\begin{cases}(r\text{ - }\hat{e}Nq)/r & \text{if} \quad \hat{e}\le\dfrac{r(Pq-w)/N}{2rv+Pq^2},\\ \dfrac{(N+1)vr+Nwq}{(N+1)vr+NPq^2} & \text{if} \quad \hat{e}>\dfrac{r(Pq-w)/N}{2rv+Pq^2}.\end{cases} \tag{3.9}$$

Comparing (3.4) to (3.8) and (3.5) to (3.9), we find that $X^{NC}\le X^{SO}$ and $e^{NC}(X)\ge e^{SO}(X)$, and the social optimum and non-cooperative equilibrium coincide only if either

$\hat{e} \leq \dfrac{r(Pq-w)/N}{2rv+Pq^2}$ or $N=1$. Furthermore, if $\hat{e} > \dfrac{r(Pq-w)/N}{2rv+Pq^2}$, the larger the community (N), the more resource rents are dissipated absent any cooperation. If N is sufficiently large such that $N/(N+1) \approx 1$, all resource rents are dissipated as soon as the time constraint ceases to be binding in harvesting:

$$\pi_i^{\mathrm{NC}} = \left(PqX - w - vN\left(\frac{PqX-w}{v(N+1)}\right)\right) e_i^{\mathrm{NC}} + w\hat{e} \approx w\hat{e} \tag{3.10}$$

if $w/(Pq) < X < (w + v\hat{e}(N+1))/(Pq)$.

3.2.2 Modeling cooperation, defection and the dynamics of social interaction

We assume that agents choose between two modes of behavior, acting cooperatively or non-cooperatively (also referred to as defection). We assume that cooperating agents take the instantaneous Class II problem into account, while agents acting non-cooperatively just try to maximize their own instantaneous income, taking the effort of all other agents as given. In this subsection, we first derive the effort rates chosen by the cooperators and defectors, and then discuss the mechanisms inducing cooperators to defect, and those inducing defectors to start cooperating. Following Bischi et al. (2004), we assume that cooperators always put in their fair share of the aggregate effort that would maximize instantaneous social welfare given the current size of the resource stock, $E^{SO}(X)$. In other words, cooperators choose $e^C = E^{SO}(X)/N = e^{SO}(X)$ for all X, where $E^{SO}(X)$ is given by (3.4).

Next, we derive what harvesting effort defectors choose to maximize their private welfare, taking into account both the size of the resource stock X and the number of cooperators and defectors in the community – as the numbers of cooperators and defectors influence the aggregate extraction effort invested by the defectors. We use $C(t)$ and $D(t)$ to respectively denote the number of cooperators and defectors in the community at time t, where $D(t) \equiv N - C(t)$. Then a defector maximizes his instantaneous profits, facing the aggregate effort of the $N-1$ other agents $E_{-i} = Ce^{\mathrm{SO}} + (D-1)e^{\mathrm{BR}}$. Substituting this expression into (3.7) and using (3.4), we find that the equilibrium effort rate defectors allocate to harvesting is equal to:

$$e^D(X,C)=\begin{cases}\hat{e} & \text{if} \quad X \geq \dfrac{w}{Pq}+\dfrac{2vN\hat{e}(N-C+1)}{Pq(2N-C)}, \\ \dfrac{(PXq-w)(2N-C)}{2vN(N-C+1)} & \text{if} \quad \dfrac{w}{Pq} \leq X < \dfrac{w}{Pq}+\dfrac{2vN\hat{e}(N-C+1)}{Pq(2N-C)}, \\ 0 & \text{if} \quad 0 \leq X < \dfrac{w}{Pq}.\end{cases} \tag{3.11}$$

Note that the harvesting effort of defectors does not only depend on the size of the resource X, but also on the number of agents acting cooperatively C at every instant in time. Furthermore, consistent with intuition, $e^D(X,C)=e^{NC}(X)$ if $C=0$; cf. (3.8) and (3.11). If all other agents act non-cooperatively, the best response of a defector is to choose the non-cooperative equilibrium effort rate too.

Next, we model social interaction by taking into account two countervailing forces. One is that agents are tempted to act non-cooperatively because of the higher profits associated with acting selfishly. The other is that agents see the need of solving the instantaneous Class II problem, and hence cooperators have an incentive to try to convince defectors of the social desirability of reducing their harvesting effort to the cooperative rate. We explain the two countervailing forces one by one.

Regarding the temptation to start acting selfishly, we assume that agents are more likely to defect the larger is the income associated with acting non-cooperatively as compared to that of acting cooperatively. This assumption is consistent with the observation that individuals tend to consider relative payoff differences rather than absolute ones (Azar, 2007). More specifically we assume that the fraction of cooperators that decide to defect at time t because of the temptation of higher income is equal to

$$\frac{dC/dt}{C}=-\beta\left(1-\frac{\pi^C(X(t),C(t))}{\pi^D(X(t),C(t))}\right),$$

where β is a parameter capturing the extent to which cooperators are tempted to become defectors for a given payoff ratio $\pi^C(t)/\pi^D(t)$. Next, we assume that whenever a cooperator meets a defector, there is a probability μ that the former succeeds in convincing the latter that he should act cooperatively. While we do not specifically rule out that cooperators turn into defectors after an encounter, we do assume that the net effect is favoring cooperation. This is a plausible assumption because defectors have no incentive to convince cooperative individuals to defect. The occurrence of discrete encounters can be modeled as a Poisson process, with λ being the Poisson parameter. The probability of an encounter taking place in the time interval $(t, t+\Delta t)$ is proportional with Δt and with the number of cooperators $C(t)$ and defectors $D(t)$. For Δt sufficiently small, the possibility of a community member having more than one encounter is negligible. Consequently, the probability of cooperator meeting a defector in the time interval $(t, t+\Delta t)$ is equal to $\lambda C(t) D(t) \Delta t / N$. Taking into account the fixed size of the community and defining $\alpha \equiv \lambda\mu$, the expected value of the change in $C(t)$ is equal to $C(t+\Delta t) - C(t) = \alpha C(t) D(t) \Delta t / N$, and hence $dC/dt = \alpha C(t)\left(N - C(t)\right)/N$. Combining the effects of moral persuasion and temptation, the number of cooperators develops over time according to the following differential equation:

$$dC/dt = \frac{\alpha}{N} C(t)\left(N - C(t)\right) - \beta C(t)\left(1 - \frac{\pi_C(t)}{\pi_D(t)}\right).^{6} \tag{3.12}$$

We can now analyze how the behavioral composition of resource users in the population affects the size of the resource stock, and vice versa. Suppose there are $C(t)$ cooperators at time t (and hence $N - C(t)$ defectors). In equilibrium, we have (i) $dX/dt = 0$, and (ii) $dC/dt = 0$; see equations (3.1) and (3.12), respectively. The number of cooperators does not change over time ($dC/dt = 0$) if the number of cooperators defecting equals the number of defectors being persuaded to act cooperatively:

$$\frac{\alpha}{N} C\left(N - C\right) = \beta C\left(1 - \frac{\pi^C}{\pi^D}\right) \tag{3.13}$$

and $dX/dt = 0$ requires that

$$rX\left(1 - X\right) = qX\left(Ce^C + \left(N - C\right)e^D\right). \tag{3.14}$$

We proceed as follows. In section 3.3 we analyze the case where there is no alternative economic activity $(w = 0)$ so that the resource good is the only source of income for the community. This assumption allows us to derive analytical results because $\pi^C / \pi^D = e^C / e^D$ (cf. (3.2)), thus considerably facilitating the analysis of (3.12) and (3.13). With $w > 0$ analytical results cannot be obtained, and hence we resort to a numerical analysis presented in section 3.4.

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[6] If the success rate of convincing defectors is too low (more specifically, if $\alpha \le \beta\left(N-1\right)/2N$), cooperators will completely disappear from the population – as shown in section 3.4.2. Inserting $C = 0$ in equation (3.12) it is easy to see that the disappearance of cooperators results in $dC/dt = 0$ independent of whatever policy intervention a regulator may want to undertake. This is neither plausible nor very interesting, and hence we assume that $\alpha > \beta\left(N-1\right)/2N$.

3.3 The coevolution of the resource stock and cooperation without an outside option

Suppose the community is self-sufficient and there is no alternative economic activity. The absence of an alternative economic activity can be captured by setting $w=0$, which can be inserted directly into equations (3.1)–(3.14).

3.3.1 The steady states of the social-ecological system

Combining (3.4) and (3.11) and setting $w=0$, we can identify three regimes (R1–R3).

If $X \geq \dfrac{2vN\hat{e}}{Pq}$, we have $e^C = e^D = \hat{e}$. (3.15.R1)

If $\dfrac{2vN\left(N-C+1\right)\hat{e}}{Pq\left(2N-C\right)} \leq X < \dfrac{2vN\hat{e}}{Pq}$, we have $e^C = \dfrac{PqX}{2vN}$ and $e^D = \hat{e}$. (3.15.R2)

If $0 \leq X < \dfrac{2vN\left(N-C+1\right)\hat{e}}{Pq\left(2N-C\right)}$, we have $e^C = \dfrac{PqX}{2vN}$ and $e^D = \dfrac{PqX\left(2N-C\right)}{2vN\left(N-C+1\right)}$. (3.15.R3)

Before analyzing the three regimes, let us first have a closer look at the two boundaries separating the three regimes. As is evident from (3.4), the boundary between regimes 1 and 2 is

$$X(C)\big|_{R1/R2} = 2vN\hat{e}/(Pq) \tag{3.16}$$

and this is a horizontal line in the (C,X) space. For all $X \geq X(C)\big|_{R1/R2}$ every agent in the community chooses the maximum effort rate $\hat{e}$, while the cooperative agents always choose interior harvesting effort rates when $X < X(C)\big|_{R1/R2}$. As implied by equation (3.11), the boundary separating regimes 2 and 3 is given by

$$X(C)\big|_{R2/R3} = \frac{2vN\hat{e}(N-C+1)}{Pq(2N-C)} \tag{3.17}$$

and this boundary is downward-sloping and concave in the (C,X) space. For any number of cooperators C, defectors continue harvesting at the maximum effort rate $\hat{e}$ for all $X \geq X(C)\big|_{R2/R3}$, whereas every community member chooses interior harvesting effort rates when $X < X(C)\big|_{R2/R3}$.

Let us now turn to analyzing the dynamics of the size of the resource stock and of the number of cooperators in the three regimes. All results are summarized in Table 3.1. Row A of this table contains the regime boundaries (3.16) and (3.17), while rows B and C represent $e^C(X)$ and $e^D(X,C)$ respectively, in each of the three regimes; cf. (3.15.R1)–(3.15.R3).

Table 3.1. Overview of the equilibrium effort rates of the cooperators and defectors in the three regimes, and the dynamics of the resource stock and of the number of cooperators and defectors.

Row	Variable	Regime 1	Regime 2	Regime 3
A		$X \geq \frac{2vN\hat{e}}{Pq}$	$\frac{2vN(N-C+1)\hat{e}}{Pq(2N-C)} \leq X < \frac{2vN\hat{e}}{Pq}$	$0 \leq X < \frac{2vN(N-C+1)\hat{e}}{Pq(2N-C)}$
B	e^C	$\hat{e}$	$PqX/(2vN)$	$PqX/(2vN)$
C	e^D	$\hat{e}$	$\hat{e}$	$PqX(2N-C)/(2vN(N-C+1))$
D	E	$N\hat{e}$	$CPqX/(2vN)+(N-C)\hat{e}$	$PqXZ(C)/v \approx pqX/v$
E*	$\frac{dX/dt}{X}$	$r(1-X)-qN\hat{e}$	$r(1-X)-q\left(\frac{CPqX}{2vN}+(N-C)\hat{e}\right)$	$r(1-X)-Pq^2XZ(C)/v \approx$ $r(1-X)-Pq^2X/v$
F*	$X(C)$	$\begin{cases} (r-qN\hat{e})/r & \text{if } \hat{e} \leq \frac{rPq/N}{2rv+Pq^2} \\ \text{NA} & \text{if } \hat{e} > \frac{rPq/N}{2rv+Pq^2} \end{cases}$	$\begin{cases} \text{NA} & \text{if } \hat{e} \leq \frac{rPq/N}{2rv+Pq^2} \\ \frac{2vN[r-q\hat{e}(N-C)]}{2vNr+Pq^2C} & \text{if } \hat{e} > \frac{rPq/N}{2rv+Pq^2} \end{cases}$	$\begin{cases} \text{NA} & \text{if } \hat{e} \leq \frac{rPq/N}{2rv+Pq^2} \\ \frac{vr}{Pq^2Z(C)+vr} \approx \frac{vr}{Pq^2+vr} & \text{if } \hat{e} > \frac{rPq/N}{2rv+Pq^2} \end{cases}$
G	$\frac{dC/dt}{C}$	$\frac{\alpha}{N}(N-C)$	$\frac{\alpha}{N}(N-C)-\beta\left(1-\frac{PqX}{2vN\hat{e}}\right)$	$\frac{\alpha}{N}(N-C)-\beta\left(\frac{N-1}{2N-C}\right)$
H	$C(X)$	N	$(2v(\alpha-\beta)N\hat{e}+\beta PqX)/2v\hat{e}\alpha$	$\frac{3}{2}N-\frac{1}{2}\sqrt{N^2+\frac{4\beta N}{\alpha}(N-1)} > 0$

* $Z(C)=\frac{N-C+C/2N}{N-C+1} \approx 1, \quad \frac{\partial Z(C)}{\partial C}=\frac{(1/N)-1}{2(N-C+1)^2} \approx 0.$

Defining $E(X,C) \equiv Ce^{C} + (N-C)e^{D}$, the community's aggregate effort in each of the three regimes are presented in row D. It is then straightforward to calculate both the relative change in the size of the resource stock $\left(\frac{1}{X}\frac{dX}{dt}\right)$ and the relative change of the number of cooperators $\left(\frac{1}{C}\frac{dC}{dt}\right)$ in each of the three regimes by inserting $E(X,C)$ into the associated differential equations (3.1) and (3.12) – see rows E and G. Then, we calculate the combinations of X and C that give the isoclines $dC/dt = 0$ and $dX/dt = 0$ (see equations (3.13) and (3.14)) – the so-called nullclines for the two state variables X and C; see rows F and H, respectively.

Let us first have a look at the nullcline of the resource stock in the three regimes; see row F in Table 3.1. We only have $dX/dt = 0$ for $C \geq 0$ in regime 1 if the community's aggregate time endowment is too small – given the other parameter values – for the agents to collectively draw down the stock to $X \leq 2vN\hat{e}/(Pq)$; see also equation (3.9). [7] To have the system reach regimes 2 and 3, we need $\hat{e} > \frac{rPq/N}{2rv + Pq^2}$. If that is the case, the nullcline of X in regime 2 is equal to $X(C) = \frac{2vN\left[r - q\hat{e}\left(N-C\right)\right]}{2vNr + Pq^2C}$, and is hence an upward-sloping and concave function of the number of cooperators, C; see row F for regime 2 in Table 3.1. Finally, the nullcline of X is upward-sloping and concave in regime 3, as given by $X(C) = \frac{vr}{Pq^2Z(C) + vr}$, where $Z(C) = \frac{N - C + C/2N}{N - C + 1}$. Because $Z(C) \approx 1$ if C is small relative to N, the nullcline is almost horizontal in regime 3. Finally, there is a trivial nullcline at $X = 0$ (not shown): once the renewable natural resource is fully extinct, it will never recover.

The nullcline of the resource stock can also be depicted graphically; see panel A of Figure 3.1. In this panel the regime boundaries (3.16) and (3.17) are shown. We do not show the case of $\hat{e} \leq \frac{rPq/N}{2rv + Pq^2}$ because from Table 3.1 we know that the system then has only one non-trivial steady state in total (which is located in regime 1, with all community members being coopera-

[7] If $\hat{e} \leq \frac{rPq/N}{2rv + Pq^2}$, the steady state resource stock in regime 1 is equal to $\left(r - \hat{e}Nq\right)/r$. If $\hat{e} > \frac{rPq/N}{2rv + Pq^2}$ instead, $dX/dt < 0$ for all $C \geq 0$ in regime 1 because all agents harvest at maximum effort.

tors). Instead, Figure 3.1 depicts the more interesting case of $\hat{e} > \frac{rPq/N}{2rv + Pq^2}$. In this case, as indicated by row F of Table 3.1, the nullcline for X is an almost horizontal line in regime 3, it is concave and upward-sloping in regime 2, and in regime 1 we always have $dX/dt < 0$ for all $0 \le C \le N$.

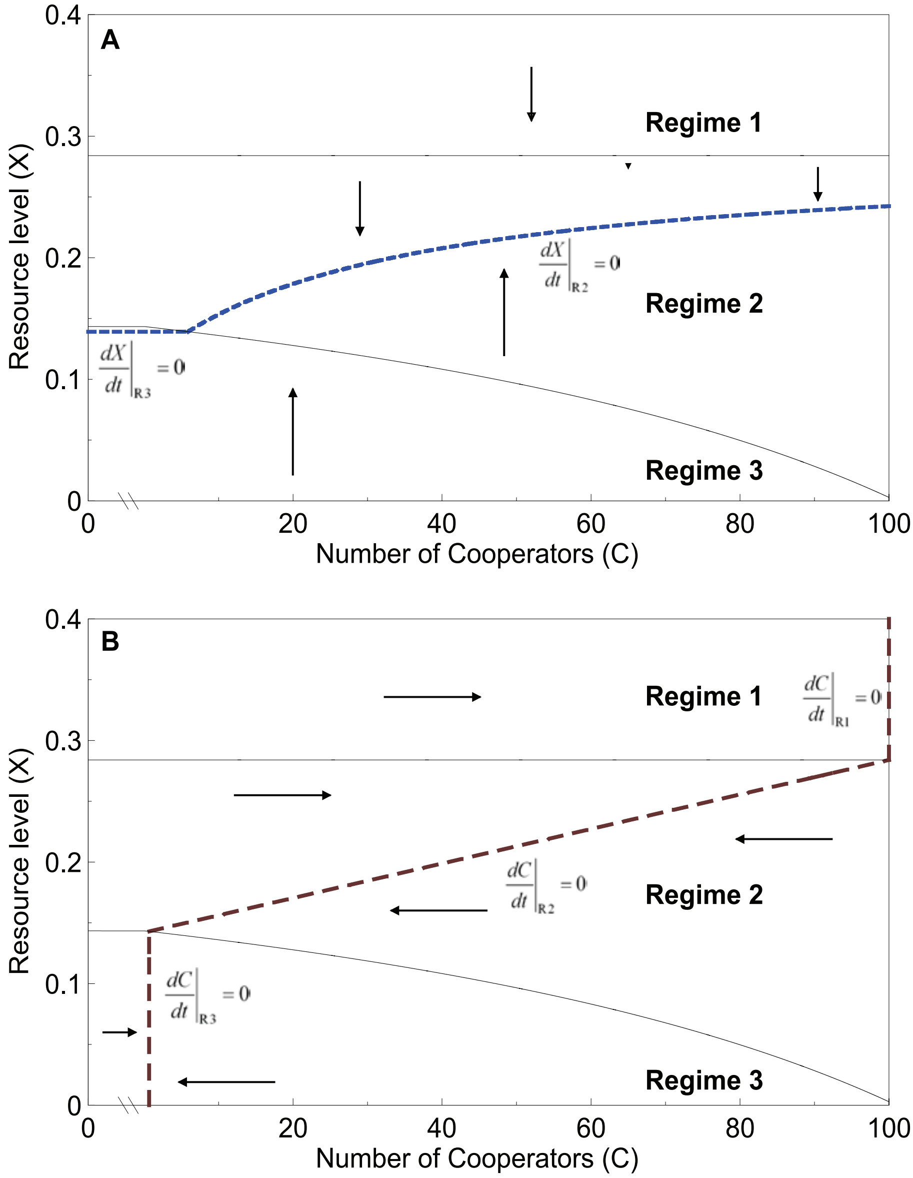

Figure 3.1. The nullclines of the resource stock (panel A) and of the number of cooperators (panel B).[8]

[8] Trivial nullclines $dC/dt = 0$ and $dX/dt = 0$ are not shown. Unless stated otherwise, all figures are drawn using the following parameter values: $\hat{e} = 0.71$, $N = 100$, $P = 50000$, $q = 0.01$, $r = 0.8$, $\alpha = 0.1$, $\beta = 0.2$.

Finally, the dynamics of the resource stock are indicated in this figure by means of arrows; for all stocks above (below) the nullcline of X, the resource stock is declining (increasing) over time. Next, let us analyze the location of the $dC/dt = 0$ isocline in the three regimes; see row H and also panel B in Figure 3.1. There is a trivial nullcline at $C = 0$ (not shown); if there are no cooperators, nobody is available to convince the defectors of the desirability of acting cooperatively. In addition, the nullcline of C is vertical at $C = N$ in regime 1: because $e^C(X) = e^D(X,C) = \hat{e}$, all agents are cooperators if $X \geq 2vN\hat{e}/Pq$. Next, in regime 2 the nullcline of C is a linear upward-sloping function of X: $C(X) = (2v(\alpha-\beta)N\hat{e} + \beta PqX)/(2v\hat{e}\alpha)$. Finally, the nullcline of C is vertical in the (C,X) space in regime 3: $C(X) = \frac{3}{2}N - \frac{1}{2}\sqrt{N^2 + \frac{4\beta N}{\alpha}(N-1)} > 0$.[9]

As was the case with the resource stock in panel A of Figure 3.1, we can graphically depict the dynamics of the number of cooperators in the community; see panel B of Figure 3.1. For all numbers of cooperators left (right) of the non-trivial nullcline of C (as defined in row H of Table 3.1), the number of cooperators in the community increases (decreases) over time. Let us now determine the steady states of the system in the three regimes, which requires calculating the intersection points of the $dX/dt = 0$ and $dC/dt = 0$ isoclines (see also Appendix 3.A). Both the number of equilibria as well as their location in the various regimes depend on the relative sizes of all parameters of both the social-economic and ecological subsystems. If $\hat{e} \leq \frac{rPq/N}{2rv + Pq^2} \equiv \hat{e}_0$, the system has one non-trivial steady state, $(C,X) = (N,(r - \hat{e}Nq)/r)$, and this steady state is located in regime 1; see row F in Table 3.1. The case of $\hat{e} > \hat{e}_0$ can be represented graphically by superimposing panels A and B of Figure 3.1; see Figure 3.2. From panels A–C in Figure 3.2 we see that there is no steady state in regime 1 if $\hat{e} > \hat{e}_0$, but there may be 0, 1 or 2 steady states in regime 2, and there is maximally one stable steady state in regime 3.[10]

We proceed with analyzing the various cases. Regarding the number of steady states in regime 2 if $\hat{e} > \hat{e}_0$, we have zero steady states in that regime if the $dC/dt = 0$ isocline is located strictly to the North-West of the $dX/dt = 0$ isocline for all values of C; see panel A in Figure

[9] Recall from footnote 6 that we assume $\alpha > \beta(N-1)/2N$. That means that indeed $C(X) > 0$ for all $X > 0$.

[10] All equilibria with either $X = 0$ or $C = 0$ are trivial, and all of them are unstable. Because of these reasons, we ignore them in the rest of the chapter

3.2. In equation (3.A6) of Appendix 3.A we show that this is the case if $\hat{e} > \hat{e}_1$, where $\hat{e}_1$ is defined by

$$\hat{e}_1 = \frac{4\beta r P^2 q^3}{N\left(4\beta P q^2\left(2rv + Pq^2\right) - \alpha\left(4rv\left(rv + Pq^2\right) + P^2 q^4\right)\right)}. \tag{3.18}$$

Here, $\hat{e}_1$ is defined as the effort endowment for which $dC/dt = 0$ and $dX/dt = 0$ are tangent in regime 2. If $\hat{e} > \hat{e}_1$, there are zero equilibria in regimes 1 and 2, and there is exactly one equilibrium in regime 3; see E_3 in panel A of Figure 3.2.[11] If, however, $\hat{e}_0 < \hat{e} \le \hat{e}_1$, there are either one or two equilibria in regime 2, and one or zero in regime 3; see panels B and C in Figure 3.2. First, the nullclines of X and C are tangent if $\hat{e} = \hat{e}_1$, giving rise to just one equilibrium in regime 2, and also one in regime 3 (because of the same argument as presented above for $\hat{e} > \hat{e}_1$). Second, the nullclines may intersect once or twice in regime 2 in case $\hat{e}_0 < \hat{e} < \hat{e}_1$. Note that the nullcline of C in regime 2 is an upward-sloping straight line that intersects the top regime boundary at $C = N$.[12] Also note that the nullcline of X is upward-sloping and concave in regime 2, and that it hits the $C = N$ axis at resource stock level that is strictly below the top regime.[13] That means that the two nullclines always intersect twice in regime 2 unless the intersection point of the C nullcline with the lower boundary is to the South-East of that of the X nullcline. Using the specifications of the nullclines of X and C from either regime 2 or regime 3, these nullclines intersect at the boundary between regimes 2 and 3 if $\hat{e}$ is equal to $\hat{e}_2$, where $\hat{e}_2$ is derived in equations (3.A7) and (3.A8) in Appendix 3.A:

$$\hat{e}_2 = \frac{\beta r P q}{\alpha N\left(rv + Pq^2\right) + \beta\left(2rNv + Pq^2(2N-1)\right) - \sqrt{\alpha}\sqrt{N}\sqrt{\alpha N + 4\beta(N-1)}\left(rv + Pq^2\right)}. \tag{3.19}$$

If $\hat{e} < (>)\ \hat{e}_2$ the nullcline of C intersects the lower regime boundary at a point South-East (North-West) of the point where the nullcline of X hits the lower boundary. Hence, a necessary

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[11] This can easily be inferred from panel A of Figure 3.2. The nullcline of C is vertical in regime 3 while the associated nullcline of X is upward sloping and concave (albeit near–horizontal); see rows F and H in Table 3.1. That means that the two nullclines always intersect in regime 3 as long as the nullcline of C intersects the boundary between regimes 2 and 3 to the North–West of the point where the nullcline of X intersects that boundary. This condition is always met if the nullcline of C is located strictly to the North–West of the nullcline of X in regime 2 – because the nullcline of C is an upward–sloping and straight line in regime 2 while the associated nullcline of X is upward–sloping and concave.

[12] This can be verified by inserting $X = 2vN\hat{e}/(Pq)$ into $C(X) = (2v(\alpha - \beta)N\hat{e} + \beta PqX)/(2v\hat{e}\alpha)$ in row H, Table 3.1.

[13] This can be verified by inserting $C = N$ into $X(C) = \dfrac{2vN\left[r - q\hat{e}(N - C)\right]}{2vNr + Pq^2 C}$ in row F of Table 3.1.

condition for having two equilibria in regime 2 and one in regime 3 is that $\hat{e}_1 > \hat{e} > \hat{e}_2$, while a necessary condition for having just one equilibrium in total is that $\hat{e} < \hat{e}_2$ (where the equilibrium is located in regime 1 if $\hat{e} < \hat{e}_0$, and where it is located in regime 2 if $\hat{e}_0 < \hat{e} < \hat{e}_2$).

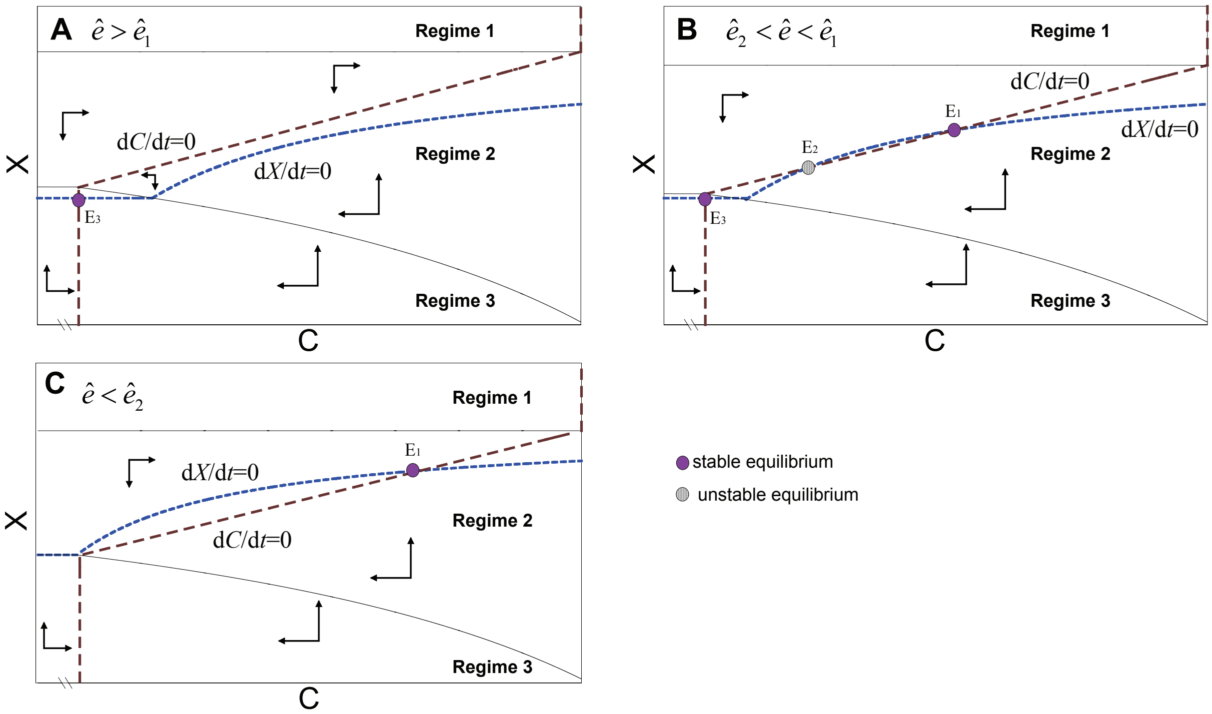

Figure 3.2. Phase planes for different values of $\hat{e}$ giving rise to either no non-trivial equilibrium in regime 2 and one in regime 3 (panel A), two equilibria in regime 2 and one in regime 3 (panel B), or one equilibrium in regime 2 and none in regime 3 (panel C).[14]

So we can now fully describe all possible equilibrium constellations for all possible levels of effort endowments. First, if $\hat{e} \le \hat{e}_0$, there is one stable equilibrium in regime 1 (not shown in Figure 3.2). Next, if $\hat{e}_0 < \hat{e} < \hat{e}_2$, the system still has just one stable equilibrium, E_1, and this equilibrium is located in regime 2; see panel C of Figure 3.2. If $\hat{e}_2 < \hat{e} < \hat{e}_1$, we have three equilibria in total; a stable and an unstable one in regime 2 (E_1 and E_2, respectively), and a stable one (E_3) in regime 3

[14] Values for all parameters as before, except for the focal parameter $\hat{e}$, which is 0.75 (panel A), 0.713, (panel B) 0.689 (panel C). Again, the trivial nullclines are not shown. Note that the intersection point of $dC/dt = 0$ and the horizontal axis $X = 0$ and the intersection point of $dX/dt = 0$ and the vertical axis $C = 0$ are equilibria too, and so is the origin of the system; cf. (3.13) and (3.14). As these three equilibria are unstable, we omit them in this figure.

– as depicted by panel B in Figure 3.2. Finally, if $\hat{e} > \hat{e}_1$, there is just one stable equilibrium in regime 3, as depicted in Panel A of Figure 3.2.

The stability properties of the situations depicted for $\hat{e} > \hat{e}_1$ (Panel A) and $\hat{e}_0 < \hat{e} < \hat{e}_2$ (Panel C) are straightforward, but they are slightly more complicated in case of $\hat{e}_2 < \hat{e} < \hat{e}_1$ (Panel B). The phase diagram associated with the latter case is presented more clearly in Figure 3.3. Panel A of that figure shows that equilibria E_1 and E_3 are locally stable nodes, while they are separated by an unstable saddle-point E_2. Panel B of that figure shows that if the system is in the "good" equilibrium E_1 ("bad" equilibrium E_3), a shock that reduces (increases) the number of cooperators can move the system to the "bad" equilibrium E_3 ("good" equilibrium E_1), but only if the shock is sufficiently large to make the system jump into the basin of attraction of E_3 (E_1) beyond the separatrix going through the unstable intermediate equilibrium E_2.

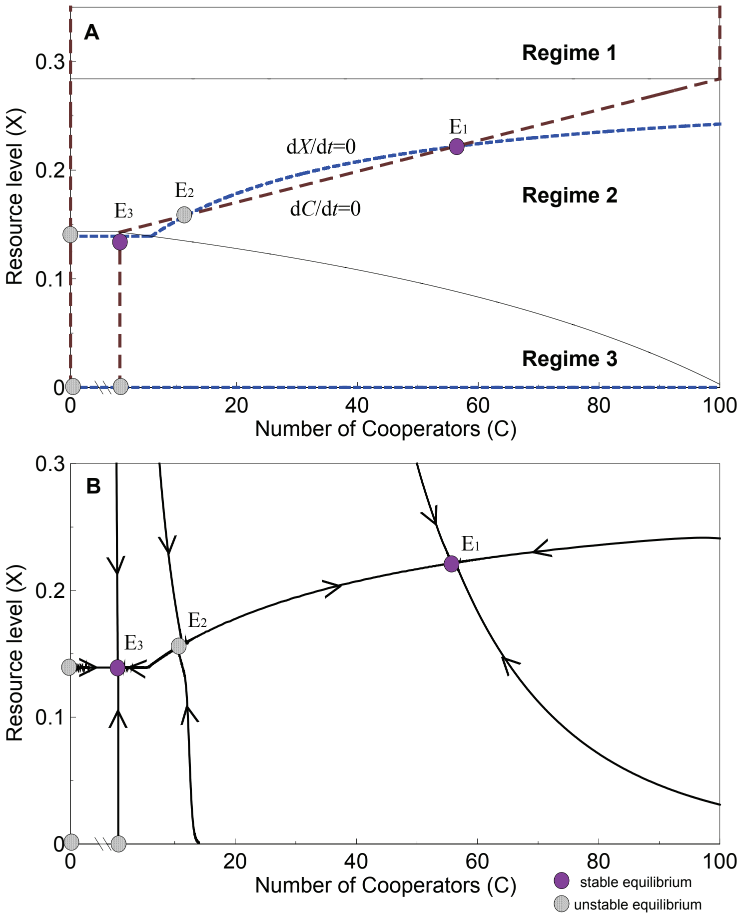

Figure 3.3. A phase diagram of the system with alternative stable states (panel A) and the associated trajectories towards these steady states (panel B).

So, our social-ecological system can be in a variety of steady states, and the number of steady states depends on the parameters of the system.

Figure 3.4 shows a conceptual figure of the feedback structure of the system and explains which mechanisms cause these multiple equilibria. Moral persuasion is stronger if there are more cooperators, but so is the temptation to defect – because the best-response function of a defector is a decreasing function of the aggregate effort put in by the other community members; cf. (3.7). The two effects stabilize each other, and the interaction of cooperators and defectors alone does not cause multiple equilibria. Rather, it is the fact that individual harvesting effort can never be larger than $\hat{e}$ that produces these alternative stable states. Because individual labor is in limited supply, a positive feedback arises between the size of the resource stock and the number of defectors. The smaller the resource stock, the larger the payoffs of defectors relative to those of cooperators (see the dotted line between the resource stock and the number of defectors in Figure 3.4). And the more defectors there are, the smaller is the resource stock (see the uninterrupted line between the resource stock and the number of defectors in Figure 3.4).

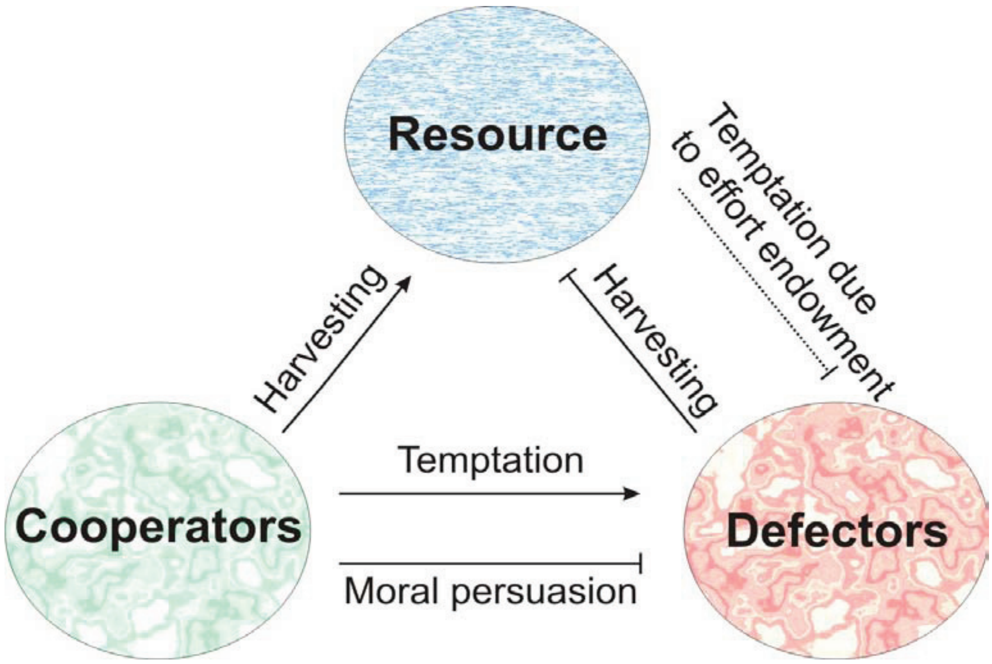

Figure 3.4. The social-economic dynamics are driven by moral persuasion and temptation. The presence of the effort endowment leads to a non-linearity, giving rise to alternative stable states. Positive feedbacks are indicated by arrows and negative feedbacks are depicted by bars.

This positive feedback produces alternative stable states and may lead to a catastrophic transition in the end. Without the effort constraint, the relative change in effort rates of cooperators and defectors would be the same, keeping the payoff ratio, and hence the temptation to defect, constant. Therefore, multiple equilibria would not occur.

3.3.2 The availability of effective labor time and system collapse

In the previous sub-section we have shown that the social-ecological system exhibits alternative stable states. In principle, the system can be moved from one stable equilibrium to another because of exogenous changes in the harvesting technology (q), the size of the community (N), the agents' time endowment $(\hat{e})$, or other parameters in either the ecological or the social-economic system. In this subsection we analyze how the system is affected when exogenous developments increase the community's aggregate time endowment available for resource harvesting. Here, $\hat{e}$ may change because of efficiency improvements in household chores. Admittedly, from the list of potential drivers of system collapse this is the least plausible one, but analytically it is the easiest one. Therefore, we focus on the impact of changes in $\hat{e}$ in this subsection, but we present the full analysis of the consequences of, among others, population growth and technological progress in section 3.4.2.

The typical pattern for the equilibrium values of C and X as a function of $\hat{e}$ (keeping all other parameters constant) are presented in panels A and B of Figure 3.5, respectively. Starting from the point where agents have time endowments less than $\hat{e}_0$, the aggregate effort available $N\hat{e}$ is insufficient for the community to reduce the resource stock below $X = 2vN\hat{e}/Pq$ (cf. (3.16)) even if all community members spend all their effort on resource harvesting, and hence all agents harvest at $\hat{e}$ (see also rows B and C in regime 1 of Table 3.1). Because defectors and cooperators all harvest at the maximum rate, temptation to defect is absent and hence $C = N$ for all values of $\hat{e}$ below $\hat{e}_0$; see panel A of Figure 3.5. The equilibrium resource stock is equal to $X = (r - Nq\hat{e})/r$, and hence the size of the resource stock is falling linearly as a function of $\hat{e}$; see panel B of Figure 3.5. This is the case for all $\hat{e} < \hat{e}_0$ and the steady states associated with each effort endowment are all located in regime 1.

So, for $\hat{e} < \hat{e}_0$, the system displays a gradual decline of the resource stock X if $\hat{e}$ increases. However, changes in $\hat{e}$ affect the system in multiple respects; all proofs are presented in Appendix 3.B. First, any increase in $\hat{e}$ results in the boundary between regimes 1 and 2 shifting up (the cooperators switch to interior harvesting effort rates at larger remaining stock sizes if $\hat{e}$ increases), while the boundary between regimes 2 and 3 rotates to the North-East (see (3.16) and (3.17), but also equations (3.B1) and (3.B2) in Appendix 3.B). Second, the locations of the $dX/dt=0$ and $dC/dt=0$ isoclines are also affected by changes in $\hat{e}$. From row F in Table 3.1 we see that the nullcline of X shifts down in regime 2 if the agents' time endowment increases, while the nullcline of C shifts to the left in that regime – as shown in row H in Table 3.1 (see also equations (3.B3) and (3.B4) in Appendix 3.B). When $\hat{e}$ becomes larger than $\hat{e}_0$ and continues to increase, the system moves through the phase diagrams and equilibria as presented in panels A–C in Figure 3.2 – but in reverse order. So, for $\hat{e}_0 \le \hat{e} < \hat{e}_2$, the two nullclines intersect zero times in regimes 1 and 3 and just once in regime 2, and the associated equilibrium, E_1, is globally stable; see panel C of Figure 3.2. If $\hat{e}$ continues to increase, the $dX/dt=0$ isocline shifts further down while the $dC/dt=0$ isocline shifts further to the left, and hence E_1 moves to the South-West. That means that C and X continue to gradually decline if $\hat{e}$ continues to increase in the range $\hat{e}_0 \le \hat{e} < \hat{e}_2$; see Figure 3.5. In regime 2 the cooperators reduce their effort rates below $\hat{e}$, and the more so the smaller is X; see row B in Table 3.1. The defectors, however, keep on harvesting at the maximum rate, and hence the difference in profits between cooperators and defectors increases with $\hat{e}$. As a result C falls, but the fall in X is less pronounced because the cooperators reduce their harvesting effort.

If $\hat{e}$ continues to increase so that it moves into the range $\hat{e}_2 \le \hat{e} < \hat{e}_1$, the nullclines have shifted such that they intersect twice in regime 2; see panel B of Figure 3.2. Now two new equilibria emerge, in addition to the initial (and locally stable) equilibrium E_1: the unstable equilibrium E_2, located in regime 2, and the locally stable equilibrium E_3, located in regime 3. The three equilibria are represented in Figure 3.5 – with the stable equilibria being connected by continuous lines and the unstable equilibria being connected by the dotted line. As long as $\hat{e} < \hat{e}_1$, the system remains in the good equilibrium E_1, and further exogenous increases in $\hat{e}$ just move the nullcline of C more to the left while the nullcline of X continues to shift down. That means that E_1 continues to gradually move to the South-West. This implies that the basin of attraction of the "good" equilibrium shrinks. Unnoticed the system loses resilience.

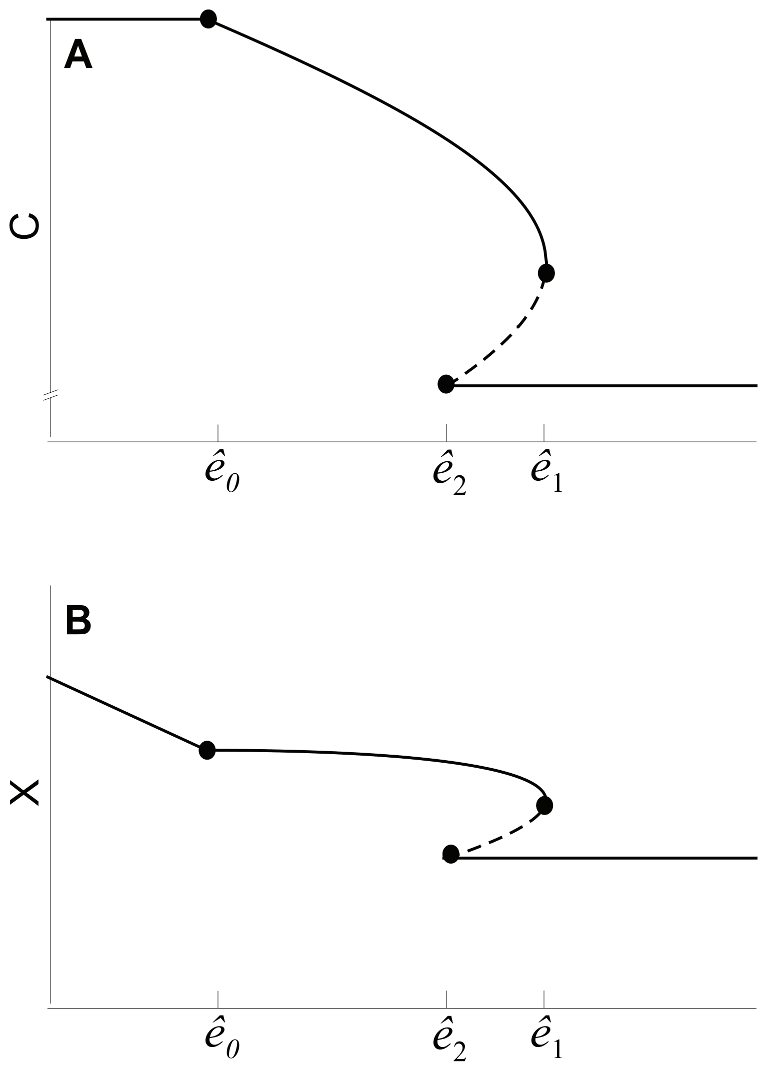

Figure 3.5. Bifurcation diagram showing internal equilibria of the number of cooperators C (panel A) and the resource stock X (panel B) for different values of the effort endowment $\hat{e}$. Stable equilibria are shown by solid lines, unstable equilibria are shown by dotted lines. Dots denote the two tipping points $\hat{e}_1$ and $\hat{e}_2$ and the point $\hat{e}_0$ where the social dilemma materializes.

While the decline in resource conservation and cooperation is thus gradual for increases in $\hat{e}$ as long as $\hat{e} < \hat{e}_1$, the system collapses at $\hat{e} = \hat{e}_1$. At this critical threshold (or tipping point), there is just one equilibrium in regime 2 (and also one in regime 3), and even an infinitesimally increase in $\hat{e}$ beyond $\hat{e}_1$ results in equilibrium E_1 vanishing, moving the system to the one remaining equilibrium, E_3, in regime 3; see panel A in Figure 3.2. The system collapses, and in Figure 3.5 the equilibrium values of C and X drop to the lower branches. During the transition from the good equilibrium E_1 to the bad equilibrium E_3, the profits of cooperators fall with an increase in $\hat{e}$

because (i) they reduce their effort rates and (ii) the return on whatever effort they decide to put in, falls too. Profits of the defectors fall too, but less so because they do not reduce their effort yet – and hence only suffer from the lower returns to effort. Hence, the temptation to defect continues to increase, the more X falls. Eventually, the system comes to a halt at the bad (but stable) equilibrium, E_3. Because of the fact that the $dX / dt = 0$ isocline is almost horizontal in regime 3, the equilibrium size of the resource is very close to the steady state resource stock absent any cooperation, as defined in equation (3.9). In regime 3, all agents harvest at interior effort rates, and hence subsequent increases in the effort endowment beyond $\hat{e}_1$ neither affects the number of cooperators, nor the size of the resource – as is also shown in Figure 3.5.

This analysis has important implications for policy makers as it makes clear that choosing the moment to intervene is of crucial importance to prevent the collapse of the social-ecological system. As shown in panel B of Figure 3.5, we find that increases in the time endowment result in intermediately fast decreases in the size of the resource stock if it is sufficiently plentiful (if $\hat{e}$ increases in the range $0 < \hat{e} < \hat{e}_0$). If $\hat{e}$ continues to increase in the range $\hat{e}_0 \leq \hat{e} < \hat{e}_1$ the rate of resource depletion actually *decreases* until $\hat{e}$ reaches $\hat{e}_1$, at which point the resource suddenly collapses. This pattern makes it hard for a potential manager of the resource (the regulator, or the government) to decide *when* to intervene. When observing resource depletion tapering off, the manager may falsely conclude that the system is stabilizing so that it becomes less urgent to intervene, while in fact the system is getting closer to collapse.

So what can the manager do once the system has collapsed? In this model the system collapse is reversible, but at substantial cost. Reducing effort back to $\hat{e}_1$ is not enough to restore the system to the good equilibrium E_1. The equilibria E_1 and E_2 in regime 2 re-emerge when $\hat{e}$ falls below $\hat{e}_1$ (see panel B of Figure 3.2) but the system does not jump from E_3 to E_1 because the community is "trapped" in the basin of attraction of the "bad" equilibrium E_3; see also Figure 3.3. The system only flips back to E_1 if $\hat{e}$ is decreased below $\hat{e}_2$, so that E_3 disappears again. Hence, $\hat{e}_2$ is the second tipping point of the system.

3.4 Model extensions

The analysis in section 3 focused on the impact of changing the time endowment, $\hat{e}$, on the social-ecological system, when resource harvesting is the community's only source of income. The assumption of $w = 0$ was introduced to obtain analytical results, not because it is more plausible than the case of $w > 0$. Also, exogenous changes in $\hat{e}$ are not likely to be the most important driver of change in the use of renewable resource systems. We remedy these two shortcomings by presenting a numerical analysis of the impact of changes in $\hat{e}$ when $w > 0$ in subsection 3.4.1, and by subsequently analyzing (in subsection 3.4.2) whether the same patterns of gradual change and sudden collapse can emerge if other parameters in the social-ecological system are subject to exogenous change, such as technological progress and population growth.

3.4.1 The role of outside labor market in the resilience of the social-ecological system

If labor markets are present and $w > 0$, analytical solutions can no longer be obtained. The main reason for this is that with $w > 0$ we no longer have $\pi^C / \pi^D = e^C / e^D$ (cf. (3.2)), which proved to be very convenient in working with equations (3.12) and (3.13). For the case of $w > 0$ we resort to a numerical analysis, showing that the presence of labor markets leads to results that are qualitatively very similar to those obtained in section 3.3. Figure 3.6 shows the internal equilibria of the two state variables C and X for different values of the effort endowment $\hat{e}$. As before, for sufficiently small time endowments there is no social dilemma, and hence the temptation for cooperators to defect is absent; $C = N$ in panel A of Figure 3.6. Larger time endowments do imply, however, that more can be harvested at every time t, and hence the resource stock decreases linearly; see panel B of Figure 3.6. If the time endowment of individual agents continues to increase, the social-ecological system moves into regime 2 where cooperators start using less effort for harvesting than is available, while the defectors continue to harvest at the maximum rate. If the time endowment increases even more, the social-ecological system collapses in exactly the same way as described in section 3.3. A positive feedback emerges because a reduction in the size of the resource stock results in cooperators reducing their effort rates while defectors do not, and the subsequent relative increase in defection payoffs reduces the number of cooperators, which, in turn, reduces the moral pressure to act cooperatively, and hence the resource stock falls even more. That means that all qualitative results obtained analytically assuming $w = 0$ carry over to the case of $w > 0$, and also the policy implications remain unchanged. If the resource stock seems to stabilize at an intermediately high level this is no guarantee that the system is resilient against shocks. And if the system has collapsed, restoring the system to the good equilibrium requires a reduction in the individual time endowments to a level that is (much) lower than the level at which the system was observed to collapse.

The only novel insight obtained from this analysis using $w > 0$ is that cooperation increases if the time endowment continues to increase after collapse; see panel A of Figure 3.6. This increase in cooperation materializes because $\lim_{\hat{e} \to \infty} \pi^C / \pi^D \uparrow 1$ if $w > 0$. If $\hat{e} > \hat{e}_1$, the social-ecological system is in equilibrium E_3 located in regime 3 (where all agents choose interior harvesting effort levels), and hence increases in w only increase the amount of money earned at the external labor markets, where the same wage rate applies to cooperators and defectors alike. Hence, the larger $\hat{e}$, the larger the income share of wages earned at the external labor market, and hence the closer the payoff ratio is to 1 (cf. equation 3.2).

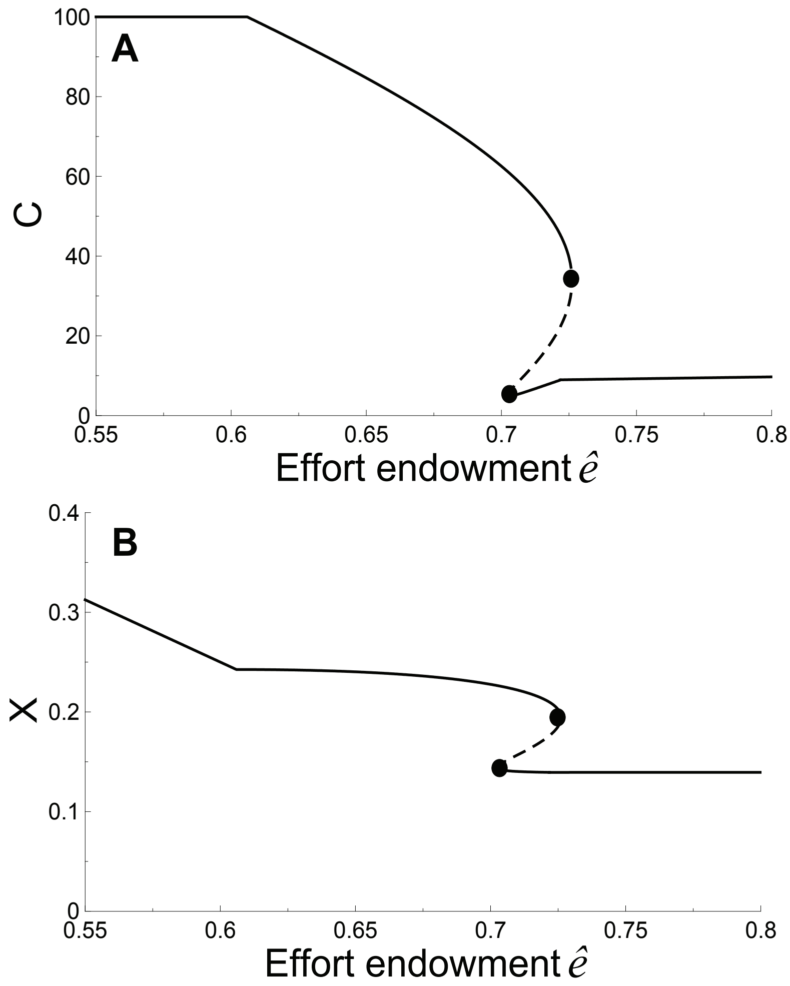

Figure 3.6. Bifurcation diagram showing internal equilibria of the number of cooperators C (panel A) and the resource stock X (panel B) for different values of the effort endowment $\hat{e}$ and with $w = 0.1$. Dots denote the two tipping points.

That means that the increase in cooperation following environmental collapse should not be interpreted as a sign that the system is moving back to a better equilibrium; it is purely the result of a decreasing difference in payoffs between acting cooperatively and non-cooperatively, resulting in less temptation to defect. Similarly, if the system has collapsed, the regulator should not be concerned about the fact that policies aimed at reducing the agents' time endowment actually results in a *decrease* in cooperation – reducing the time endowment of all agents reduces the wage share in total income and hence the payoff ratio between cooperators and defectors falls, so that it becomes more tempting to defect. As was the case in section 3.3, the system only flips back to the good equilibrium if the time endowment is reduced to a level well below the one that triggered collapse in the first place.

3.4.2 The role of other drivers in ecosystem collapse

Changes in the agents' time endowment is not the most plausible development driving the social-ecological system; population growth (an increase in N) and technological progress regarding the effectiveness of harvesting effort (q) are likely to be more relevant in practice. In addition, it is interesting to see how the system is affected by increases in the crowding externality (v) and in the strength of moral persuasion (α). Let us discuss each of these four in turn (while maintaining the assumption of $w>0$); see Figure 3.7. In panels A and B of Figure 3.7 we show the impact of increases in the harvesting technology parameter, q, on the steady state levels of C and X, respectively. The patterns are very similar to those presented in Figure 3.6, including the observation that cooperation increases if q continues to increase after collapse. The only difference is that in this case the steady state resource stock continues to fall with increases in q. Higher levels of q increase the marginal productivity of resource harvesting and hence agents allocate more labor to resource harvesting, and the resulting fall in X then restores the equilibrium between the net marginal productivity of resource harvesting and its opportunity cost, the outside wage rate w. In panels C and D we show the impact of increases in population size (N) on C and X. The consequences of increases in N (for given $\hat{e}$) are qualitatively identical to the ones observed in Figure 3.6 (resulting from an increase in $\hat{e}$ for a given N). Indeed, increases in N affect resource exploitation in essentially the same way as increases in $\hat{e}$: both result in the increase in the community's available harvesting time while leaving the marginal productivities unaffected.

Next, let us analyze how the system responds to increases in v, the external crowding parameter. Not surprisingly, the size of the resource stock increases with v (see panel F) – if the crowding externality is not very severe. The reason is that a higher v reduces the returns from resource harvesting, and hence agents spend more time at the external labor market, the larger is v. The consequences for cooperation, however, are surprising: the higher v, the higher the need for cooperation, but panel E shows that this does not generally translate into larger numbers of cooperators. The reason is that the cooperators tend to attach much larger weight to the crowding externality than do the defectors, and hence the profit ratio π^C / π^D is decreasing in v.

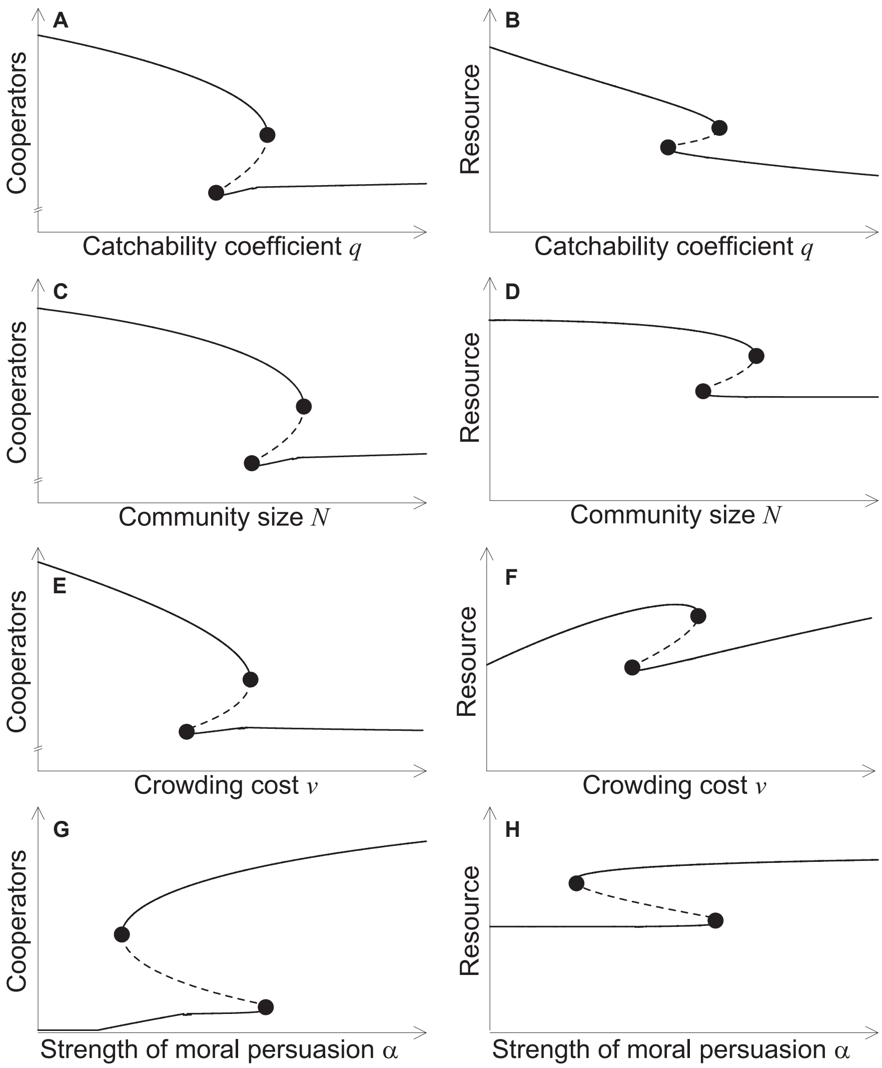

Figure 3.7. Bifurcation diagrams showing the internal equilibria of the number of cooperators and the resource stock for different focal parameters. Dots denote the two tipping points.

If the crowding externality becomes really large, the temptation to defect becomes very large too, and the social-ecological system collapses. In this case, however, continued increases in the crowding externality beyond the tipping point now result in higher levels of resource conservation, even though cooperation remains at a very low level – because the higher crowding costs continue to decrease profitability of resource harvesting, inducing defectors to allocate an increasingly large share of their labor time to working at the external labor market.

Finally, we analyze the impact of increases in the strength of the moral persuasion parameter, α, in panels G and H of Figure 3.7. These panels show that for low initial levels of α, increases in the strength of persuasion does not have much impact on either cooperation or the size of the resource stock – until α reaches a critical threshold. After crossing this threshold the system jumps to a much higher level of both cooperation and resource conservation, and the system is also quite robust against possible weakening of moral persuasion: if α has increased sufficiently for the system to flip to the good equilibrium, α can fall substantially before the system flips back to the bad equilibrium. Again, this is a direct result of the properties of systems exhibiting alternative stable states.

So, exogenous developments in population size, state of the technology, and the severity of the harvesting externality all give rise to the same dynamics as do changes in the time endowment itself – as analyzed analytically in section 3.3.2. Note that while the drivers analyzed in section 3.4.2 do not directly affect the agents' nominal time endowment, they do affect the effective time endowment. Better technologies or less severe harvesting externalities increase the marginal productivity of resource harvesting, and hence increase the relative scarcity of time. This shows that not the change, but the mere presence of constraints on economic activity is sufficient to produce multiple equilbria. Technological development, influx of agents from outside the community, small changes in the defectors' susceptibility to moral persuasion, all these changes can give rise to a sudden collapse. A gradual change in the key parameters may reduce the resilience of the system, thus paving the way to a catastrophic transition.

3.5 Conclusions

We developed a model of renewable resource use in which agents can decide to act cooperatively with respect to resource harvesting, or not. Social harvesting norms can spread through the community because of interpersonal relationships between cooperators and defectors (because the former try to convince the latter of the social desirability of acting cooperatively), but community members also always face the temptation to act non-cooperatively – because of the higher profits. The resulting social-ecological system is characterized by multiple equilibria, so that small changes in key parameters (including the size of the community, and the rate of technological progress) can trigger catastrophic transitions from relatively high levels of cooperation to widespread norm violation – causing the demise of the resource. Our setup is unique in that (i) it does not need to assume the ecological system to be highly non-linear to generate multiple equilibria, and (ii) tipping points emerge even though both the ecological and the social-economic systems, by themselves, are inherently stable.

In our social-ecological system, tipping points arise not because we introduce them explicitly in the differential equations, but because of the interaction between the ecological and social-economic subsystems. Here, positive feedback relationships occur because of the fact that, in closed communities, the amount of labor a community member can allocate to resource harvesting is necessarily finite because the common property regime usually does not allow members to hire external labor. If, for whatever reason, the resource becomes scarcer, the cooperators in the community decrease their harvesting effort while defectors continue to allocate all their available time to harvesting – if the net private marginal benefits of harvesting are strictly positive. A decrease in the size of the resource thus increases the relative profitability of defecting, and cooperation starts to decrease. Reduced cooperation further reduces the resource stock, thus reinforcing the temptation to defect, and also reduces the moral pressure on the non-cooperators. Thus, a positive feedback between the resource stock and the number of cooperators emerges endogenously in our system – possibly resulting in the collapse of the social-ecological system. And if the exogenous development is reversed (possibly because of government intervention), the system may flip back to the good equilibrium because of an opposite positive feedback loop. A larger resource stock induces cooperators to increase their harvesting effort, while defectors are still harvesting at the maximum rate. As a result, the relative profit difference falls, and so does the temptation to defect. Lower temptation increases the number of cooperators, which increases the resource stock, and lowers temptation even more. However, while a small change in the driver may cause the system to collapse, a non-marginal change in the driver is needed to restore the system to its good state. Hence, the system is characterized by hysteresis.

Our model thus shows that even if the system looks smooth on the outside, the regulator must be aware of potential collapses of cooperation in communities having access to renewable common pool resources. In that sense, this chapter is complementary to the literature analyzing inherently non-linear natural resource systems (such as shallow lakes and grazing lands) as including non-linear resource dynamics in our model would render the system even more complex.

While our model is purely theoretical in nature, it does give rise to one important policy implication. A regulator monitoring the ecological system should not be led to believe that any decrease in the rate of depletion of a natural resource is evidence of the system stabilizing. The decrease in depletion rates may be caused by some agents reducing their effort levels while others do not. If so, the subsequent relative increase in the returns to defecting may trigger a positive feedback between increased defecting and ever lower resource stocks, ultimately resulting in a sudden collapse of the resource stock.

Appendix 3.A: Derivation of the equilibria and tipping points for all possible values of $\hat{e}$

Deriving the non-trivial equilibria in each of the three regimes requires identifying the intersection points of the X and C nullclines as provided in rows F and H in Table 3.1. Consider first the case where $\hat{e} \leq \frac{rPq / N}{2rv + Pq^2} \equiv \hat{e}_0$. Noting that $C = N$ in regime 1, the system then has only one non-trivial equilibrium with the following coordinates:

$$(C_0, X_0) = (N, (r - \hat{e}Nq) / r) \tag{3.A1}$$

Next, consider the case where $\hat{e} > \frac{rPq / N}{2rv + Pq^2} \equiv \hat{e}_0$. Then there are zero steady states in regime 1.

The steady states in regime 2 can be identified by equating $X = \frac{2vN\left[r - q\hat{e}(N - C)\right]}{2vNr + Pq^2C}$ and $X{=}2v\hat{e}\left(\alpha C - \left(\alpha - \beta\right)N\right) / \beta Pq$. This system potentially provides two interior solutions, (C_1, X_1) and (C_2, X_2), where

$$X_1 = \frac{\sqrt{\hat{e}}\sqrt{N}\cdot v\left(\sqrt{\alpha}\sqrt{\alpha\hat{e}N\left(4r^2v^2 + 4rPvq^2 + P^2q^4\right) + 4\beta Pq^2\left(rPq - \hat{e}N\left(2rv + Pq^2\right)\right)} - \sqrt{\hat{e}}\sqrt{N}\left(\alpha\left(2rv + Pq^2\right) - 2\beta Pq^2\right)\right)}{\beta P^2 q^3}, \tag{3.A2}$$

$$C_1 = \frac{\sqrt{N}\left(\sqrt{\alpha\hat{e}N\left(4r^2v^2 + 4rPvq^2 + P^2q^4\right) + 4\beta Pq^2\left(rPq - \hat{e}N\left(2rv + Pq^2\right)\right)} + \sqrt{\alpha}\sqrt{\hat{e}}\sqrt{N}\left(Pq^2 - 2rv\right)\right)}{2\sqrt{\alpha}\sqrt{\hat{e}}Pq^2}, \tag{3.A3}$$

$$X_2 = -\frac{\sqrt{\hat{e}}\sqrt{N}\cdot v\left(\sqrt{\alpha}\sqrt{\alpha\hat{e}N\left(4r^2v^2 + 4rPvq^2 + P^2q^4\right) + 4\beta Pq^2\left(rPq - \hat{e}N\left(2rv + Pq^2\right)\right)} + \sqrt{\hat{e}}\sqrt{N}\left(\alpha\left(2rv + Pq^2\right) - 2\beta Pq^2\right)\right)}{\beta P^2 q^3}, \tag{3.A4}$$

$$C_2 = -\frac{\sqrt{N}\left(\sqrt{\alpha\hat{e}N\left(4r^2v^2+4rPvq^2+P^2q^4\right)+4\beta Pq^2\left(rPq-\hat{e}N\left(2rv+Pq^2\right)\right)}+\sqrt{\alpha}\sqrt{\hat{e}}\sqrt{N}\left(2rv-Pq^2\right)\right)}{2\sqrt{\alpha}\sqrt{\hat{e}}Pq^2}. \quad (3.A5)$$

Here, (C_1, X_1) and (C_2, X_2) are the coordinates of the equilibrium points E_1 and E_2, respectively; see panel B of Figure 3.2 and also Figure 3.3. Whether these equilibria exist depends on the parameter constellations. We find that $X_1 = X_2$ and $C_1 = C_2$ (so that E_1 and E_2 coincide) if

$$\hat{e} = \frac{4\beta rP^2q^3}{N\left(4\beta Pq^2\left(2rv+Pq^2\right)-\alpha\left(4rv\left(rv+Pq^2\right)+P^2q^4\right)\right)} \equiv \hat{e}_1. \quad (3.A6)$$

Solutions (C_1, X_1) and (C_2, X_2) only have real solutions if $\hat{e} < \hat{e}_1$. In fact, $\hat{e}_1$ is a so-called tipping point because the equilibria E_1 and E_2 vanish if $\hat{e}$ is infinitesimally larger than $\hat{e}_1$. So, if $\hat{e} > \hat{e}_1$ there are no equilibria in regimes 1 and 2, and the only equilibrium is E_3 located in regime 3. Equilibrium values X_3 and C_3 are equal to:

$$X_3 = \frac{2rv\sqrt{N}\left(\sqrt{N}\sqrt{\alpha N+4\beta(N-1)}+\sqrt{\alpha}\left(2-N\right)\right)}{\left(2rNv+Pq^2\left(2N-1\right)\right)\sqrt{\alpha N+4\beta(N-1)}-\sqrt{\alpha}\sqrt{N}\left(2rv(N-2)+Pq^2\left(2N-3\right)\right)}, \quad (3.A7)$$

$$C_3 = \frac{3}{2}N - \frac{1}{2}\sqrt{N^2+\frac{4\beta}{\alpha}(N^2-N)} > 0. \quad (3.A8)$$

The equilibrium in regime 3 only exists if (C_3, X_3) as defined by (3.A7) and (3.A8) is located below the regime boundary separating R2 from R3. Therefore an equilibrium in R3 only exists if $X_3 < \frac{2vN(N-C+1)\hat{e}}{Pq(2N-C)}$. Hence, substituting C_3 and X_3 into $X(C)\big|_{R2/R3} = \frac{2vN(N-C+1)\hat{e}}{Pq(2N-C)}$ and solving, we find that this is the case if $\hat{e} < \frac{Pq(2N-C_3)X_3}{2vN(N-C_3+1)} \equiv \hat{e}_2$, or, after having substituted in the state variables C_3 and X_3,

$$\hat{e}_2 = \frac{\beta rPq}{\alpha N\left(rv+Pq^2\right)+\beta\left(2rNv+Pq^2\left(2N-1\right)\right)-\sqrt{\alpha}\sqrt{N}\sqrt{\alpha N+4\beta(N-1)}\left(rv+Pq^2\right)}. \quad (3.A9)$$

Appendix 3.B: Analysis of the consequences of a change in the endowment of effort

Taking the first derivative of (3.16) and (3.17), we have

$$\frac{\partial X(C)\big|_{R1/R2}}{\partial \hat{e}} = 2vN/(Pq) > 0, \tag{3.B1}$$

$$\frac{\partial X(C)\big|_{R2/R3}}{\partial \hat{e}} = \frac{2vN\left(N-C+1\right)}{Pq\left(2N-C\right)} > 0. \tag{3.B2}$$

So, if $\hat{e}$ increases, both regime boundaries shift upward in the (C,X) space. Because $\frac{\partial^2 X(C)\big|_{R2/R3}}{\partial \hat{e}\partial C} < 0$ an increase in $\hat{e}$ induces the R2/R3 boundary to rotate to the North-East.

Next, we analyze the consequences of increasing $\hat{e}$ for the two nullclines. Taking the first derivatives of the nullclines of X and C (as presented in rows F and H of Table 3.1) with respect to $\hat{e}$, we have:

$$\frac{\partial X(C)\big|_{dX/dt=0}}{\partial \hat{e}} = \begin{cases} \dfrac{-2vNq(N-C)}{2vNr + Pq^2 C} \le 0 & \text{if} \quad \dfrac{2vN\left(N-C+1\right)\hat{e}}{Pq\left(2N-C\right)} \le X < \dfrac{2vN\hat{e}}{Pq} \qquad \text{(R2)} \\ 0 & \text{if} \quad 0 < X < \dfrac{2vN\left(N-C+1\right)\hat{e}}{Pq\left(2N-C\right)} \qquad \text{(R3)} \end{cases} \tag{3.B3}$$

$$\frac{\partial C(X)\big|_{dC/dt=0}}{\partial \hat{e}} = \begin{cases} 0 & \text{if} \quad X \ge \dfrac{2vN\hat{e}}{Pq} \qquad \text{(R1)} \\ \dfrac{-\beta PqX}{2v\alpha\hat{e}^2} < 0 & \text{if} \quad \dfrac{2vN\left(N-C+1\right)\hat{e}}{Pq\left(2N-C\right)} \le X \le \dfrac{2vN\hat{e}}{Pq} \qquad \text{(R2)} \\ 0 & \text{if} \quad 0 \le X < \dfrac{2vN\left(N-C+1\right)\hat{e}}{Pq\left(2N-C\right)} \qquad \text{(R3)} \end{cases} \tag{3.B4}$$

Hence, the $dX/dt = 0$ isocline shifts down in (C,X) space in regime 2 while the vertical intercept in regime 3 remains unchanged. The $dC/dt = 0$ isocline shifts to the left in the (C,X) space in regime 2 while the horizontal intercept in regime 3 remains unchanged.

4

Egocentric sanctions evolve spontaneously to overcome social dilemmas

Abstract

Social sanctions are powerful mechanisms for enforcing social norms that stabilize cooperation, especially when reward and punishment are combined. In many cases these social sanctions effectively overcome a social dilemma, as joint access to a common pool resource. In principle, sanctioning institutions can stabilize cooperation, but often they are inefficiently costly or may be used for other purposes than increasing group payoff. The question whether social sanctions are able to overcome social dilemmas efficiently, is therefore still unsolved. In our model, institutions, such as a moral value system, change slower than economic decision and both evolve endogenously from a continuum of strategies. With agent-based simulations and an analytical model, we show that a combination of punishments and rewards will effectively overcome any social dilemma if own behavior is used as a moral demarcation line between good and bad behavior. This is even the case if social sanctions are weak or costly, and if individuals make mistakes, or some harvesting activity may stay undetected.

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4.1 Introduction

A social dilemma occurs if private interests do not coincide with collective interests. Typically, the actions of one agent give rise to an – unintended – externality that affects the payoff of other individuals. Such an externality can be positive (when contributing to a public good) or negative (when harvesting a common pool resource). If agents do not take this externality into account, the outcome of their actions will be socially sub-optimal. Social dilemmas, such as sharing a resource or contributing to a public good are ubiquitous in daily life.

However, often those dilemmas can be overcome without central intervention with the help of social sanctions, for instance when individuals jointly manage a common pool resource (Baland and Platteau, 1996; Bischi et al., 2004; Janssen and Ostrom, 2006; Noailly et al., 2003; Ostrom, 1990; Ostrom et al., 1992; Sethi and Somanathan, 1996). Therefore, social sanctions are powerful mechanisms for enforcing social norms that stabilize cooperation (Fehr and Gächter, 2002; Gächter et al., 2008; Gurerk et al., 2006; Sigmund, 2007), especially when reward and punishment are combined (Andreoni et al., 2003; Rockenbach and Milinski, 2006). What makes social sanctions so powerful is the fact that norms of cooperation can be enforced even when single individuals do neither understand the reason, nor the exact functioning or the history behind these norms. According to Hayek (1978, p.12),

> *this adaptation to the general circumstances […] is brought about by his observance of rules which he has not designed and often does not even know explicitly, although he is able to honour them in action.*

In spite of this, social sanctions fail to raise group payoff in some cases. While sanctioning institutions can stabilize cooperation, they are often inefficiently costly or may be used for other purposes than increasing group payoff (Dawes et al., 2007; Dreber et al., 2008; Efferson et al., 2008; Fehr et al., 2008; Fehr and Schmidt, 1999; Fowler et al., 2005; Herrmann et al., 2008; Hsu et al., 2008; Johnson et al., 2009). The question whether social sanctions are able to overcome social dilemmas efficiently, is therefore still unsolved (Dreber et al., 2008; Gächter et al., 2008; Herrmann et al., 2008; Ohtsuki et al., 2009).

Additionally, norm compliance alone neither answers the question which social norms evolve, nor explain how beneficial such an evolving norm is. Indeed, individuals that blindly enforce *any* social norm may sustain a cooperative equilibrium, but may also prevent cooperation from arising by enforcing an anti-social equilibrium – or for that matter, enforce *any* other equilibrium (Boyd and Richerson, 1992). While some norms are indeed maladaptive (Elster, 1989), it seems that most social norms do serve a clear purpose, at least at the time they have evolved. According to Coleman (1990, p. 241) norms do not emerge randomly, but evolve for a reason. This sounds plausible, but what is the underlying mechanism?

Two explanations are currently available why social norms may evolve towards a level that maximizes group welfare. First, if groups that hold different social norms compete with each other, this can explain why the community with the most group-beneficial social norm survives (Henrich, 2004; Traulsen and Nowak, 2006; Wilson, 1975). This mechanism is known as group selection or multi-level selection. Second, group-beneficial social norms do not come automatically. Instead, it requires some form of enlightenment to achieve a sensible system of social norms. This argument, going back to Immanuel Kant, says that the more individuals know about the consequences of their actions, the better they know which behavior is appropriate (Ayala, 2010; Hauser, 2006). Knowing how to solve a social dilemma obviously facilitates the possibilities to find a way to coordinate to that equilibrium, even though it still may not happen. In any case, this explanation cannot answer the question how group-beneficial social norms evolve when individuals are fully unaware of the consequences of their actions, such as in the situation sketched by Hayek (1978). In this chapter we show that neither group selection, nor enlightenment is needed to explain the evolution of group-beneficial norms.

In economics, the replicator equation (Taylor and Jonker, 1978) is the standard tool to describe evolutionary processes. This approach requires a discrete set of strategies: mutations and the emergence of new strategies are usually not considered. Also the use of continuous strategies in the form of mixed strategies is problematic (Dieckmann and Metz, 2006). Adaptive dynamics is a modeling tool that can analyze continuous strategies and does not require assuming a priori any discrete strategies (Dercole and Rinaldi, 2008; Doebeli et al.; Geritz et al., 1998; Hofbauer and Sigmund, 1990; Kisdi and Geritz, 2010; Metz et al., 1996; Nowak and Sigmund, 1990). Usually adaptive dynamics makes use of the assumption that evolutionary processes are slower than ecological processes. Here, we assume that institutions (a system of moral preferences) change slower than economic decisions. We are aware of two studies that have used adaptive dynamics in the field of economics before. The first is a study on technological change, in which markets are assumed to clear rapidly, while innovations by individual firms are rare (Dercole et al., 2008) and the second investigates the coevolution of trade and human speech (Horan et al., 2008).

The assumption that agents change their private investments much more frequently than their social preferences builds on the tradition of new institutional economics, which has established that institutional layers are nested (North, 1990; Williamson, 2000). In our model, investment decisions are nested within informal institutions – in this case an enforcement mechanism based on sanctioning preferences. To facilitate the analysis, we assume that the institutional time scale (the sanctioning preference) changes infinitely slower than the effort allocation time scale (the investment decision). Therefore, a change in the sanctioning preference is analyzed when every agent is behaving optimally in the current environment, i.e. playing the best response, given the set of sanctioning preferences in the community. This assumption is in line with the standard economic assumption that agents are rational and make optimal choices.

Many social dilemmas do not have a temporal dimension, and can be well described with a static model. Polluting a river, for instance, affects all other individuals instantaneously, even though the effects may be long lasting. In our model, we will assume in most cases that actions have immediate consequences, but we will also investigate whether a time lag between actions and arising consequences is an obstacle for the evolution of cooperative norms.

This chapter contributes to the literature in several ways. First, we present a coherent mathematical framework that formalizes the tradition of new institutional economics, in which institutional levels are nested within each other (North, 2005; North, 1991; Williamson, 2000). We build on the observation that institutions, such as a moral value system, change slower than economic decision. Economic actions are embedded in informal constraints or institutions, which change typically very slow (Granovetter, 1985). Second, most evolutionary models are based on the replicator equation (Taylor and Jonker, 1978) and impose discrete strategies, such as cooperation and defection, into the model. In our model, the chosen amount of exploitation and peer pressure evolve endogenously from a continuum of strategies (Doebeli et al., 2004; Le Galliard et al., 2005; Nakamaru and Dieckmann, 2009). Third, we show that evolving social sanctions can guide the community towards socially optimal behavior, even if single individuals are myopic and can neither observe the resource stock, nor know what would be optimal. Fourth, and most important, our model shows that group selection is not needed to explain the evolution of optimal social sanctions. Instead, our mechanism makes use of the assumption that social preferences change slower than economic decisions. With agent-based simulations and an analytical model, we show that a combination of punishments and rewards will effectively overcome any social dilemma if own behavior is used as a moral demarcation line between good and bad behavior. Unexpectedly we find that social sanctions will be much less efficient when not own, but group behavior is used as a moral yardstick. Furthermore, the evolving sanctioning system will compensate for obstacles such as large group size or high profits that can be made by not cooperating, but it does so usually only up to a certain level. Then, sanctioning suddenly collapses and the community switches drastically from socially optimal exploitation to massive overexploitation if key parameters, such as perception errors or lack of appreciation for social sanctions, pass a critical threshold. The setup of this chapter is as follows. In section 4.2 we develop an analytical mini-model that captures the essence of the enforcement mechanism presented in this chapter. In section 4.3, we corroborate these findings with agent-based simulations that include mistakes, noise, and a very realistic utility function (a detailed description can be found in the Appendix). These simulations show that when agents make few mistakes and noise is low, the emerging sanctioning preference will evolve towards a level that overcomes the social dilemma. This is even the case if individuals value profits from harvesting much more than the effects of social sanctions. The simulations further reveal conditions under which social sanctions break down, indicating limits of peer pressure. In section 4.4 we use an extensive analytical model to show that this mechanism is generic and works in groups of arbitrary size, even when sanctioning is very costly. These findings hold both in common pool resource games and also public goods games. Finally, section 4.5 concludes.

4.2 The analytical mini-model

In the simplest case we consider a situation where only two agents i and j face a symmetric social dilemma. Each agent invests individual effort e that negatively affects the payoff of the other agent. Individual i chooses effort level e_i, which delivers marginally decreasing private income $\ln(e_i) - Ce_i$, while it gives rise to an externality cost Ee_i for agent j. Furthermore, we assume that agents have the possibility to punish and reward each other – both will be referred to as social sanctions. Each individual holds a sanctioning preference τ which determines punishments or rewards given to the other agent, depending on both effort levels. Agents ignore the externality that they impose on each other, while they are sensitive towards social sanctions. We assume that punishments and rewards are used symmetrically and that own behavior is used as the moral benchmark. This dichotomy of the good and the bad (or the right and the wrong), together with the emphasis on own behavior is rooted in many moral systems. A good example is the stylized picture of two shoulder angels whispering advice into a person's ear. Hence, individual i may receive sanctions in the form of punishments or rewards from agent j given by $\tau_j(e_j - e_i)$.

In our model, the decision to allocate resources towards the investment in a common good or the extraction from a common pool resource is embedded in a social context (Granovetter, 1985). Mathematically the preference is then an argument in the effort function, as given by $e(\tau)$. In the analytical work we assume that effort levels are chosen optimally when a change in the sanctioning preference occurs. The idea that individuals make optimal choices, while preferences change only slowly is common in many economic models (Güth and Kliemt, 1998). This implies that economic decisions are optimal, given the enforcement mechanism in place, i.e. the sanctioning preferences of both agents. For example, if both individuals are indifferent towards sanctioning each other $(\tau = 0)$, no enforcement takes place. In such a case, the open access Nash equilibrium would occur.

Total utility of individual i is given by $U_i = \ln(e_i) - Ce_i - Ee_j + \tau_j(e_j - e_i),$ while agent j receives $U_j = \ln(e_j) - Ce_j - Ee_i + \tau_i(e_i - e_j).$ Privately optimal choices for agent i require $\partial U_i / \partial e_i = 0,$ so that the effort allocation equilibrium is identified as $e_i = 1/(C + \tau_j).$ Note that the effort level of both individuals in the absence of any enforcement mechanism is given by $e_{i,j} = 1/C$, while the socially optimal effort level is $e_{i,j} = 1/(C + E).$ The slow institutional dynamics will be in equilibrium when a change in the sanctioning preference τ_i cannot increase utility, as given by $\left(\partial U_i / \partial e_i\right)\left(de_i / d\tau_i\right) + \left(\partial U_i / \partial e_j\right)\left(de_j / d\tau_i\right) = 0.$ If the effort allocation is privately optimal, the institutional equilibrium requires that $(\tau_j - E)de_j / d\tau_i = 0.$ Provided that sanctions have an effect and $de_j / d\tau_i \neq 0,$ institutions are in equilibrium if $\tau_{i,j} = E.$ This preference value is the one that will induce individuals to deliver the socially optimal effort level. In section 4.4 we show that this equilibrium is the only evolutionarily stable one. As a result, the social dilemma will be solved.

4.3 The agent-based model

Our starting point is a small community consisting of n individuals who harvest a common pool resource; see the Appendix for a detailed description of the agent-based model. We follow the Gordon-Schaefer-model (Clark, 1990), in which harvests depend linearly on the amount of effort e_i, the size of the resource stock X, and a technology parameter γ, while the resource exhibits logistic growth. The constant price per unit of harvested biomass p and unit costs of effort c are assumed to be exogenous. Consequently, in each time step agent i receives income $M_i = e_i(p\gamma X - c)$. In principle, all actions are observable, but agents make mistakes in perceiving each other's effort levels or in implementing their intended sanctions. Sanctions have a direct effect on the utility an agent obtains. This could reflect punishments or rewards having direct monetary consequences, such as destruction of equipment or a gift, but one could also think about a gain or loss of social esteem or social status affecting utility. Therefore, we will refer to the sum of all social sanctions as the social status of one individual. We assume that imposing a punishment and giving a reward are equally costly. In our agent-based simulations, we use the Stone-Geary Cobb Douglas utility function. This utility function assumes that individuals may not only receive income from resource harvesting, but also from other economic activities. Furthermore, it takes into account that income and social status are imperfect substitutes.

Individuals increase their welfare by adjusting their effort levels through trial and error. Randomly selected agents first evaluate the utility that their current effort level gives. Then, they evaluate a slightly modified exploitation strategy and choose the one that offers higher utility. If agents change their behavior often enough, this will lead to the well-known Cournot-Nash equilibrium which can be analytically determined by maximizing utility with respect to individual effort for each agent. The social preference is imitated, as is typically the case in cultural learning (Boyd and Richerson, 2005; Henrich and McElreath, 2003). A focal player is randomly selected and matched with a co-player. If the co-player currently enjoys a higher utility, the focal agent adopts the other player's sanctioning preference. Alternatively, we have implemented simulations where the criterion for success was the average utility obtained since the last comparison round. Cultural learning also has a random component: after each encounter, there is a small probability that the social preference of the focal individual faces a small and random variation, analogous to a mutation in genetic evolution. We find that the emerging sanctioning preference leads to socially optimal exploitation (Fig. 4.1a). Consequently, the resource stock is close to what would be socially optimal (Fig. 4.1b).

The literature has identified several factors that can hinder self-governance (Ostrom, 2003; Ostrom, 2010). Figure 4.2 shows long-term values of the sanctioning preference and resource stock for different parameter configurations; see the Appendix for a description of the agent-based model and its parameters. If exploiting the resource is very profitable, a stronger sanctioning preference evolves, compensating for the increased private incentive to exploit (Fig. 4.2a). If sanctioning is very costly, cooperation persists, even though very high costs can lead to overexploitation, given that individuals make errors (Fig. 4.2b). If we relax the assumption of perfect symmetry between utility obtained from punishments and rewards ($\kappa = 1$), the results are still robust for small deviations from $\kappa = 1$ (Fig. 4.2c). Note that if punishments have a stronger effect than rewards on an individual's utility, the outcome is sub-optimal, but not far from the social optimum. If, on the contrary, individuals are less sensitive towards punishments than rewards, the enforcement mechanism suddenly collapses below a critical value.

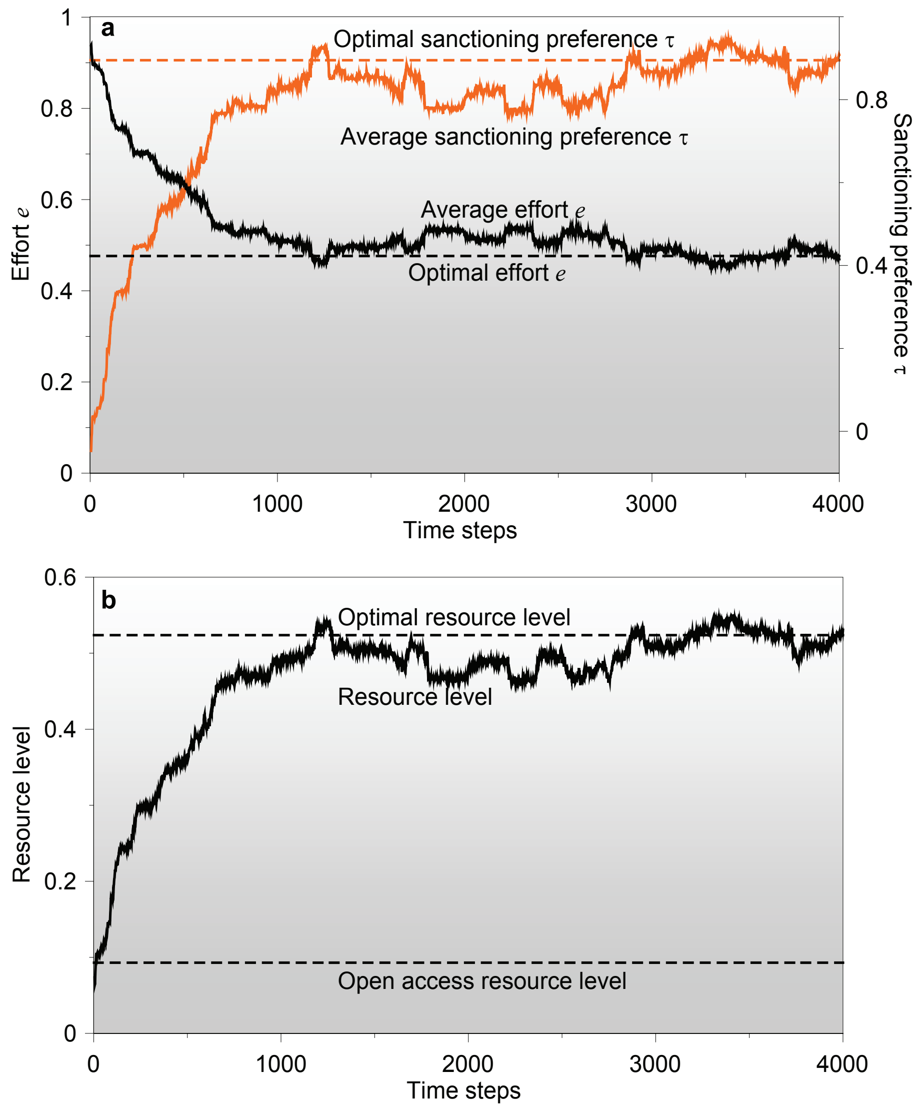

Figure 4.1. Sanctioning evolves towards a level that leads to socially optimal effort. **a**, The average sanctioning preference evolves towards a level that enforces an effort level that is socially optimal. **b**, As a result, the resource level is very close to the social optimum. In the absence of a sanctioning institution, selfish agents would overexploit, leading to the open access Nash equilibrium resource level. See Appendix for parameter values.

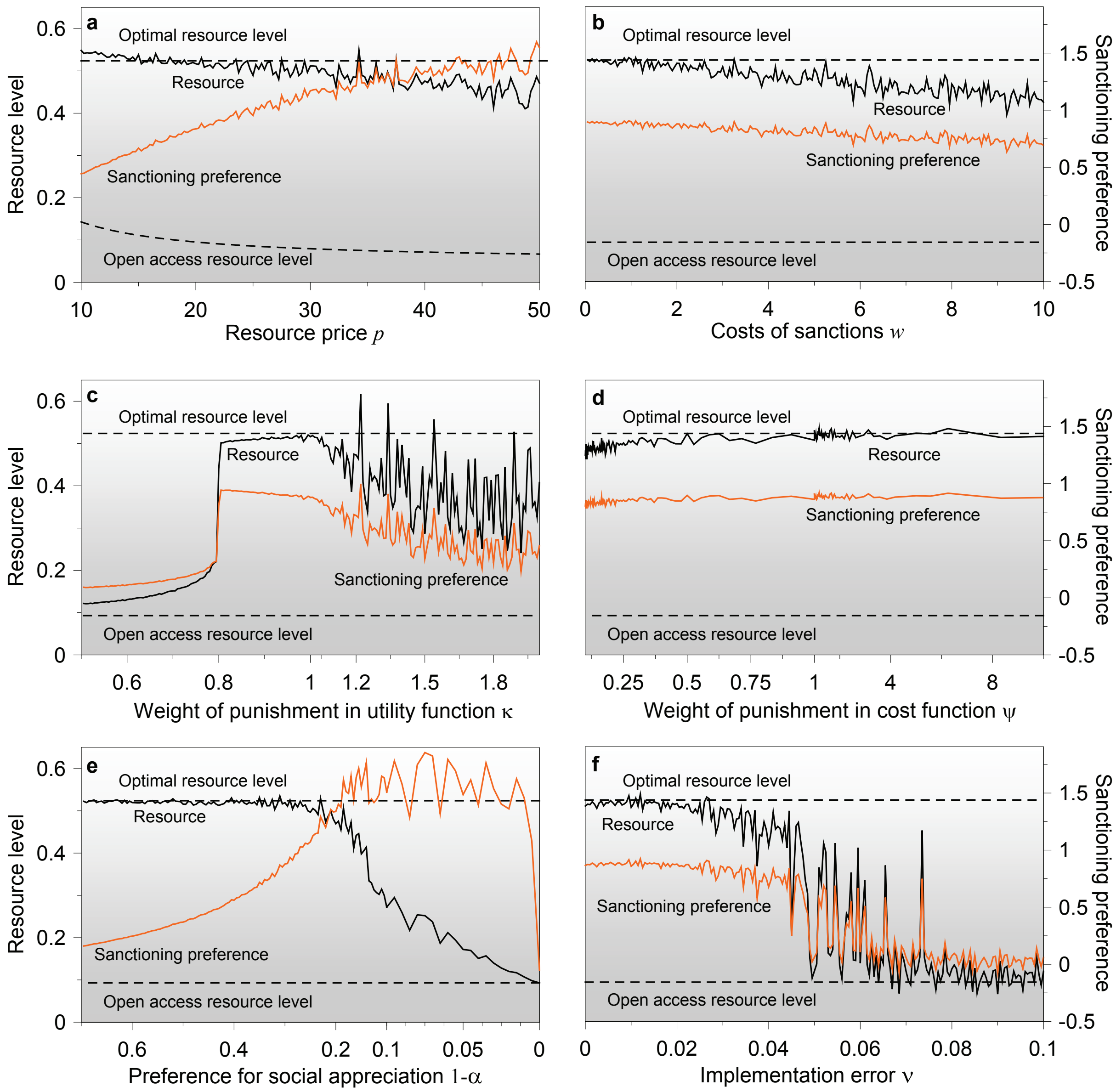

Figure 4.2. The resulting values after a fixed large number of time steps are robust towards changes in key parameters that make cooperation more difficult. **a**, If the private benefits from exploiting increase, a higher sanctioning preference compensates and assures optimal effort levels. **b**, If the costs of sanctioning increase overexploitation occurs, but for low costs the social optimum is maintained. **c**, If punishments and rewards affect the utility of the receiver in a non-symmetric way, the results are fairly robust. If sensitivity of utility towards punishment drops below a critical level, cooperation drastically collapses. **d**, If punishments and rewards affect the utility of the sender in a non-symmetric way, the results are robust. **e**, Even when individuals assign little importance to social status compared to income, the harvesting level will be very close to the social optimum. **f**, If agents make implementation errors and deviate from what their sanctioning preference dictates, results are robust. But again, there is a critical level, above which cooperation collapses. The figure shows values of the sanctioning preference and the resource level after 3000 time steps. See Appendix for parameter values.

In principle, punishing can either be more expensive or cheaper than rewarding, because individuals may fear retaliations and feel stressed by punishing or they may enjoy "sweet revenge" (Knutson, 2004). Relaxing the assumption that punishments and rewards are equally costly ($\psi = 1$) does not alter our results substantially (Fig. 4.2d). Even if individuals assign little importance to social status, and much more to income, the harvesting level will be very close to the social optimum (Fig. 4.2e). If individuals make mistakes in implementing a sanction as dictated by the sanctioning preference, the results are fairly robust (Fig. 4.2f). Again, the emerging pattern is non-linear. Above a critical error level, the enforcement system collapses and the community switches drastically from almost socially optimal exploitation to the open access exploitation level.

The egocentric sanctioning mechanism works because the incentive structure of all agents is altered in such a way that an institutional equilibrium is reached, in which marginal private benefits from resource extraction equals marginal social costs from depleting the resource stock (Fig. 4.3). The effort allocation time scale, which reflects economic and social decisions and outcomes, is in equilibrium when utility cannot be raised by changing the exploitation strategy. This implies that marginal private benefits from harvesting equal marginal private benefits (or costs) from social sanctions caused by a change in own effort level. In the institutional equilibrium economic choices are optimal. In that case, a change in the sanctioning preference τ_i is only beneficial if the utility U_i is increased due to choices made by other agents.

Figure 4.3 shows how a change in the sanctioning preference of focal individual i induces other individuals to change their effort level, which feeds back on the utility of agent i through the resource abundance and the social status. The institutional equilibrium is reached when the marginal utility change due to these two effects (change in resource abundance and social status – both due to altered behavior of other agents) balance each other out. This implies that the marginal utility gain from increased resource abundance equals the marginal loss in social status, where both are caused by changes in effort levels of other agents. If punishments and rewards are used symmetrically and own effort level is used as a moral benchmark, a change in effort of agent i has the opposite marginal effect on the social status of i than a change in effort of a co-player. This effectively links the effort allocation and the institutional equilibrium. If a change in the resource stock affects all agents in the same way, marginal private benefits from exploitation equal the marginal resource externalities imposed on others. This is a necessary and sufficient condition to overcome a social dilemma. Our findings are in line with earlier work that has shown that if evolutionary processes change slower than behavior, cooperation can be sustained (Akçay et al., 2009).

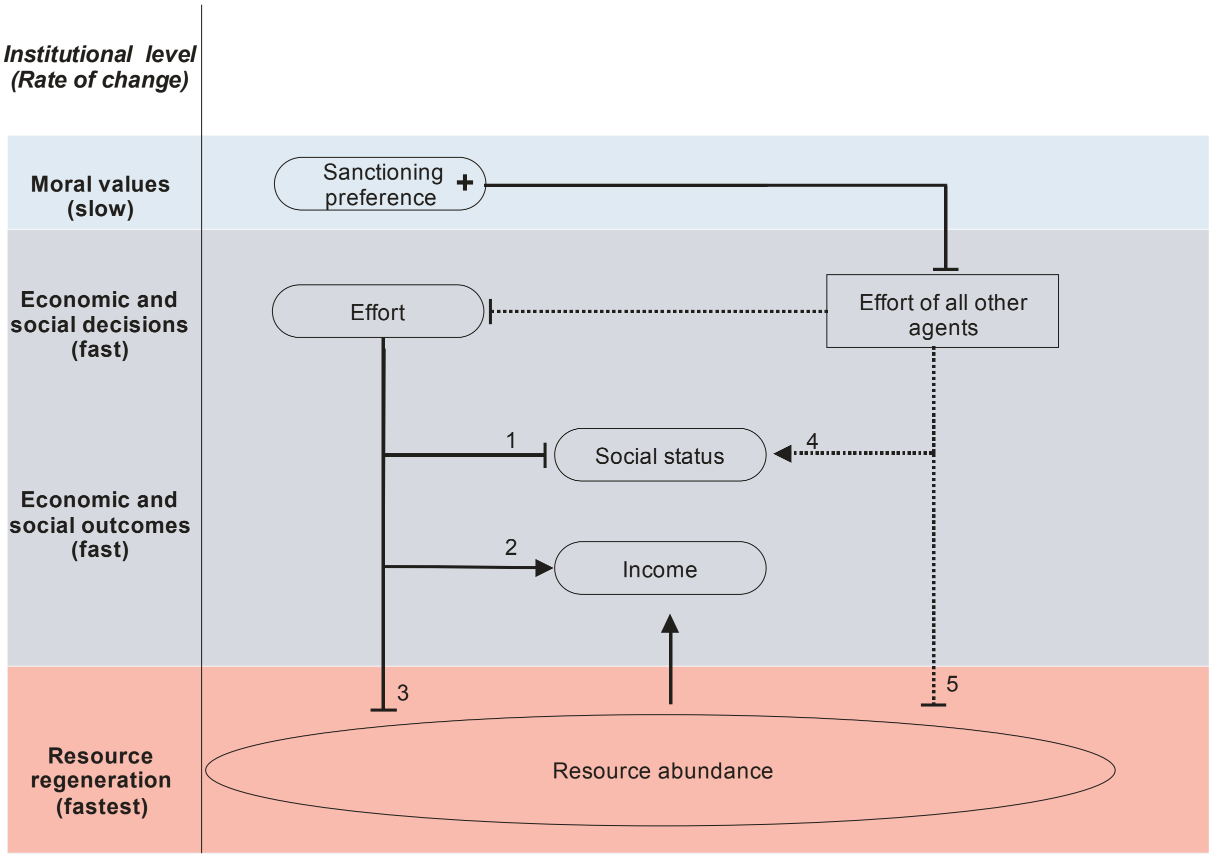

Figure 4.3. The feedback structure of a higher sanctioning preference of agent *i*. Positive feedbacks are indicated by arrows, while negative feedbacks are indicated by bars. Choices made by agent *i* are depicted by solid lines, while actions beyond the control of agent *i* are depicted by dotted lines. In response to a higher sanctioning preference, all individuals reduce their effort levels (squared box), while agent *i* responds to this by increasing his own effort. These decisions affect social and economic outcomes. If the other agents hold, on average, a positive sanctioning preference, agent *i* suffers a loss of social status due to higher effort (1). Higher effort leads to more income for agent *i* (2), but a lower resource level partly squanders income of agent *i* (3). The marginal losses from a change in social status (1) will equal the marginal gains from an increase in income (2 and 3) if decisions are optimal. What remains, are economic repercussions beyond the control of agent *i*. The reduced effort by other agents leads to a loss of social status of agent *i* (4), while income is raised due to a higher resource level (5). In the institutional equilibrium the marginal gains from a change in social status (4) equal the marginal gains from a higher resource stock (5). The social dilemma will be overcome if the marginal effects of a change in private income by harvesting, given by (2) and (3), equal the marginal social costs, given by (5). This condition is met if punishments and rewards are used symmetrically with own effort as a moral benchmark, and the marginal effects of (1) and (4) sum up to zero.

We have already identified key parameters that may form obstacles for the evolution of a successful enforcement mechanism. Additionally, we find that the sanctioning preference compensates for the fact that harsher sanctions are required to sustain cooperation in larger groups (Fig. 4.4a). However, beyond a certain group size, overexploitation occurs, given that strong sanctions are too costly if agents make mistakes. If there is a chance that some harvesting goes undetected, overexploitation occurs (Fig. 4.4b). This shows, like earlier work (Nowak and Sigmund, 1998), that a lack of monitoring possibility can undermine cooperation. In a similar vein, an error term that reflects the failure to perceive the correct effort levels, and send each partner a sanction at random (dictated by a fraction of their sanctioning preference) forms a critical obstacle (Fig. 4.4c).

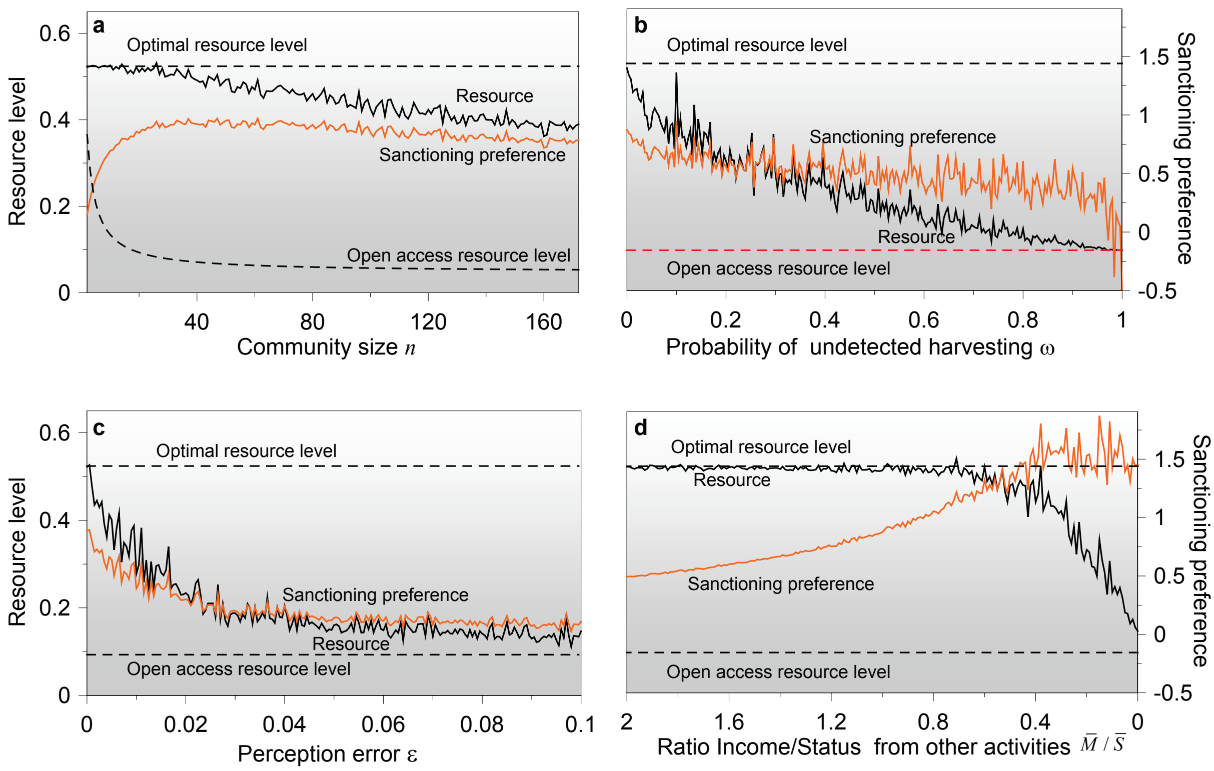

Figure 4.4. Several key parameters can be identified that undermine cooperation and affect the long-term results. **a,** If group size increases above a critical level the social optimum will not be achieved. **b,** If monitoring is imperfect and some harvesting activity goes undetected, massive overexploitation occurs. **c,** If agents make errors in perceiving exploitation and deviate from what their sanctioning preference recommends them to do, overexploitation occurs. **d,** If income from resource harvesting is much more important than social status coming from that activity, harsher sanctions evolve. There is a point where the required sanctions are too high and cooperation cannot be sustained. The figure shows values of the sanctioning preference and the resource level after 3000 time steps. See Appendix for parameter values.

Also the fact that individuals have outside options (Tarui, 2007) that deliver income and social status is important. If income from resource harvesting is much more important than social status coming from that activity, harsher sanctions are required and the sanctioning system may not evolve to these high levels (Fig. 4.4d).

4.3.1 Robustness of the agent-based model

Here, we relax the assumption that the resource system is always in equilibrium by introducing a resilience parameter ξ that measures the speed at which the resource regenerates and approaches a steady state. Resilience ξ is considered to be very high when the unharvested resource regenerates from 1% of carrying capacity to 90% of carrying capacity. The other extreme is no resilience at all; the resource never regenerates and would stay at 1% of carrying capacity. In between we have divided the actual regeneration by the maximum (90%) to obtain a measure between zero and 1. We find that only when resilience is very weak, the enforcement mechanism collapses (Fig. 4.5a). Furthermore, we have analyzed the case where each agent has only a small number of trials to learn about appropriate effort levels. Even if this is the case, the results are still very close to the social optimum (Fig. 4.5b).

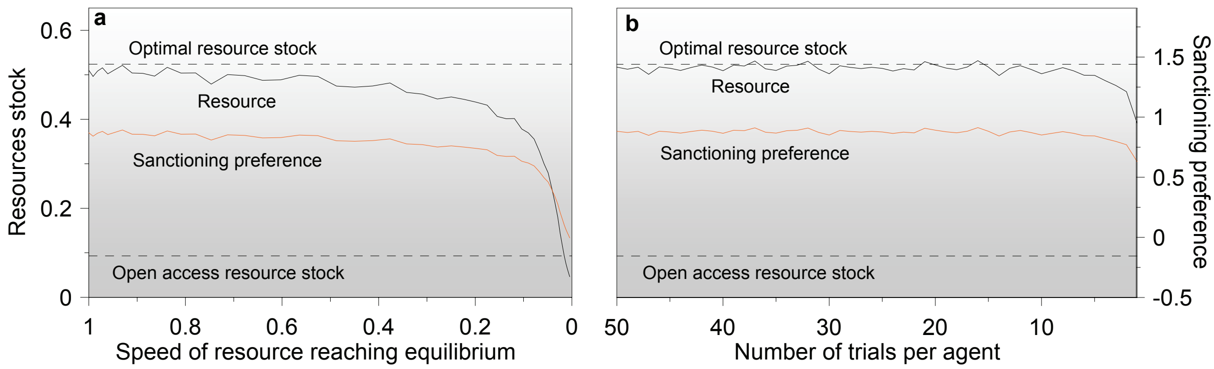

Figure 4.5. Relaxing the assumption that the system is in equilibrium. **a,** If the resource responds slowly to exploitation and resilience is very low, too much of the resource is extracted. At a critical level, cooperation breaks down, and massive overexploitation occurs. **b**, If individuals have only few trials to revise their strategy, the resource is still only moderately exploited. The figure shows values of the sanctioning preference and the resource level after 3000 time steps. See Appendix for parameter values.

Surprisingly, if not own behavior, but group behavior is used as a benchmark for deciding whom to punish or reward, the emerging harvest scenario is further away from the social optimum, compared to a situation when only own behavior is used (Fig. 4.6).

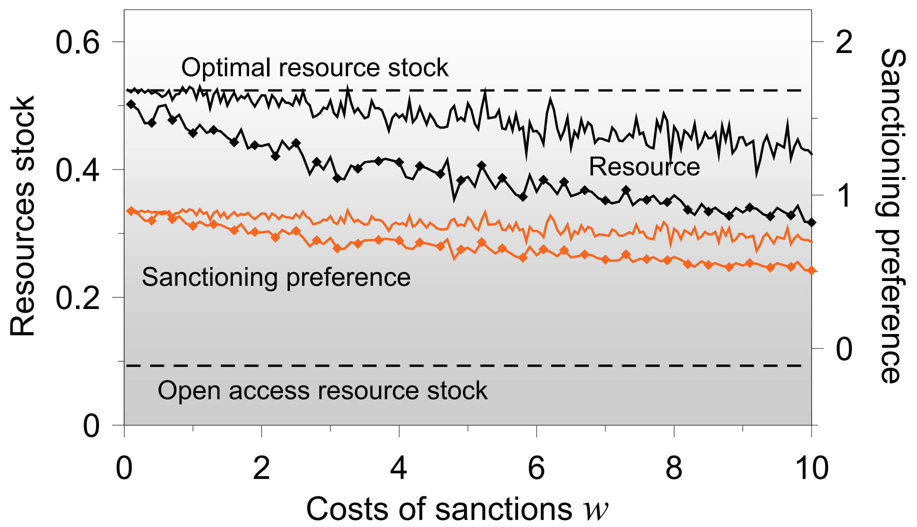

Figure 4.6. Not own behavior, but group behavior is used as a moral benchmark. If individuals do not use own effort levels (solid lines), but average effort in the population (diamonds) as a moral benchmark, the evolving sanctioning preference and effort levels will be further away from the social optimum. This is especially the case if the costs of sanctioning go up. The figure shows values of the sanctioning preference and the resource level after 3000 time steps. See Appendix for parameter values.

Interestingly, if individuals do not imitate each other's sanctioning preferences, but infer them through trial and error, the emerging exploitation level will still lead to optimal resource abundance (Fig.4.7b). However, the individuals in the community will not hold the same sanctioning preference. Instead we see social polymorphisms arising, ranging from high sanctions to negative preference values (Fig. 4.7a), which reflect anti-social punishment (Herrmann et al., 2008). Analytically, this can indeed be identified as a branching point.

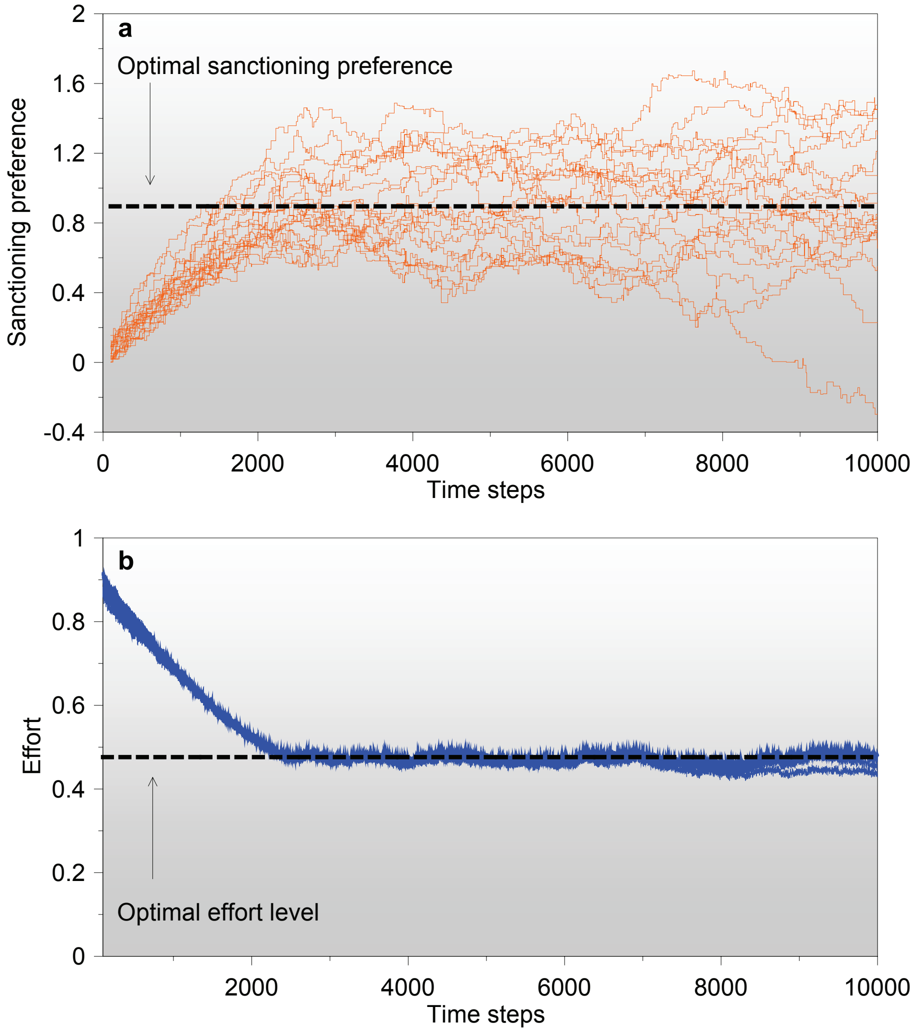

Figure 4.7. The sanctioning preference is learned through trial and error. **a,** If individuals do not imitate each other's sanctioning preferences, but infer them through trial and error sanctioning preferences diverge. On average, sanctioning preferences will still be optimal, but while some individuals have a very strong preference, others have a very weak one. **b,** In spite of holding different sanctioning preferences, individual effort levels are homogenous and close to the optimum.

4.4 A general analytical model

We start our analysis with assuming that two agents face a social dilemma. Later, this will be extended to a community consisting of $n > 2$ individuals. In that case, some simplifying assumptions are necessary to keep the system analytically tractable. Since changes in moral preferences are rare, we assume that everyone in society holds the same sanctioning preference when a single individual changes his mind and a new preference occurs (through mutation). This assumption is obviously more realistic for small communities with little contact to the outside world, where social cohesion is typically strong, but it may also occur in large societies.[15] This analytical framework allows us to analyze a continuum of strategies. The analysis is feasible, because we only compare two strategies at the same time. In our model each individual holds a sanctioning preference τ_i that evolves over time. If a new sanctioning preference occurs, the new strategy may offer a higher utility than the old one. In such a case, it is likely that the new strategy "invades" and spreads in the population. We assume that a better strategy always replaces an inferior strategy. If one wants to analyze more rigorously the role of stochastic processes in small populations one could rely on Nowak et al. (2004). In our setup, evolution of the sanctioning preference comes to a halt, when a preference change of agent i will not increase utility U_i anymore and $dU_i / d\tau_i$, also referred to as the selection gradient $D(\tau_i)$, is equal to zero. This will be analyzed in section 4.4.2. While the assumption that individuals maximize their own utility is common in economics, it is less clear whether this also holds for the willingness to discipline peers, or for a moral system in general. More often, individuals compare their own social norms or moral preferences with norms of other community members and try to outcompete them, striving for relative superiority of their moral values. In that case, the selection gradient may be defined as $d(U_i - U_j) / d\tau_i$, as will be assumed in section 4.4.3. As long as the selection gradient is positive, the strategy i will be established in the population. Therefore, the selection gradient allows us not only to find the evolutionary equilibria, but also the evolutionary dynamics themselves. The change of the sanctioning preference τ over time can be expressed as an ordinary differential equation known as the canonical equation of adaptive dynamics (Champagnat et al., 2002; Dieckmann and Law, 1996) of the type

$$d\tau / dt = kD(\tau_{i,j})\Big|_{\tau=\tau_i,\tau_j} \tag{4.1}$$

where $D(\tau_{i,j})$ is given by either $d(U_i - U_j) / d\tau_i$ or $dU_i / d\tau_i$. Parameter k scales the rate of evolutionary change and depends on the probability of a mutation, the number of individuals, and the variance of a mutation. For constant population sizes, it can be set to 1 without loss of generality (Doebeli et al., 2004). At $d\tau / dt = 0$ the so-called fitness landscape is flat and therefore evolution may come to a halt, depending on the stability conditions that will be derived following Geritz et al. (1998). An evolutionary endpoint is reached if the following two conditions are met. First, an evolutionary stable strategy (ESS) is required, i.e. the selection gradient is at a (local) maximum. Second, that equilibrium needs to be a convergent stable strategy (CSS), i.e. that equi-

[15] Think about Holden Caulfield in J.D. Salinger's *The Catcher in the Rye.*

librium needs to be an attractor. If a strategy is CSS and ESS, it can be identified as an evolutionary endpoint known as a continuously stable strategy. We analyze the stability of the evolutionary process in section 4.4.4.

4.4.1 A model of a general social dilemma

A community consisting of $n > 1$ individuals faces a social dilemma. Individuals invest effort in an activity that has repercussions on the income of all other community members. The externality coming from this activity may be positive, in the form of a public good, or negative, in the form of a common pool resource. Self-interested individuals typically ignore this externality giving rise to the so-called Tragedy of the Commons (Hardin, 1968). As a result, community payoff will be sub-optimal.

In a fairly generic model of a social dilemma, agent i chooses individual effort e_i that delivers private benefits (or costs) B_i, while it bears a negative (positive) externality $\sum_{j \neq i} E_{ji}$ on each of the other agents present. Note that E_{ji} is the externality that agent i imposes on agent j. If $\partial B_i / \partial e_i > 0$ and $\partial E_{ji} / \partial e_i < 0$ the social dilemma can be considered a common pool resource game. If $\partial B_i / \partial e_i < 0$ and $\partial E_{ji} / \partial e_i > 0$ the social dilemma resembles a public goods game. A selfish agent would choose effort such that his private benefits are maximal and $\partial B_i / \partial e_i = 0$, while it would be socially optimal if everyone chose

$$\partial B_i / \partial e_i + \sum_{j \neq i} E_{ji} / \partial e_i = 0 \tag{4.2}$$

4.4.2 The evolution of a sanctioning institution for a general social dilemma when absolute utility determines the evolutionary success

In this sub-section we analyze the situation where absolute utility is the criterion for evolutionary success. If a new sanctioning preference enhances the utility of an agent, it will be pursued. Therefore, the selection gradient is given by $dU_i / d\tau_i$.

The two player case

In the simplest case we ignore that imposing sanctions is costly and look at a situation where only two agents face a social dilemma $(n = 2)$. Individual i holds a sanctioning preference τ_i that determines the size of sanctions or rewards $S_{ji}(e_i, e_j, \tau_i)$ agent i gives to agent j, depending on the effort level of the two agents. Total utility of agent i is given by

$U_i(e_i, e_{j,}, \tau_j) = B_i(e_i) + E_{ij}(e_j) + S_{ij}(e_i, e_j, \tau_j)$, for $i, j = \{1, 2\}$ and $i \neq j$ [16].

Agents ignore the externality that they impose on others, but they are sensitive towards the social sanctions that are imposed on them. Therefore, decisions are embedded in a social context. Given the presence of social sanctions, privately optimal choices require

$$\frac{\partial U_i}{\partial e_i} = \frac{\partial B_i}{\partial e_i} + \frac{\partial S_{ij}}{\partial e_i} = 0 . \tag{4.3}$$

The first order condition in (4.3) gives the effort allocation equilibrium. Comparing (4.2) with (4.3) reveals that the social optimum will be achieved if and only if

$$\frac{\partial S_{ij}}{\partial e_i} = \frac{\partial E_{ji}}{\partial e_i}. \tag{4.4}$$

We assume that agents change their private investment much more frequently than their sanctioning preference. Mathematically, this implies that effort is a function of the sanctioning preference, as given by $e(\tau)$. The slow institutional dynamics are in equilibrium when a change in the sanctioning preference does not increase utility anymore, as given by

$$\frac{dU_i}{d\tau_i} = \frac{\partial B_i}{\partial e_i}\frac{de_i}{d\tau_i} + \frac{\partial S_{ij}}{\partial e_i}\frac{de_i}{d\tau_i} + \frac{\partial E_{ij}}{\partial e_j}\frac{de_j}{d\tau_i} + \frac{\partial S_{ij}}{\partial e_j}\frac{de_j}{d\tau_i} = 0 . \tag{4.5}$$

Since preferences change much slower than economic decisions, we assume that condition (4.3) is met when (4.5) is evaluated. Therefore, (4.5) simplifies to

$$\left(\frac{\partial S_{ij}}{\partial e_j} + \frac{\partial E_{ij}}{\partial e_j}\right)\frac{de_j}{d\tau_i} = 0. \tag{4.6}$$

Given that $de_j / d\tau_i \neq 0$, institutions are in equilibrium when

$$\frac{\partial S_{ij}}{\partial e_j} + \frac{\partial E_{ij}}{\partial e_j} = 0 . \tag{4.7}$$

If the nature of the externality each individual imposes on the other is only dependent on individual effort levels but not on other specific characteristics, we can assume that

$$\frac{\partial E_{ij}}{\partial e_j} = \frac{\partial E_{ji}}{\partial e_i}. \tag{4.8}$$

[16] From now on, arguments will be omitted, unless omitting them may cause confusion.

Comparing (4.7) and (4.8) with (4.4) shows that the social dilemma will be solved when the following symmetry condition is met:

$$\frac{\partial S_{ij}}{\partial e_i} = -\frac{\partial S_{ij}}{\partial e_j}. \tag{4.9}$$

In that case, $\partial B_i / \partial e_i + \partial E_{ji} / \partial e_i = 0,$ which is the condition for social optimality. The symmetry that is required in condition (4.9) is met if social sanctions are based on punishments and rewards for which the own behavior is used as a moral benchmark. This equilibrium involves both players delivering the socially optimal amount of effort. In section 4.4.4 we show how this mechanism works for a common pool resource game and in section 4.4.5 for a public goods game. We also demonstrate that this equilibrium is the only evolutionarily stable one.

The n>2 player case and costly sanctions

We assume that the community consists of $n > 2$ agents. Additionally, we assume that decentralized sanctioning occurs bilaterally and is costly. As a result, agent i receives sanctions $S_{ij}(e_i, e_j, \tau_j)$ from agent j, while paying costs $C_{ij}(e_i, e_j, \tau_i)$ for any sanctions he may impose on j. Consequently, the utility function U_i of agent i is given by

$$U_i(e,\tau) = B_i(e_i) + \sum_{j\neq i} E_{ij}(e_j) + \sum_{j\neq i} S_{ij}(e_i, e_j, \tau_j) - \sum_{j\neq i} C_{ij}(e_i, e_j, \tau_i). \tag{4.10}$$

Besides, we analyze the case where punishments and rewards are used in a symmetric way and own effort levels are used as a moral benchmark that separates good from bad behavior. Therefore, we assume that

$$S_{ij} = \tau_j \left(e_j - e_i\right), \tag{4.11}$$

which implies that $\partial S_{ij} / \partial e_i = -\partial S_{ij} / \partial e_j.$ Additionally, we assume that costs are non-negative and depend on sanctions imposed. For convenience we choose the functional form

$$C_{ij} = \left(\tau_i \left(e_i - e_j\right)\right)^2, \tag{4.12}$$

which implies that $\partial C_{ij} / \partial e_i = -\partial C_{ij} / \partial e_j.$ If investment decisions are privately optimal the following condition must hold:

$$\frac{\partial U_i}{\partial e_i} = \frac{\partial B_i}{\partial e_i} + \sum_{j\neq i}\left(\frac{\partial S_{ij}}{\partial e_i} - \frac{\partial C_{ij}}{\partial e_i}\right) = 0. \tag{4.13}$$

The effort allocation equilibrium is reached when (4.13) holds for all n agents. This will be socially optimal if and only if

$$\sum_{j\neq i}\left(\frac{\partial S_{ij}}{\partial e_i}-\frac{\partial C_{ij}}{\partial e_i}\right)=\sum_{j\neq i}\frac{\partial E_{ji}}{\partial e_i}. \tag{4.14}$$

Evolution comes to a halt, when a preference change will not increase utility anymore and $dU_i / d\tau_i = 0$. If effort allocation is optimal and hence in equilibrium, (4.13) holds, and the potential evolutionary endpoint is given by

$$\frac{dU_i}{d\tau_i}=\sum_{j\neq i}\left(\frac{\partial E_{ij}}{\partial e_j}+\frac{\partial S_{ij}}{\partial e_j}-\frac{\partial C_{ij}}{\partial e_j}\right)\frac{de_j}{d\tau_i}-\sum_{j\neq i}\left(\frac{\partial C_{ij}}{\partial \tau_i}\right)=0. \tag{4.15}$$

If effort levels of community members are fairly homogeneous and $e_i \approx e_j$, it follows from (4.12) that $\partial C_{ij} / \partial \tau_i \approx 0$. If individuals are sensitive towards sanctions, i.e. $de_j / \partial \tau_i \neq 0$ it follows from (4.15) that

$$\sum_{j\neq i}\left(\frac{\partial E_{ij}}{\partial e_j}+\frac{\partial S_{ij}}{\partial e_j}-\frac{\partial C_{ij}}{\partial e_j}\right)\approx 0. \tag{4.16}$$

Using (4.8), (4.11), (4.12), and (4.16) yields

$$\sum_{j\neq i}\left(\frac{\partial E_{ji}}{\partial e_i}-\frac{\partial S_{ij}}{\partial e_i}+\frac{\partial C_{ij}}{\partial e_i}\right)\approx 0, \tag{4.17}$$

which can be rearranged to

$$\sum_{j\neq i}\left(\frac{\partial S_{ij}}{\partial e_i}-\frac{\partial C_{ij}}{\partial e_i}\right)\approx\sum_{j\neq i}\frac{\partial E_{ji}}{\partial e_i}. \tag{4.18}$$

This is indeed the condition for social optimality; see (4.14).

4.4.3 The evolution of a sanctioning institution in a general social dilemma when relative utility determines the evolutionary success

In this section we analyze the case, where individuals try to outcompete each other. Thus, an evolutionary endpoint of the institutional dynamics may be reached when the selection gradient $d(U_i - U_j) / d\tau_i$ is zero.

The two player case

Let us again start with the case where only two agents are present. Agent i tries to outcompete agent j. Utility of agent i is given by

$$U_i(e_i,e_j,\tau_i,\tau_j)=B_i(e_i)+E_{ij}(e_j)+S_{ij}(e_i,e_j,\tau_j)-C_{ij}(e_i,e_j,\tau_i). \tag{4.19}$$

If effort allocation is optimal, the first order condition for both agents is

$$\frac{\partial U_i}{\partial e_i}=\frac{\partial B_i}{\partial e_i}+\left(\frac{\partial S_{ij}}{\partial e_i}-\frac{\partial C_{ij}}{\partial e_i}\right)=0 \text{ for } i,j=\{1,2\} \text{ and } i\neq j. \tag{4.20}$$

Social optimality will be reached if and only if

$$\left(\frac{\partial S_{ij}}{\partial e_i}-\frac{\partial C_{ij}}{\partial e_i}\right)=\frac{\partial E_{ji}}{\partial e_i} \text{ for } i,j=\{1,2\} \text{ and } i\neq j. \tag{4.21}$$

The institutional equilibrium is given by

$$\frac{d\left(U_i-U_j\right)}{d\tau_i}=\frac{\partial U_i}{\partial e_j}\frac{de_j}{d\tau_i}-\frac{\partial U_j}{\partial e_i}\frac{de_i}{d\tau_i}+\frac{\partial U_i}{\partial \tau_i}-\frac{\partial U_j}{\partial \tau_i}=0. \tag{4.22}$$

If the sanctioning preferences of the two agents are very similar and $\tau_i \approx \tau_j$, this implies that the effort levels are also very similar and $e_i \approx e_j$. From (4.11) and (4.12) it directly follows that in that case $\partial S_{ji}/\partial\tau_i \approx 0$ and $\partial C_{ij}/\partial\tau_i \approx 0$. Therefore, (4.22) reduces to

$$\frac{d\left(U_i-U_j\right)}{d\tau_i}=\frac{\partial U_i}{\partial e_j}\frac{de_j}{d\tau_i}-\frac{\partial U_j}{\partial e_i}\frac{de_i}{d\tau_i}\approx 0.$$

Using assumption (4.8), (4.11) and (4.12), the institutional equilibrium is given by

$$\frac{d\left(U_i-U_j\right)}{d\tau_i}=\left(\frac{de_j}{d\tau_i}-\frac{de_i}{d\tau_i}\right)\left(-\frac{\partial S_{ij}}{\partial e_i}+\frac{\partial C_{ij}}{\partial e_i}+\frac{\partial E_{ji}}{\partial e_i}\right)\approx 0. \tag{4.23}$$

Provided that the first bracket on the right hand side is non zero, using (4.23) will lead to the social optimum; see (4.21).

4.4.4 An example of a simple common pool resource game

A community comprises n individuals that engage in the same economic activity. Individual i receives a return b for each unit of invested effort e_i. We also assume that resource harvesting gives rise to a negative externality: the return any agent receives on her effort negatively depends on the total effort put in by all community members. The corresponding utility U_i is given by

$$U_i(e) = e_i\left(b - c\sum_{j=1}^{n} e_j\right). \tag{4.24}$$

Here, c can be thought of as the costs of congestion (for example because agents interfere with each other or have to compete for the bests spots, but also as any other negative interaction between contemporaneous harvesting activities (Clark, 1990; Wilson, 1982)). When social pressure is absent and every agent maximizes his individual profit, each agent faces the following best response function: $e_i = (b - c\sum_{j \neq i} e_j) / 2c$. Assuming identical agents, the open access equilibrium effort level equals $e_{Nash} = b/(n+1)c,$ while the aggregated effort is $nb/(n+1)c.$ If $n > 1$, this is larger than the socially optimal aggregate, which is $b/2c.$ If a cooperative system with n users agrees on a common effort level to share profits equally, the symmetric individual effort level would be $e_{opt} = b/2cn.$

Effort allocation

We assume that utility for agent i is given by

$$U_i(e,\tau) = B_i(e_i) + \sum_{j \neq i} E_{ij}(e_j) + \sum_{j \neq i} S_{ij}(e_i, e_j, \tau_j) - \sum_{j \neq i} C_{ij}(e_i, e_j, \tau_i), \tag{4.25}$$

where

$$B_i(e_i) = w_1 e_i (b - c e_i), \tag{4.26}$$

$$E_{ij}(e_j) = -w_1 c e_i e_j, \tag{4.27}$$

$$S_{ij}(e_i, e_j, \tau_j) = w_2 \tau_j (e_j - e_i), \tag{4.28}$$

$$C_{ij}(e_i, e_j, \tau_i) = w_3 \left(\tau_i (e_i - e_j)\right)^2. \tag{4.29}$$

The parameters w_1, w_2, w_3 are weights in the utility function that determine the preference for income, social appreciation, and the burden of sanctioning others. The components for private benefits (4.26) and the negative externality (4.27) come directly from the common pool resource game (4.24). The utility obtained from receiving and sending sanctions is modeled in the same way as before; see (4.11) and (4.12). A necessary condition for private utility maximization is

$$\frac{\partial U_i}{\partial e_i} = \frac{\partial B_i}{\partial e_i} + \sum_{j \neq i} \left(\frac{\partial E_{ij}}{\partial e_i} + \frac{\partial S_{ij}}{\partial e_i} - \frac{\partial C_{ij}}{\partial e_i} \right) = 0. \tag{4.30}$$

To keep the system analytically tractable, we assume that the community is initially in equilibrium and all individuals hold the same sanctioning preference τ_r (referred to as the preference of the resident). Then, one individual changes his sanctioning preference (referred to as the preference of the mutant). In what follows, we will determine whether this new sanctioning preference – denoted by τ_m – can spread in the population. (4.30) can be rewritten for the mutant and the resident as

$$\frac{\partial U_m}{\partial e_m} = \frac{\partial B_m}{\partial e_m} + (n-1)\left(\frac{\partial E_{mr}}{\partial e_m} + \frac{\partial S_{mr}}{\partial e_m} - \frac{\partial C_{mr}}{\partial e_m}\right) = 0, \tag{4.31}$$

$$\frac{\partial U_r}{\partial e_r} = \frac{\partial B_r}{\partial e_r} + (n-2)\left(\frac{\partial E_{r\bar{r}}}{\partial e_r} + \frac{\partial S_{r\bar{r}}}{\partial e_r} - \frac{\partial C_{r\bar{r}}}{\partial e_r}\right) + \left(\frac{\partial E_{rm}}{\partial e_r} + \frac{\partial S_{rm}}{\partial e_r} - \frac{\partial C_{rm}}{\partial e_r}\right) = 0, \tag{4.32}$$

where indices with a bar refer to the $(n-2)$ other residents that are present in the community (vis-à-vis a random resident). Given that all agents are assumed to make decisions at the same time, their effort level will converge to the same level and therefore $e_r = \bar{e}_r$. Combining (4.26)-(4.29) and (4.31)-(4.32) yields the optimal exploitation strategies for the mutant and the residents. Thus, (4.31) and (4.32) can be given as

$$\frac{\partial U_m}{\partial e_m} = w_1\left[b-(n-1)ce_r - 2ce_m\right] - \tau_r w_2 (n-1) - 2w_3\tau_m^{\;2}(n-1)(e_m - e_r) = 0 \tag{4.33}$$

and

$$\frac{\partial U_r}{\partial e_r} = w_1\left[b - c(ne_r + e_m)\right] - \tau_r w_2(n-2) - w_2\tau_m - 2w_3\tau_r^{\;2}(e_r - e_m) = 0. \tag{4.34}$$

This allows us to derive the reaction function for the two different strategies:

$$e_m = \frac{w_1 b - e_r\left(w_1 c(n-1) - 2w_3(n-1)\tau_m^{\;2}\right) - w_2\tau_r(n-1)}{2w_1 c + 2w_3(n-1)\tau_m^{\;2}}, \tag{4.35}$$

$$e_r = \frac{w_1 b - e_m\left(w_1 c - 2w_3\tau_m^{\;2}\right) - w_2\tau_r(n-2) - w_2\tau_m}{w_1 cn + 2w_3\tau_r^{\;2}}. \tag{4.36}$$

The effort differences are given by

$$e_m - e_r = \frac{w_2(\tau_m - \tau_r)}{w_1 c + 2w_3\left(n\tau_m^{\;2} + \left(\tau_r^{\;2} - \tau_m^{\;2}\right)\right)}. \tag{4.37}$$

If the two sanctioning preferences are very similar and $\tau_m \approx \tau_r$, this implies that the effort levels are also very similar and $e_m \approx e_r$. Then the equilibrium effort level is given by

$$e\big|_{\tau_m = \tau_r} = \frac{bw_1 - \tau w_2(n-1)}{cw_1(n+1)}. \tag{4.38}$$

Comparing (4.38) with the socially optimal effort level derived earlier reveals that a sanctioning preference of $\tau^* = bw_1 / (2nw_2)$ would lead to the social optimum.

The institutional equilibrium

The starting point of the institutional time scale is the utility function (4.25). In the institutional time scale, economic decisions are assumed to be optimal and all individuals with the same sanctioning preference will deliver the same effort level; see (4.35) and (4.36). In that case, a change in the sanctioning preference increases relative utility if

$$\left.\frac{d\left(U_m - U_r\right)}{d\tau_m}\right| = \frac{\partial U_m}{\partial e_r}\frac{de_r}{d\tau_m} - \frac{\partial U_r}{\partial e_m}\frac{de_m}{d\tau_m} + \frac{\partial U_m}{\partial \tau_m} - \frac{\partial U_r}{\partial \tau_m} > 0. \tag{4.39}$$

Using the assumption that $\tau_m \approx \tau_r$ and (4.26) – (4.29) gives a potential evolutionary endpoint as

$$\left.\frac{d\left(U_m - U_r\right)}{d\tau_m}\right|_{\tau=\tau_m,\tau_r} = \left(\frac{\partial e_m}{\partial e_r}\frac{de_r}{d\tau_m} - \frac{de_m}{d\tau_m}\right)\left(-cw_1 e + w_2\tau\right) \approx 0. \tag{4.40}$$

Provided that the first term on the right hand side is non-zero, (4.40) will be zero when $\tau = cw_1 e / w_2$. Using (4.38) shows that the fitness landscape is flat at $\tau = bw_1 / (2nw_2)$, which is a potential evolutionary endpoint. This is indeed the preference value that leads to socially optimal exploitation. To make sure that this equilibrium is evolutionarily stable and the sanctioning preference will indeed converge towards that value, we will analyze the selection gradient (4.40) more thoroughly. The evolutionary process is inherently stochastic, but since mutations are assumed to be small $(\tau_m \approx \tau_r)$, evolutionary trajectories can be described with an ordinary differential equation; cf. Dercole and Rinaldi (2008) and Dieckmann and Law (1996). The deterministic path of the evolutionary dynamics can be directly described by the selection gradient, as given by

$$\frac{d\tau}{dt} = \left.\frac{d\left(U_m - U_r\right)}{d\tau_m}\right|_{\tau=\tau_m,\tau_r}, \tag{4.41}$$

see also (4.1). Equation (4.41) shows the direction of the (residential) trait in the population. Note that the evolutionary direction does not point into the direction that is best for the resident (or the whole community), but what is best for the mutant. Combining (4.36), (4.37), (4.40), and (4.41) gives the canonical equation of adaptive dynamics (Champagnat et al., 2002; Dieckmann and Law, 1996) as

$$\frac{d\tau}{dt} = \left(\frac{w_2\left(bw_1 - 2w_2 n\tau\right)}{\left(n+1\right)\left(cw_1 + 2nw_3\tau^2\right)} \right). \tag{4.42}$$

At $d\tau / dt = 0$ the fitness landscape is flat and therefore evolution may come to a halt. We find that $d\tau / dt = 0$ if $\tau = bw_1 / (2nw_2)$ or $\tau = \pm\infty$. Following Geritz et al. (1998), the first solution is evolutionarily stable, because

$$\partial^2\left(U_m - U_r\right) / \partial^2\tau_m \Big|_{\tau_m, \tau_r = bw_1/(2nw_2)} < 0. \tag{4.43}$$

It converges to that equilibrium, because

$$\partial^2\left(U_m - U_r\right) / \partial^2\tau_r \Big|_{\tau_m, \tau_r = bw_1/(2nw_2)} > 0, \tag{4.44}$$

$$\partial^2\left(U_m - U_r\right) / \partial^2\tau_m - \partial^2\left(U_m - U_r\right) / \partial^2\tau_r \Big|_{\tau_m, \tau_r = bw_1/(2nw_2)} < 0. \tag{4.45}$$

Hence, $\tau = bw_1 / (2nw_2)$ can be identified as continuously stable strategy (Geritz et al., 1998). Repeating the same steps for the other solution $\tau = \infty$ reveals that $\tau = \infty$ is evolutionarily stable, but it will never converge to that point. Therefore, the adaptive dynamics will lead to the sanctioning preference that maximizes social welfare.

4.4.5 An example of a public goods game

Again, a community comprises n individuals that engage in the same economic activity. Each individual receives a return b for each unit of invested effort by a community member. The unit costs c are, however, private. The corresponding utility U_i is given by

$$U_i\left(e\right) = b\sum_{j=1}^{n} e_j - ce_i^2. \tag{4.46}$$

The selfish Nash equilibrium in absence of punishment and reward is:

$$e_{Nash} = b / (2c), \tag{4.47}$$

while the socially optimal provision can be given by

$$e_{opt} = nb / (2c). \tag{4.48}$$

Following the same steps as before, the following canonical equation of adaptive dynamics can be identified as

$$d\tau / dt = \left(\frac{-w_2\left(bw_1 + w_2\tau\right)}{2\left(cw_1 + nw_3\tau^2\right)} \right). \tag{4.49}$$

We find that there is one continuously stable strategy $\tau = -bw_1 / w_2$. This equilibrium involves all players delivering the socially optimal amount of effort.

4.5 Conclusion

With an analytical model and agent-based simulations we have explored the possibilities for a small community to overcome the Tragedy of the Commons. Our model formalizes theories from new institutional economics, in which moral values are more persistent than economic decisions (North, 2005; North, 1991; Williamson, 2000) and investment strategies and moral values evolve endogenously from a continuum of strategies. We neither use group selection as a mechanism, nor do we assume that individuals understand the nature of the dilemma. Surprisingly, we find that individuals do not need to know about socially desirable behavior as long as decisions are embedded in a social context and a simple moral rule is followed: taking own behavior as a benchmark and punishing and rewarding individuals who deviate from that benchmark in a symmetric way dictated by a moral preference. This may hint at the evolutionary origins of many moral codes that are based upon own behavior as a moral yardstick. Using Adam Smith's words (1759, section 1.3.9):

> *When we judge in this manner of any affection, as proportioned or disproportioned to the cause which excites it, it is scarce possible that we should make use of any other rule or canon but the correspondent affection in ourselves.*

Additionally, we find that social sanctions will be much less efficient when not own, but group behavior is used as a moral yardstick. Furthermore, the evolving enforcement mechanism compensates for obstacles such as large group size or high profits that can be made by acting against the group interest. Interestingly, the enforcement mechanism compensates only up to a critical for these obstacles. If disciplining peers is costly and individuals make small mistakes in their decision to sanction or reward, the outcome is inefficient, but still close to the social optimum. However, if sanctioning is very costly, individuals make large mistakes, or individuals cannot monitor each other sufficiently, the sanctioning system cannot be maintained. If one of these key parameters passes a critical threshold, the enforcement mechanism collapses. This sheds interesting light on the limits of community governance.

Appendix 4: Supplementary description of agent based simulations

The agent based simulations were performed in MATLAB. We use the protocol suggested by Grimm et al. (2006) to describe the setup of this model.

Purpose

The purpose of this model is to understand under which conditions agents that harvest a common pool resource (CPR) can overcome the so-called Tragedy of the Commons. In particular, we analyze the evolution of an enforcement mechanism, based on punishments and rewards.

State variables and scales

The community comprises a fixed number of individuals that harvest a CPR and punish or reward each other's behavior, depending on individual effort levels. Punishments and rewards will both be referred to as social sanctions. The model operates at three different time scales: the resource regeneration scale, the economic decisions and outcomes scale (effort allocation and implementing sanctions), and the institutional scale (the preference to sanction peers). The resource loop is nested within the economic loop, which is nested within the institutional loop. The dynamics of the resource stock depend on the aggregated effort level of all agents and will adjust in the resource time scale. Each agent pursues a certain exploitation strategy that is revised during the economic time scale. Additionally, each agent holds a certain sanctioning preference that may be revised during the institutional time scale.

Process overview and scheduling

The institutional module proceeds in discrete time steps ($t_{\text{institution}}$). Within each time step, the model processes first the economic module and then the cultural imitation module. The economic module starts with the resource module, and the payoff module. Then, the decision-making module is repeated t_{economic} times. The final values for the resource stock, the effort levels, and the sanctioning preference values enter the institutional module, and consequently, the cultural imitation module.

Design concepts

Emergence: The resource stock, the effort levels and the sanctioning preference coevolve endogenously. In the basic case, we impose the following moral constraints: Punishment and rewards are used in a symmetric way, and each individual uses his own effort level as a demarcation line that separates "good" behavior that will be rewarded from "bad" behavior that will be punished. All of those assumptions can and will be relaxed. We do not impose any limitation on the nature of the sanctioning preference. Therefore, we also allow for no sanctioning at all, or sanctions that are anti-social (Herrmann et al., 2008), where individuals with low effort levels are punished and individuals with high effort levels are rewarded.

Adaptation: Individuals can modify their effort level and sanctioning preference.

Fitness: Concerning economic decisions, individuals compare two different strategies and choose the one that yields a higher utility. Concerning the sanctioning preference, each individual compares himself with a randomly drawn partner. If the partner has a higher utility, the focal (i.e. assigned) individual adopts the partner's sanctioning preference.

Prediction: Individuals are modeled to be myopic and do not anticipate the consequences of their social sanctions. This rules out the possibility that individuals are willing to accept a lower payoff now, hoping that this investment will pay off in the future.

Sensing: Individuals can observe each other's effort levels, while they do not know about the resource stock. Agents do not observe each other's sanctioning preferences directly when deciding on effort. Only at the institutional time scale, agents compare each other's sanctioning preferences and imitate each other. The assumptions concerning cognitive requirements of the agents are therefore not very demanding. Each agent only chooses the best out of two effort levels and one out of two sanctioning preferences.

Interaction: The success of a harvesting strategy depends implicitly and explicitly on the institutions in place. Implicitly because harsh sanctioning will induce other individuals to reduce their effort level; explicitly, because social sanctions affect payoffs directly of the sender and of the receiver.

Stochasticity: Individuals make implementation and perception errors when observing and sanctioning each other. Additionally, monitoring is imperfect and there is a chance that harvesting activity goes undetected.

Initialization

Different initialization routines have been tried, giving rise to the same emerging patterns. In the simulations presented in the manuscript, the initial resource stock X_0 is randomly drawn from a normal distribution with a mean of 0.5 and a standard deviation of 0.1. The initial effort level $e_{0,i}$ is different for each agent and randomly drawn from the same distribution as the resource stock, i.e. with a mean 0.5 and a standard deviation of 0.1. The initial sanctioning preference $\tau_{0,i}$ is different for each agent and drawn from a normal distribution with a mean of zero and a standard deviation of 0.1.

Submodels

Resource growth:

Two cases will be distinguished. First, it is assumed that the resource is always in equilibrium, given the exploitation of all agents. Second, it is assumed that the resource regeneration depends on exploitation, but it is not necessarily in equilibrium.

If the resource and the economic time scales are perfectly separated, the size of the resource is given by $X_t = 1 - \gamma \sum_{i=1}^{n} e_{it}$, (4.A1)

where X_t is the equilibrium resource stock level at time t, γ is the catchability coefficient, and e_{it} is effort of agent i in time step t. We assume that the resource cannot go extinct. If $\sum_{i=1}^{n} e_{it} > 1/\gamma$, the resource stock is given by $X_t = 10^{-6}$. If the assumption that the resource is always in equilibrium is relaxed, the resource grows continuously between time $s = 0$ and $s = \xi$. The new value is defined as $X_t = X(\xi)$. The rate of resource change is given by the following differential equation

$$\frac{dX}{ds} = X(\zeta)\left(1 - X(\zeta)\right) - \gamma X(\zeta) \sum_{i=1}^{n} e_i(\zeta) , \qquad (4.A2)$$

with initial condition $X(0) = X_{t-1}$ if $t > 2$ and $X(0) = X_0$ otherwise, as given in the initialization subsection. Matlab's ODE45 solver is used to calculate the new resource level.

Payoff module

The profits from harvesting M_{it} are determined for each agent and given by

$$M_{it} = (p\gamma X_t - c) e_{it} , \qquad (4.A3)$$

where p is the benefit obtained from harvesting one unit of the resource, while c is the cost of one unit of effort. Additionally, all social sanctions are calculated based on bilateral interactions. There is a chance that individuals make an error in perceiving the correct effort level of each other, or make an error in implementing the social sanctions, reflected in the error terms for perception errors ε_{ijt} and implementation errors v_{ijt}, that are randomly drawn from a normal distribution with a mean of zero and standard deviation of σ_ε and σ_v. Consequently, agent i receives the following payoff from social sanctions S from agent j:

$$S_{ijt} = \begin{cases} \kappa\left(\tau_{jt} + v_{ijt}\right)\left(e_{jt} - e_{it} + \varepsilon_{ijt}\right) & \text{if } e_i > e_j, \\ \left(\tau_{jt} + v_{ijt}\right)\left(e_{jt} - e_{it} + \varepsilon_{ijt}\right) & \text{if } e_i \le e_j, \end{cases} \tag{4.A4}$$

where the parameter κ scales the relative strength of punishments compared to rewards. If $\kappa > 1$ ($\kappa < 1$) punishments have a stronger (weaker) effect than rewards. The costs of social sanctions C_{jit} agent i imposes on agent j is given by

$$C_{jit} = \begin{cases} \psi S_{ijt}^{\ 2} & \text{if } e_i > e_j, \\ S_{ijt}^{\ 2} & \text{if } e_i \le e_j, \end{cases} \tag{4.A5}$$

where the parameter ψ scales the relative costs of punishments compared to rewards. If $\psi > 1$ $(\psi > 1)$, sending punishments is more (less) costly than rewards.[17] We assume that these utility components will enter an individual's utility function that satisfies Stone-Geary Cobb Douglas preferences and can be given as

$$U_{it} = \left(\overline{M} + M_{it}\right)^{\alpha} \left(\overline{S} + \frac{1}{n-1}\sum_{j \ne i} S_{ijt} - \frac{w}{n-1}\sum_{j \ne i} C_{ijt}\right)^{1-\alpha}, \tag{4.A6}$$

where $\overline{M}$ and $\overline{S}$ respectively are exogenous profits and social status obtained from other activities than harvesting, and w and α are weighting parameters.[18]

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[17] We have also tried the cost function $C_{jit} = \begin{cases} \psi \left|S_{ijt}\right| & \text{if } e_i > e_j \\ \left|S_{ijt}\right| & \text{if } e_i \le e_j \end{cases}$, which gave rise to similar patterns.

[18] Alternatively, we have tried the simpler weighted utility function $U_{it} = w_1 M_{it} + \frac{w_2}{n-1}\sum_{j \ne i}^{n} S_{ijt} - \frac{w_3}{n-1}\sum_{j \ne i} C_{ijt}$,where $w_1, w_{2,} w_3$ are preference weights in the utility function. This gave rise to similar emerging patterns.

Decision-making module

Each individual is assigned once in random order. Updating is asynchronous, i.e. after each individual decision the state variables are changed. First, the profits for the focal (i.e. assigned) individual are calculated, as given in (4.A3). Then, the corresponding social sanctions are determined, as given by (4.A4) and (4.A5). With probability ω harvesting activity goes undetected. Then the corresponding social sanctions S_{ijt}, S_{jit}, and their costs C_{jit}, C_{ijt} are set to zero. Finally, the corresponding utility is calculated as given in (4.A6). This obtained utility is used as a benchmark for the focal individual for evaluating a slightly different strategy. The new effort level is given by $e_{i,t} = \tilde{e}_i + \chi_{e,i,t}$, where $\tilde{e}_i$ is the benchmark effort level and $\chi_{e,i,t}$ is a randomly normally distributed number with mean zero and standard deviation σ_e. If the new effort level would be smaller than zero, the effort level will be given by $e_{i,t} = 0$. Afterwards the corresponding resource stock is calculated and the corresponding utility is calculated. If the new utility is higher than the benchmark level, the individual adopts $e_{i,t}$ as a new benchmark strategy. The corresponding utility level and the new resource stock level are stored. If the new utility is lower, the individual will reject it and the old effort $\tilde{e}_i$ remains the benchmark strategy that the individual will keep choosing.

Cultural imitation module

Each agent compares his current utility with that of a randomly drawn partner. If the partner has a higher utility, the focal individual adopts the partner's sanctioning preference. If this is not the case, the focal individual keeps his own sanctioning preference. Since moral belief systems cannot be copied perfectly or change due to experiences unrelated to resource harvesting, we assume that there is a probability that a sanctioning preference changes in a stochastic way. With probability φ, the new sanctioning preference is given by

$$\tau_{i,t+1} = \tau_{i,t} + \chi_{\tau,i,t}, \qquad (4.A7)$$

where $\chi_{\tau,i,t}$ is a randomly normally distributed number with mean zero and standard deviation σ_τ. This new sanctioning preference will be stored.

Table 4.A1. Variables and parameters of the agent-based model.

Parameter	Description	Value
τ	Sanctioning preference	endogenous
e	Exploitation	endogenous
X	Resource stock	endogenous
M	Income	endogenous
S	Payoff from sanctions	endogenous
C	Costs of sanctioning	endogenous
U	Total utility	endogenous
χ_e	Deviation from benchmark exploitation	endogenous
χ_τ	Deviation from benchmark sanctioning trait	endogenous
ε	Implementation error	endogenous
υ	Perception error	endogenous
n	Number of agents	20
p	Market price	21
c	Cost per unit of effort	0.05
γ	Catchability coefficient	0.05
α	Preference for profits	0.7
w	Costs of sanctions (Stone-Geary)	1
ψ	relative cost of punishments vs. rewards	1
κ	relative utility effect punishments vs. rewards	1
$w1$	Utility weight income (additive utility)	–
$w2$	Utility weight sanctions (additive utility)	–
$w3$	Utility weight costs of sanctions (additive utility)	–
ω	Probability of undetected harvesting	0
φ	Probability of mutation sanctioning trait	0.02
$\overline{M}$	Exogenous income	1
$\overline{S}$	Exogenous sanctions	1
σ_e	Standard deviation mutation exploitation	0.02
σ_τ	Standard deviation mutation sanctioning preference	0.03
σ_ε	Standard deviation implimentation error	0.0002
σ_v	Standard deviation perception error	0.0002
$t_{\text{institution}}$	Time steps institutional time scale	4000
t_{economic}	Time steps economic time scale	20
ξ	Resource speed	in equilibrium

5

Towards the optimal management of the Northeast Arctic cod fishery

Abstract

The objectives pursued by governments managing fisheries may include harvesting the fish stocks to maximize profits, to minimize the impact of harvesting on the marine ecosystem, or to secure jobs in the fishing industry. These objectives all require adjusting the composition of the fishing fleet as the various vessel types differ with respect to their operating costs, their environmental impacts, etc. In this chapter we develop a management plan that allows the regulator to determine what fleet structure maximizes her objectives, what total allowable catch to harvest per year, and what long-run level of remaining biomass to target. In addition, the management plan also allows the regulator to compare the long-run welfare levels between the various management options to determine the costs and benefits associated with each. We apply the model to the case of Northeast Arctic cod, and econometrically estimate not only the cost and harvesting functions of the various vessel types, but also the parameters of the biological model as well as those of the demand function for cod. Our study thus provides key insights regarding the optimal management of this valuable fish species.

This chapter is a slightly modified version of: Richter, A.P., A.M. Eikeset, D.P. van Soest, and N.C. Stenseth, 2011. *Towards the Optimal Management of the Northeast Arctic cod fishery*, Manuscript. We are grateful to Per Sandberg from the Directorate of Fisheries (Bergen, Norway) for providing the data on individual vessels and very helpful advice and discussions concerning this data. We would also like to thank Pia Rønnevik from Statistics Norway for helpful advice and discussions on price and export data and Johan Grasman for valuable comments on an earlier version of this manuscript.

5.1 Introduction

The world's largest stock of Atlantic cod *(Gadus morhua)* is located in the Northeast Arctic, along the coast of Norway and in the Barents Sea. The history of the Northeast Arctic (or NEA) cod fishery since World War II is one of continued increases in landings, suddenly giving way to a near collapse of the fishery in 1989. In response to the cod crisis, a management regime was introduced that imposed a fishing quota on each ocean-going vessel in the industry. The question is how to allocate those quotas most efficiently and – even more important – how to set a total allowable catch (TAC) to prevent another cod crisis while meeting broader management objectives.

Fisheries management in Norway has a long tradition and regulations and management objectives have been considerably changed over time (Årland and Bjørndal, 2002; Hannesson, 2004; Hersoug, 2005; Holm, 1995; Nakken et al., 1996). Årland and Bjørndal (2002) have identified the main objectives of Norwegian fisheries regulations as (i) increasing the profitability of the fisheries sector, (ii) protecting the resource base, and (iii) securing employment opportunities in coastal communities to maintain the settlements along the coast. These different objectives give most resource economists a little headache, as optimizing them requires equating the marginal costs and benefits – and such data is usually not available. One could still give policy recommendation by constructing a social welfare function that contains preferences for (proxies of) these different objectives; see Dankel et al. (2009) for such a model.

In this chapter we suggest an alternative by treating these different objectives as constraints on fleet activity in the optimization problem of a resource manager. First, we analyze the scenario that the policy maker intends to maximize simply the rents from harvesting cod – harvesting should take place at lowest costs. Second, we consider the case that a policy maker maximizes rents on the condition that only boats are used that have least impact on the ecosystem. Third, we take into account that a diverse fleet is preferred (for the sake of regional development and cultural diversity). For all of these objectives, we determine the optimal TAC as well as the most efficient allocation of individual catch quotas over the various types of fishing vessels for various management objectives including maximization of the rents of cod harvesting.

Thus, this paper offers a management plan that can be adapted to a variety of government objectives. In addition, it allows the decision maker to estimate the costs of pursuing objectives other than cost minimization by comparing maximum financial welfare associated with the various objectives to that obtained under cost minimization.

Although the stock of NEA cod lies within the exclusive economic zones of both Russia and Norway, we focus our analysis on the Norwegian fleet of ocean-going vessels because it consists of a wide variety of boat types including trawlers, factory trawlers and longliners (Sandberg, 2006; Standal, 2008). Hence, determining the optimal fleet composition for the various management objectives is complex. We develop an analytical model to derive the optimal levels of biomass and the associated TAC, and estimate all parameters of the model using data on the NEA cod fishery. More specifically, we estimate the cost and production functions of the various vessel types in the industry (trawlers, longliners and factory trawlers), the demand function for cod (to determine how its value changes with quantity supplied), as well as the parameters of the growth function of cod.

This study has several unique features. First of all, our model differs from the previous studies in its econometric rigor, as it takes into account the many estimation problems associated with estimating the harvesting, cost, growth and demand functions including serial correlation and endogeneity. In this respect we improve on the earlier work by Arnason et al. (2004) and Kugarajh et al. (2006) in estimating the demand function for cod, and by explicitly acknowledging that there are not only variable costs associated with harvesting cod, but that there are fixed adjustment costs too (Asche, 2009). Second, we combine the empirically estimated functions into a model which allows policy makers to infer (i) the steady state levels of biomass that maximize their objectives (either unconstrained rent maximization, or rent maximizations taking into account environmental and/or social constraints), (ii) the associated optimal total allowable catch and the allocating thereof over the various vessel types, and (iii) the optimal harvest control rule (HCR) that informs the decision maker about the optimal TAC and its allocation over the boat types for every level of biomass – independent of whether it is the optimal steady state stock, or not. Third, our study also provides a flexible framework to include constraints regarding the supply side of fleet composition – the fact that a cost-minimizing long-run strategy cannot be implemented instantaneously, as boats that operate at lower costs cannot replace more costly ones in the short-run. As a result, our model provides an important bridge between analytical fisheries models that have little empirical content, and highly detailed econometric studies that do not deliver any direct policy advice.

While an optimal allocation between the coastal and the ocean-going fleet has received some attention in the literature (Armstrong, 1999, 2000; Armstrong and Sumaila, 2001), the size of an optimal individual quota per boat is usually not addressed. This is somewhat surprising, given that the question how to allocate a TAC over a certain number of boats is one of the most obvious management problems a fishery faces. An exception is Asche et al. (2009) who have addressed this question for the Norwegian trawler fleet. In most bioeconomic models individual boats do not exist – often costs are estimated at the aggregated level and hence, the fleet can only be analyzed as one entity; see Bromley (2009). This is an obvious shortcoming, as increasing and decreasing returns to scale operate at the boat level – not at the industry level. It is sometimes argued that a policy maker does not need to worry about how to distribute harvesting rights because a market for individually tradable quotas will ensure the efficient allocation (Grafton et al., 2006; Hannesson, 2004). We would like to note that this is not true for two reasons. First, the total quota size to be allocated (via grandfathering, or via auctions) crucially depends not just on the benefits of selling cod (in terms of revenues or welfare obtained), but also on the costs of harvesting it. While the benefits only depend on the quantity supplied to the market (i.e., on the TAC), the costs critically depend on the composition of the fishing fleet as some boat types are more efficient in catching cod than others. Hence, while a system of ITQs may ensure that actual harvesting takes place at minimum cost, we still need to know how the minimum cost solution looks like in order to decide on the level of the TAC itself. Second, even if ITQs result in fishing activity that operates at least costs, such an outcome would only be socially optimal if society had no other objectives than just minimizing harvesting costs. In reality, broader objectives, such as ecosystem preservation, the cultural value of a diverse fleet, or equity considerations, are pursued; see also chapter 7. Therefore, detailed information on the various vessel types is needed to be able to determine whether or not certain boat types should be prohibited from purchasing ITQs – in case the government pursues objectives other than just pure cost minimization.

Management of the NEA cod fishery is inherently complex, and any useful model – as the one developed here – has inevitably to sacrifice certain details. First of all, this study ignores important ecosystem effects. In Winter, the mature fish migrate out of the Barents Sea for about 3 months to spawn, returning to the feeding grounds in Spring. The cod eggs drift up along the Norwegian coast and the immature fish stay in the feeding grounds until maturation when they start reproducing. Obviously management could be substantially improved by acknowledging the age-structure and the productivity of the stock (Diekert et al., 2010a; Sumaila, 1997a). Second, and in a similar vein, the fact that older cod tend to cannibalize on younger cod may have management implications that are ignored here (Armstrong, 2000; Armstrong and Sumaila, 2001). Third, if harvesting pressure is very high this may induce an evolutionary response that lead to economic repercussions (Eikeset et al., 2010e); see also Chapter 6. Fourth, food-web interactions with other species are important factors driving the cod stock dynamics. For example capelin (*Mallotus villosus*) and Norwegian Spring-Spawning (NSS) herring (*Clupea harengus*) are two of the most important fish species the cod interacts with (Hjermann et al., 2007). Herring feeds on capelin larvae (Gjøsæter and Bogstad, 1998) and is therefore competing with the cod for the prey species capelin. We ignore this effect in this paper, but see Link and Tol (2006) and Sumaila (1997b). Fifth, climate plays also an important role in this ecosystem. If new species immigrate from the south, this leads to a new food-web structure (Ottersen et al., 2006). Examples of bio-economic models that have analyzed how climate may affect the management of cod are Hannesson (2007b) and Link and Tol (2009). A study that takes both climate change and multiple species into account is Eide and Heen (2002). Sixth, climate change may also affect the negotiations and the legitimacy of the Joint Norwegian-Russian Fishery Commission. If the climate gets warmer, this may trigger capelin to migrate further into Russian waters in which the cod may follow (Roderfeld et al., 2008). Our analysis does not touch upon such strategic interactions, as we assume that the management authority in place sets and enforces the quota; for examples of strategic games regarding the NEA cod fishery, see Diekert et al. (2010c), Hannesson (2007a), and Sumaila (1997a; 1997b).

This chapter is organized as follows. In section 5.2 we present an overview of the NEA cod fishery. Section 5.3 develops the optimal management plans for a variety of management objectives. We estimate the model in section 5.4, presenting the parameterizations of the production, cost, and demand functions as well as of the biological model. Next, section 5.5 combines the theoretical and empirical results and derives an optimal policy, while section 5.6 concludes.

5.2 The Northeast Arctic cod Fishery

The Northeast Arctic cod fishery consists of two parts that are geographically separate: the feeding grounds in the Barents Sea, and the spawning grounds further south along the coast of Norway. Norwegians have been fishing for over thousands of years in predominantly the spawning grounds because of their proximity to villages and ports. Since the 1930s (and especially after the second world war), technological developments facilitated the use of large ocean-going trawlers in the feeding grounds in the Barents Sea, which resulted in an increase in fishing pressure (Godø, 2003). Until the early 1970s the number of trawlers steadily increased and landings have been as high as one million tonnes per annum – for some years, the harvesting probability for individual fish was as high as 70% per year (Eikeset, 2010).

In the late 1970s it became clear that the NEA cod fishery was overexploited; see Figure 5.1. In 1977 the Norwegian government responded by starting to actively enforce the country's exclusive economic zone and by barring the entry of new trawlers (Standal and Aarset, 2008).

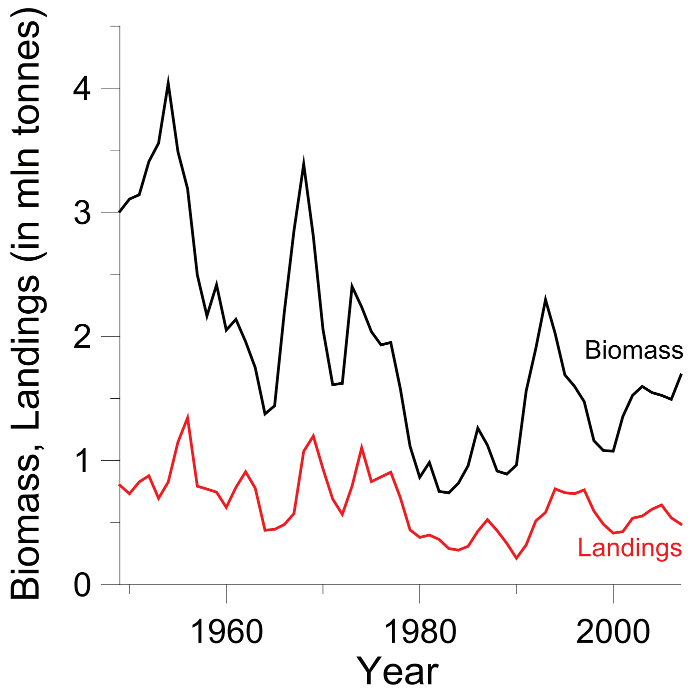

Figure 5.1. The total landings and corresponding total biomass of Northeast Arctic cod since 1949.

Also, a cap was introduced on the total amount of cod caught per year (the so-called total allowable catch, or TAC). Unfortunately, these measures were insufficient to prevent the cod crisis that occurred in 1989: the TACs in the 1980s were too lenient – especially because the cod was under severe stress already because of the population collapse of one of its main prey species, capelin. As a consequence the TAC in 1989 had to be reduced dramatically with disastrous consequences for the cod fishing industry (Hersoug et al., 2000). There was a broad political consensus that a policy change was needed. A system was introduced that gave each ocean-going vessel in the industry a quota to catch a certain amount of cod. These quotas were non-transferable at first, but later on the regulations were revised to allow vessels to transfer harvesting rights (Hersoug et al., 2000; Holm and Nielsen, 2007; Standal and Aarset, 2008). Currently, the fishery is managed by the Joint Norwegian-Russian Fishery Commission as the feeding grounds of NEA cod (in the Barents Sea) are located in the exclusive economic zones of both countries. Since 2004 a new management plan is in use that determines the annual TAC; more than 90% of the quota is allocated to Russia and Norway, while the European Union member states receive the bulk of the remaining 10%. The Norwegian TAC is divided between these ocean-going vessels and the smaller coastal vessels with a distribution key known as the trawl ladder (Armstrong and Sumaila, 2001; Hannesson, 2004, p. 106; Standal and Aarset, 2008). The TAC is adjusted annually on the basis of a so-called harvest control rule (HCR) which determines the TAC for any level of spawning stock biomass; see Eikeset et al. (2010c) and the next chapter (panel c in Figure 6.1).

5.3 The optimal management of the cod fishery

5.3.1 Deriving an optimal TAC

We assume the government aims to maximize the net present value of social welfare as the sole-owner of the resource. Because about 90% of the cod is exported, welfare maximization can best be described by maximizing profits – but see footnotes 26–27 and chapter 6. However, we acknowledge that society may have broader objectives than just maximizing rents from harvesting cod. These other considerations may be related to environmental concerns (as some boat types are more damaging to the marine ecosystem than others) or social-cultural concerns (the desire to maintain a diverse fleet because of cultural considerations). Therefore, we assume that the government aims to maximize Norway's social welfare (by maximizing the rents of cod harvesting) while it may or may not decide to impose constraints on the type of vessels used to address these considerations too. We derive the optimal management plan for three different management objectives. First, we solve the problem assuming that society chooses to use a fleet that is able to harvest a specific amount of cod, the TAC, at least total costs. Second, we consider the case in which society imposes additional constraints on the fleet composition in order to protect the marine environment by banning trawlers and factory trawlers, since they are deemed more destructive to the ecosystem than longliners (Dayton et al., 1995). The third case we consider is the one where the government, motivated by employment or cultural considerations, decides to maintain a diverse fleet by allocating harvesting rights to a variety of vessel types in the industry – as is currently done in the Norwegian cod fishery. Throughout the chapter, we follow Salvanes and Squires (1995) by assuming that all boats of a specific type are identical.

The instantaneous flow of profits (or producer surplus) is specified as follows

$$\Pi(X_t, TAC_t) = R(TAC_t) - C(X_t, TAC_t), \qquad (5.1)$$

where X_t is the biomass of cod present in the Northeast Arctic in year t, and TAC_t is the total allowable catch set by the government. Furthermore, $R(TAC_t)$ are the revenues of supplying TAC_t to the market, and $C(TAC_t, X_t)$ are the costs of catching TAC_t. The cost function is assumed to be a function not only of the quantity harvested, but also of the amount of cod biomass remaining. The reason is that the returns per unit of effort (for example, the number of days spent catching cod) may depend on the density of the fish in the sea (the so-called stock effect). Also note that the costs of catching fish are obviously also dependent on the types of vessels used – in other words, they depend on the fleet composition. In section 5.4, the empirical estimations show that $C_X < 0$ and $C_{TAC} > 0$.[19] Regarding the revenue function, we assume a linear inverse demand function for cod:

$$P_t = a - bTAC_t, \qquad (5.2)$$

so that $R(TAC_t) = \left(a - bTAC_t\right)TAC_t$. The optimal control problem the government faces is as follows:

$$W = \max_{TAC} \int_0^\infty e^{-\delta t}\Pi(X_t, TAC_t)dt \qquad (5.3)$$

subject to

$$\dot{X}_t = G(X_t) - TAC_t, \qquad (5.4)$$

where dots denote time derivatives, δ is the discount rate, and $G(X)$ is the growth function of the cod stock, which is assumed to be logistic :

$$G(X_t) = rX_t\left(1 - \frac{X_t}{K}\right), \qquad (5.5)$$

where r is the intrinsic growth rate, and K is the maximum amount of cod biomass that would materialize in the long run absent harvesting – the so-called carrying capacity. The current-value Hamiltonian $\mathscr{H}$ is then given by[20]

$$\mathscr{H} = (a - bTAC)TAC - C(TAC, X) + \varphi\left[G(X) - TAC\right], \qquad (5.6)$$

where φ is the co-state variable.

[19] Partial derivatives are denoted by subscripts. Thus $\frac{\partial C}{\partial X} = C_X$

[20] In the rest of the chapter we omit time subscripts unless doing so could cause confusion.

Using the dynamics of the resource stock (5.4) and applying the maximum principle, we obtain the following first-order conditions for an optimum:

$$\mathscr{H}_{TAC} = 0 \quad \Rightarrow \quad \varphi = a - 2bTAC - C_{TAC}, \tag{5.7}$$

$$\delta\varphi = \dot{\varphi} - C_X + \varphi \ G_X. \tag{5.8}$$

To derive the steady state optimum, we set $\dot{\varphi} = \dot{x} = 0$. Substituting (5.7) into (5.8) we have $C_X = (G_X - \delta)\left(a - 2bTAC - C_{TAC}\right)$. The optimal steady state relationship between TAC and biomass is thus as follows:

$$TAC = \frac{\left(2rX + K\left(\delta - r\right)\right)\left(a - C_{TAC}\right)}{2b\left(2rX + K\left(\delta - r\right)\right) + KC_X}. \tag{5.9}$$

As shown in (5.9), the optimal TAC depends on the cost function, and hence on the composition of the fleet – as not all vessel types are likely to be equally efficient in catching cod. While C_X (the stock effect) is transmitted through all vessels that are in operation, C_{TAC} (the cost of catching an additional amount of cod) is only transmitted through the marginal vessel type – the type that is the last to receive a quota if these quotas are handed out starting with the most preferred type (see also section 5.3.4). This difference is important if more than one vessel type is in use for catching cod. If only one boat type is operated in the cod fishery, deriving C_{TAC} and C_X is straightforward. This may be the case because one vessel type outperforms all other types in a specific aspect – one type may be able to harvest at lower costs than the others, or one type may have smaller environmental impacts than any of the other types. If, however, more than one type is used in the fishery (because the government values a diverse fleet because of social or cultural considerations, or because it faces constraints regarding the number of vessels of the preferred type), the stock effect C_X shows the impact of having an extra unit of biomass on *all* vessels (of all types used) in the fishery, while C_{TAC} only pertains to the marginal boat type as defined above. Furthermore, because the optimal TAC depends on the composition of the fishing fleet, so does the optimal steady state biomass, which can be determined by substituting $TAC = rX(1 - X / K)$ into (5.9) and solving. Having determined the optimal stock and harvesting levels, we can also calculate social welfare (as measured by producer surplus).

While steady state biomass and harvest levels are interesting in itself, they are often not very useful for management purposes, as in reality the stock will never be in steady state. Therefore, we derive a harvest control rule (HCR) that informs the decision maker about the optimal

TAC for any given stock level. In the absence of discounting, a feedback HCR can be determined relatively easily following Sandal and Steinshamn (1997; 2001); see for applications Arnason et al. (2004) or Grafton et al. (2000). From (5.7) it follows that the co-state variable φ can be rewritten as a function of the state and the control variable (X and TAC respectively). If the discount rate is zero, the Hamiltonian is constant over time, and maximizing the current-value Hamiltonian boils down to maximizing the profit flow (as defined in (5.1)) that can be obtained in a steady state, also referred to as the sustainable economic rent $\Pi^*(X)$ (Sandal and Steinshamn, 2001). We thus obtain the following analytical feedback rule:

$$TAC = \frac{2brX\left(1-\dfrac{X}{K}\right) \pm \sqrt{\left(-2brX\left(1-\dfrac{X}{K}\right)\right)^2 - 4b\left(\left(a-C_{TAC}\right)rX\left(1-\dfrac{X}{K}\right)-\Pi^*(X)\right)}}{2b}; \qquad (5.10)$$

see the Appendix for the exact derivation.

5.3.2 Deriving the optimal quota allocation for each individual vessel type

Suppose that currently there are Z different types of vessels used in the industry, such as trawlers, factory trawlers, and longliners. Furthermore, in this subsection we also assume that boats can freely enter or leave the cod fishery – an assumption that will be relaxed in section 5.3.4. The total allowable catch of the vessels of type z, $z \in [1,..,Z]$, in year t is denoted by TAC_{zt}, and the sum of these type-specific allowable catches should add up to the TAC as determined by the government for that year (that is, $TAC_t = \sum_{z=1}^{Z} TAC_{zt}$). The production process of a vessel of type z is described by a Cobb Douglas harvest function, where the amount of cod harvested in year t (h_{zt}) is a function of both that vessel's effective fishing effort (e_{zt}, as measured in efficiency units – given by the number of days catching cod multiplied by the vessel's Bruto Register Tonnage (BRT)), and of the total amount of cod biomass in the Northeast Arctic (X_t):

$$h_{zt} = q_z X_t^{\alpha_z} e_{zt}^{\beta_z}, \qquad (5.11)$$

where q_z is a catchability coefficient, α_z is the stock-output elasticity and β_z is the effort-output elasticity. All parameters are type-specific, and α_z and β_z reflect the percentage increase in harvests resulting from a one percent increase in the relevant input. In section 5.4 we show that for all boat types, $1 > \beta_z > \alpha_z > 0$.

Regarding the costs of catching cod, we distinguish between fixed adjustment costs and variable costs. Adjustment costs include changing the vessel's gear to make it suitable for catching cod, the fuel spent on sailing to the cod fishing grounds, etc., while the variable costs are the costs incurred on the days that the vessel is actually catching cod. We use f_z to denote the fixed adjustment cost components while the variable costs of effort are assumed to be constant and equal to v_z. Hence, the annual costs incurred by a vessel of type z spending e_{zt} tonnage-days catching cod in a year are given by

$$c_{zt} = f_z + v_z e_{zt} . \tag{5.12}$$

Let us first determine, for each vessel type z, the optimal effort level per boat e_z^*, and also the optimal number of boats n_z^*, if the aggregate amount of cod to be caught by all boats of type z is equal to TAC_z (that is, $n_z q_z X^{\alpha_z} e_z^{\beta_z} = TAC_z$). The Lagrangian of the cost minimization problem is as follows:

$$\Phi_z = n_z\left(v_z e_z + f_z\right) + \lambda_z\left(TAC_z - n_z q_z X^{\alpha_z} e_z^{\beta_z}\right), \tag{5.13}$$

where λ_z is the shadow price of harvesting an extra unit of cod by increasing the fleet size or the size of the quota per boat. The first order conditions associated with (5.13) are

$$\frac{\partial \Phi_z}{\partial n_z} = v_z e_z + f_z - \lambda_z q_z X^{\alpha_z} e_z^{\beta_z} = 0, \tag{5.14a}$$

$$\frac{\partial \Phi_z}{\partial e_z} = n_z v_z - \beta_z \lambda_z n_z q_z X^{\alpha_z} e_z^{\beta_z - 1} = 0, \tag{5.14b}$$

$$\frac{\partial \Phi_z}{\partial \lambda_z} = TAC_z - n_z q_z X^{\alpha_z} e_z^{\beta_z} = 0. \tag{5.14c}$$

For future reference, it is convenient to note that (5.14a) implies that the shadow price is equal to the vessel's average costs of catching cod fish:

$$\lambda_z = \frac{v_z e_z + f_z}{q_z X^{\alpha_z} e_z^{\beta_z}}. \tag{5.15}$$

Next, combining (5.14a) and (5.14b) we find that the optimal amount of effort per vessel per year is equal to

$$e_z^* = \frac{f_z \beta_z}{v_z\left(1-\beta_z\right)}. \tag{5.16}$$

This efficient scale of operating a vessel of type z is the result of two competing effects associated with increasing the amount of cod harvested. If h_z is increased, the fixed costs of adjusting the gear to cod harvesting (f_z) are spread over a larger harvest, but increasing h_z also requires a more than proportional increase in effort (e_z) because of decreasing returns to scale; see (5.11).

Hence, the average costs of harvesting cod per vessel are a U-shaped function of effort, with its turning point at e_z^*. Also, note that the efficient scale of employing a boat of type z is constant and independent of biomass. Let us now proceed by calculating the costs per vessel operating at e_z^* tonnage-days of catching cod. Using (5.12) and (5.16) we have

$$c_z^* = f_z + v_z e_z^* = f_z + v_z\left(\frac{f_z\beta_z}{v_z\left(1-\beta_z\right)}\right) = \frac{f_z}{\left(1-\beta_z\right)}. \tag{5.17}$$

Next, substituting (5.16) into (5.14c), we find that the optimal number of boats of type z is equal to

$$n_z^*\left(TAC, X\right) = \left(\frac{v_z\left(1-\beta_z\right)}{f_z\beta_z}\right)^{\beta_z}\frac{TAC_z}{q_z X^{\alpha_z}}. \tag{5.18}$$

The larger the amount of biomass, the more productive is a boat of type z, and hence the fewer boats are needed to harvest a specific TAC. So, combining (5.17) and (5.18) we identify that the harvesting costs of all boats of type z operating at the efficient scale are equal to

$$C_z\left(TAC_z, X\right) = n_z^* c_z^* = \Omega_z TAC_z / X^{\alpha_z}, \tag{5.19}$$

where $\Omega_z \equiv q_z^{-1}(v_z/\beta_z)^{\beta_z}(f_z/(1-\beta_z))^{1-\beta_z}$. When operating at the optimal scale, the average costs of catching one tonne of cod fish are equal to

$$\lambda_z^* = \tilde{c}_z = \frac{\Omega_z}{X^{\alpha_z}}. \tag{5.20}$$

5.3.3 Determining the optimal allocation of vessel quotas

We analyze the case where the government (i) chooses to use a fleet that operates at lowest costs (potentially the result of a market mechanism like an ITQ system in which boats of all types are allowed to participate) or (ii) takes broader objectives into account and imposes fleet constraints.

First, we assume that the government aims to minimize the costs of catching a certain amount of cod, the TAC. From (5.19) it is clear that this would require allocating the entire TAC quota to the vessel type that, for the relevant level of biomass, has the lowest average harvesting costs, $\tilde{c}_z = \Omega_z / X^{\alpha_z}$; see (5.20). Let us use subscripts $z = LC1$ to denote the vessel type with the lowest average costs, $z = LC2$ to denote the vessel type with the one-but-lowest average costs, etc. More formally, $z = LC1$ is defined as the type for which we have $\tilde{c}_{LC1} = \Omega_{LC1} / X^{\alpha_{LC1}} \leq \Omega_z / X^{\alpha_z} = \tilde{c}_z \quad \forall \quad z = \{1,..,Z\}$, $z = LC2$ is defined as the type for which we have $\tilde{c}_{LC2} \leq \tilde{c}_z \quad \forall \quad z = \{1,..,Z\} \setminus \{LC1\}$, and so on.

Second, we assume that governments may pursue objectives other than just pure financial profit maximization. While pure cost minimization may dictate $TAC_z = 0$ for all $z \neq LC1$, considerations other than the concern for financial cost minimization may result in $TAC_z = \theta_z TAC > 0$ for at least some $z \neq LC1$ too. In our model the government chooses a specific vector of shares $(\sum_{z=1}^{Z} \theta_z = 1)$ that matches the specific objectives and then determines the optimal TAC within these quotas constraints $TAC_z = \theta_z TAC$. This approach allows the government to calculate the welfare costs associated with imposing an allocation of quotas other than the allocation that minimizes harvesting costs. The difference in producer surplus (our measure of welfare) indicates the costs to society for not using the cost-minimizing vector of shares so that these costs can subsequently be compared, explicitly or implicitly, to the environmental or social benefits obtained, to decide whether the benefits of these decisions exceed their costs. While allocation of quotas *between* several fleet types is not necessarily cost-minimizing (as it may be determined by other policy objectives than just maximizing financial welfare), the quota allocation *within* a fleet is still assumed to be optimal – and given by the number of tonnage-days boats spend chasing cod (e_z^*). For any given vector of shares θ_z and TAC, from (5.19) we have that the total harvesting costs are then equal to

$$C^*(TAC, X) = \sum_{z=1}^{Z} \left(\Omega_z \theta_z TAC / X^{\alpha_z} \right). \tag{5.21}$$

Note that (5.21) allows the government to calculate the (marginal) harvesting costs for all possible management objectives. In case it attempts to maximize fleet profits, the cost-minimizing allocation can be recovered from (5.21) when setting $\theta_{LC1} = 1$ and $\theta_z = 0 \;\; \forall \;\; z = \{1,..,Z\} \setminus \{LC1\}$. If environmental concerns play a role, (5.21) gives the associated cost function setting $\theta_z = 1$ for the boat type that is considered least harmful (and zero shares to all other vessel types). In short, the government can simply insert the vector of harvesting shares it deems optimal into (5.21) to obtain the associated cost function.

5.3.4 Optimal quota allocation in case of fleet lock-in

In the previous sub-section, we have shown that it is cost-minimizing to use only the vessel type that has the lowest average costs. In practice, one is typically confronted with a situation where boats of a specific type cannot easily replace vessels of a different type. Instead, the fleet composition can only be changed in the short run at substantial costs – a situation which we will refer to as a fleet lock-in. In this sub-section, we analyze how to allocate a TAC if there are maximally $\overline{n}_z$ vessels of type z available in the industry. We keep the assumption that boats can be employed in a different fishery, but the maximum number of vessels of type z is given by $\overline{n}_z$. In that case, (5.16) and (5.18) can only be implemented if $n_z^* \leq \overline{n}_z$. The cost-minimizing policy in case of lock-in can then be derived as follows. First, one needs to calculate the maximum amount of cod that can be caught by the vessel type with the lowest average costs $(z = LC1)$ when all $\overline{n}_z$ boats of that type are run at their efficient scale e^*_{LC1} (as given by (5.16)):

$$\overline{TAC}_{LC1}(X) = \overline{n}_{LC1} q_{LC1} X^{\alpha_{LC1}} \left(e^*_{LC1}\right)^{\beta_{LC1}}. \tag{5.22}$$

If $TAC \leq \overline{TAC}_{LC1}$ all harvests can be caught by the vessels with the lowest costs at optimum effort level e^*_{LC1}. In that case, it is optimal to have each individual boat being run at its efficient scale, and that means that the TAC should be divided equally over $n^*_{LC1} \leq \overline{n}_{LC1}$ vessels (see (5.18)), and none to any other boats. If, however, $TAC > \overline{TAC}_{LC1}$, it is cost-minimizing to increase the quotas of boat type $LC1$ because, by definition, these boats have lower average costs than boats of other types $(\tilde{c}_{LC1} < \tilde{c}_{LC2})$. Therefore, the quota of the $\overline{n}_{LC1}$ boats of type $LC1$ should be increased until a switching point e^{**}_{LC1}, which is implicitly defined by $\frac{v_{LC1} e^{**}_{LC1} + f_{LC1}}{q_{LC1} X^{\alpha_z} \left(e^{**}_{LC1}\right)^{\beta_{LC1}}} = \frac{\Omega_{LC2}}{X^{\alpha_{LC2}}}$; it is only profitable to use the first boat of type $LC2$ if imposing higher effort on all $\overline{n}_{LC1}$ boats of type $LC1$ results in average costs higher than the minimum average costs of type $LC2$. In practice, that point may never be reached because there is a maximum limit on effort that can be exercised per boat given by e^{MAX}_{LC1} (if only because a year has 365 days). Hence, the manager should not use boats of type $LC2$ if $TAC \leq \widehat{TAC}_{LC1}$ where $\widehat{TAC}_{LC1}$ is defined by the maximum harvesting level achieved by $\overline{n}_{LC1}$ boats of type $LC1$ running at a scale equal to $\widehat{e}_{LC1} = \min\left\{e^{MAX}_{LC1}, e^{**}_{LC1}\right\}$. If $\overline{TAC}_{LC1} \leq TAC \leq \widehat{TAC}_{LC1}$, only boats of type $LC1$ are used at a scale equal to $\widehat{\widehat{e}} = \left(\frac{TAC}{\overline{n}_{LC1} q_{LC1} X^{\alpha_{LC1}}}\right)^{\frac{1}{\beta_{LC1}}}$. If $TAC > \widehat{TAC}_{LC1}$, boats of fleet type $z = LC2$ will also be used. As long as $n^*_{LC2} \leq \bar{n}_{LC2}$, it is optimal to operate these vessels at their efficient scale (see (5.16)), and hence

$$n^*_{LC2} = \frac{TAC - \bar{n}_{LC1} q_{LC1}\left(X_{LC1}\right)^{\alpha_{LC1}}\left(\hat{e}_{LC1}\right)^{\beta_{LC1}}}{q_{LC2}\left(X_{LC2}\right)^{\alpha_{LC2}}\left(e^*_{LC2}\right)^{\beta_{LC2}}}.$$

If the TAC is even higher and all boats of the fleet type $z = LC2$ are in use, effort levels e_{LC1} and e_{LC2} should both be increased (if possible) until the average harvesting costs of boats of all three types $(LC1,\ LC2,\ LC3)$ are equal. We repeat these steps for all $z = LC3,\ldots,LCZ$. The total number of boats in the cod fishery are thus equal to

$$n\sum_{z=1}^{Z} n_z(e,TAC,X) = \begin{cases} n^*_{LC1}(e^*_{LC1}) & \text{if} \quad TAC \leq \overline{TAC}_{LC1} \\ \bar{n}_{LC1}(\hat{\bar{e}}_{LC1}) & \text{if} \quad \overline{TAC}_{LC1} \leq TAC \leq \widehat{TAC}_{LC1} \\ \bar{n}_{LC1}(\hat{e}_{LC1}) + n^*_{LC2}(\hat{e}_{LC1}, e^*_{LC2}) & \text{if} \quad \widehat{TAC}_{LC1} < TAC \leq \widehat{TAC}_{LC1} + \overline{TAC}_{LC2} \\ & \quad etc. \end{cases} \tag{5.23}$$

5.4 An empirical application of the Northeast Arctic cod fishery

In this section we estimate the production and cost function from individual vessel data from the Directorate of Fisheries (Bergen, Norway), and we do so in section 5.4.1. Next, since the NEA cod fishery is not a small scale fishery, its landings affect the price at which Norway exports cod – and hence also the landing prices. We empirically derive the percentage decrease in the landing price (or ex-vessel price) resulting from a one percent increase in the quantity of cod harvested – i.e., the inverse of the price elasticity of demand, sometimes also referred to as the price flexibility; see section 5.4.2. Finally, we estimate the biological growth model for the NEA cod stock in section 5.4.3. All data sources are described in the Appendix.

5.4.1 Estimating the production and the cost function

To estimate the cost and production functions of the various boat types, we use panel data from the period 1990-2000, which covers almost all ocean-going vessels that were active in the cod fishery in that time period. We have data on the quantity of cod harvested, days spent on catching cod and on the costs incurred per year for 107 trawlers, 25 factory trawlers, and 85 longliners. Most vessels have not reported for all of the 11 years, making the panel unbalanced; see Sandberg (2006) for more information on the data. Table 5.1 gives descriptive statistics of the data that will be used in the regression analysis.

In economics, the relationship between inputs and output (the production function) is often inferred by estimating a cost function $C_z(X, TAC)$ using a flexible form.

Table 5.1. Descriptive statistics of the annual data used to estimate the cost and production function. The data covers the period from 1990-2000, and is obtained from the Norwegian Directorate of Fisheries.

Trawlers	Days fishing cod 62°N	Days fishing in total	Days on sea	Cod harv. 62°N in tonnes	BRT	Total costs in NOK	Length in metres
Mean	81.2	252.6	283.2	782	280.3	12 mln	41.8
Median	79.0	260.0	294.5	768	298.0	12 mln	46.5
Maximum	278.0	374.0	364.0	2882	499.0	32 mln	53.8
Minimum	2.0	42.0	112.0	3	33.0	0.52 mln	17.6
Std. Dev.	45.3	67.0	49.5	526	89.7	4.8 mln	8.3
Factory Trawlers							
Mean	50.5	181.5	181.5	1398	776.9	34 mln	60.3
Median	47.0	182.0	182.0	1303	660.0	34 mln	56.9
Maximum	122.0	299.0	299.0	4495	1428.0	57.7 mln	75.5
Minimum	8.0	37.0	37.0	150	473.0	15 mln	48.7
Std. Dev.	26.4	45.9	40.7	829	307.9	7 mln	6.5
Longliners							
Mean	46.5	218.0	310.7	306	216.2	12 mln	36.8
Median	43.0	217.0	318.0	291	202.0	11 mln	36.7
Maximum	134.0	342.0	356.0	874	688.0	28 mln	51.2
Minimum	10.0	106.0	207.0	109	100.0	2.6 mln	28.0
Std. Dev.	20.7	44.7	27.0	150	82.1	4.5 mln	4.3

This approach is based on the assumption that whatever amount of fish a boat has caught, it has done so at minimum cost. In such a case duality applies (Varian, 1992, Chapter 6) and the production function can be inferred from the prices of the inputs and outputs. The biggest advantage of this approach is that one can estimate a cost function, even when one does not have data on fishing effort. Further advantages are that one does need to assume a certain technological structure a priori (as we did in (5.11)) and also that it is statistically more efficient because the cost function and the first-order condition for cost minimization can be estimated jointly. In fisheries economics, this approach is less appealing because of several reasons. First of all, the standard cost function approach cannot be applied because one of the inputs in the production process, biomass, cannot be chosen freely by individual fishermen, and introducing quasi-fixed factors in the cost function typically complicates the estimation procedure considerably (Morrison, 1988; Morrison and Schwartz, 1996; van Soest et al., 2006). Second, while it is necessary to make assumptions about the behavioral objectives of fishermen when designing a model, fishermen are unlikely to always operate at minimum costs at all times. Markets are usually incomplete, fishermen face informational constraints, and payments of all inputs are not always determined by market prices directly because crew members may receive shares of the harvesting revenues rather than a fixed wage (McConnell and Price, 2006; Sandberg, 2006). Finally, it is not necessarily the case that all fishermen always try to maximize profits because other considerations (including status seeking) may play a role too (Gezelius, 2007; Ginkel, 2009; Holland, 2008; Poos, 2010; Salas and Gaertner, 2004). Because of these reasons it is preferred to estimate the technical relations (5.11) and (5.12) separately rather than using a cost function that assumes fishermen choices to be optimal (i.e. cost-minimizing). This is in line with Felthoven and Morrison Paul (2004 p.162), who argue that "fishing technology [should] be analyzed directly (through a "primal" approach), by focusing on inputs and outputs, rather than by modeling choices based on costs, profit, or market prices." Therefore, we estimate the production function (5.11) and the input cost function (5.12) separately.

Estimating the production function of the three vessel types

First, we estimate the production function for each fleet type z given in (5.11). Table 5.1 shows that the ocean-going cod fleet consists of three types: trawlers (47.8% of the boats), factory trawlers (18.5%), and longliners (33.7%). We denote these three boats types by $z = T$, $z = FT$, and $z = LL$, respectively. In our model, effort e_{izt} is defined as the number of days a boat is fishing cod north of 62 degrees latitude, multiplied by its size (Bruto Register Tonnage). Including the size of the boat takes differences in operational intensity into account (Asche et al., 2009). We cannot rule out the presence of omitted variable bias – caused for example by differences in the skillfulness of individual skippers (Sandberg, 2006; Squires and Kirkley, 1999). This poses a particular problem if the "skipper effect" is positively correlated with the size of the boat, which may happen if the best skippers steer the largest boats. Therefore, we estimate the model by means of Ordinary Least Squares (OLS) with fixed effects on the cross-sections (that is, we use vessel-specific fixed effects). As our number of cross-sections is much larger than the number of years, we use a robust variance-covariance matrix that produces panel corrected standard errors (PCSE), as proposed by Beck and Katz (1995). We estimate (5.11) for each boat type separately, thus allowing all parameters to be vessel type specific:

$$\log(h_{izt}) = \log(q_z) + \alpha_z \log(X_t) + \beta_z \log(e_{izt}) + \upsilon_{iz} + \varepsilon_{izt}, \tag{5.24}$$

where ε_{izt} is an error term, while υ_{iz} is the estimate of the fixed effects. The latter sum up to zero when aggregating over all boats of type z, and hence υ_{iz} can be interpreted as individual deviations from the average catchability coefficient $\log(q_z)$. The regression results are presented in Table 5.2. The stock-output elasticity α_z is estimated to be 0.58 for trawlers, 0.38 for factory trawlers, and 0.22 for longliners. The effort-output elasticity β_z is estimated to be 0.85 for trawlers, 0.89 for factory trawlers, and 0.92 for longliners. These coefficients are similar to the ones found by Kronbak (2004).[21]

Table 5.2. Regression output for the production functions of the three vessel types. The standard errors are presented in parentheses.

	Trawlers	Factory Trawlers	Longliners
α: stock-output-elasticity	0.58 (0.08)	0.38 (0.11)	0.22 (0.09)
β: effort-output elasticity	0.85 (0.05)	0.89 (0.06)	0.92 (0.08)
log(q)	-7.39 (1.56)	-3.29 (2.31)	-0.54 (1.88)
Durbin Watson	1.72	1.55	1.52
Adjusted R^2	0.93	0.76	0.93
Total observations	348	157	226
Number of boats	84	22	64

Estimating the cost function of the three vessel types

We estimate the fixed and variable costs of harvesting cod (as specified by (5.12)) as follows. The available cost data for each vessel contains expenses made for fuel, salt and packing, social costs, wages, vessel insurance, other insurance, vessel maintenance, gear and equipment maintenance, provisions, vessel depreciation, and a category "other costs". In total, there are 11 cost components, which are indexed $k = 1...11$. Total costs incurred by vessel i of type z in year t are given by the vector of nominal cost components, C_{izkt} which are subsequently corrected for inflation using the Producer Price Index PPI_t. We calculate the part of the total costs incurred for catching cod by the share of days vessel i spends on catching cod in the total number of days vessel i is fishing at sea. Using index j to enumerate these nine fish species (with cod being $j = 9$) and using D_{izjt} to denote the number of days in year t that vessel i of type z catches species j, the costs attributed to catching cod by a vessel i of type z in year t are

$$c_{izt} = \frac{D_{iz9t} \sum_{k=1}^{11} c_{izkt}}{PPI_t \sum_{j=1}^{9} D_{izjt}}. \tag{5.25}$$

[21] The only study we are aware of that finds an effort-output elasticity of larger than one is Eide et al. (2003). This is somewhat surprising, but may be explained by the fact that they use daily data, where effort is given by hours of trawling – economies of scale do not necessarily materialize at the very short term.

To estimate (5.12), we use (5.25) as the dependent variable and regress it on an intercept as well as on the number of tonnage-days vessel i spent harvesting cod in year t. As before, we use fixed effects and panel corrected standard errors to estimate

$$c_{izt} = a_{0z} + a_{1z}e_{izt} + a_{2zi} + \varepsilon_{izt}, \qquad (5.26)$$

where the intercept a_{0z} equals the fixed adjustment costs per boat operating in the cod fishery (f_z), while a_{1z} reflect variable costs per tonnage-day (v_z). The coefficient for the fixed effects a_{2zi} can be interpreted as individual deviations from a_{0z}. Table 5.3 shows the estimation results. Multiplying the variable costs per tonnage-day by the average size of a boat (in BRT) gives the variable costs for one day of fishing cod for each vessel type: these are 36,960 Norwegian Kroner (NOK) per trawler (of average size 280 BRT), 169,386 NOK per factory trawler (of size 777 BRT), and 51,863 NOK per longliner (of size 216 BRT). The fixed costs per year are 1.55 million NOK per trawler, 2.89 million NOK per factory trawler, and 0.275 million NOK per longliner. We have performed two robustness checks to validate our results. First we have arbitrarily split the total costs into variable and fixed costs and compared them with our findings here. Second, we have estimated the cost and production function jointly, by combining equations (5.24) and (5.26). That model was judged to be inferior compared to the one estimated directly. The results of these robustness checks and more model validations are presented in the Appendix.

Table 5.3. Regression output for the cost functions of the three vessel types. The standard errors are presented in parentheses.

	Trawlers	Factory Trawlers	Longliners
Fixed adjustment costs in million NOK	1.55 (0.21)	2.89 (0.74)	0.28 (0.15)
Variable costs per tonnage-day in NOK	131.66 (8.55)	218.49 (17.98)	239.38 (15.21)
Durbin Watson	1.36	1.25	2.24
Adjusted R^2	0.84	0.78	0.77
Total observations	348	157	226
Number of boats	84	22	64

5.4.2 The inverse price elasticity of Northeast Arctic cod

Estimating the inverse demand function for cod (see (5.2)) is complicated, because the price and quantity data are *equilibrium* outcomes of market interactions, and hence are the result of both demand and supply. Of course, when the TAC is set by a manager, the supply is exogenous and OLS would give unbiased estimates of the demand function. However, there may still be supply effects if (i) the manager decides on the TAC in an ad hoc manner and may set higher quotas when world prices are high, (ii) there is a tendency to harvest illegally when prices are high, or (iii) the quotas are not fully exercised when prices are really low. Because of these reasons we use Two Stage Least Squares (2SLS), instrumenting for landings in the first stage of 2SLS using biomass levels of the past two years as instruments.

We estimate the inverse elasticity of demand – the price flexibility – using export-prices. Ex-vessel prices and export prices are co-integrated (Asche et al., 2002), and therefore one can use the price flexibility to construct a demand function for the ex-vessel fish market. While in our theoretical model we assume the demand function for cod to be linear, econometrically it is preferred to estimate the demand function using a log-linear specification. Hence, our regression model is

$$\log(P_t) = a_{21} + a_{22}\log(H_t) + a_{23}\log(\mathrm{Inc}_t) + a_{24}\log(\mathrm{S}_t), \tag{5.27}$$

where P_t is the deflated price of cod (in NOK), H_t are the total landings, Inc_t is disposable income (given by real GDP in Europe), and S_t is the price of a substitute product (saithe).[22] The time series for landings and biomass suffer from autocorrelation. We therefore, estimate (5.27) as an ARMA(1,1) process. Following Fair (1984), the lagged values of P_t and H_t will be used as instruments; see also Pindyck and Rubinfeld (1991). We obtain the following estimates and standard errors:

$$\begin{array}{lllll} \log(P_t) = -21.87 & -\,0.50\log(H_t) & +\,1.95\log(\mathrm{Inc}_t) & +\,0.50\log(S_t) & +\,u_t, \\ \text{s.e.} \quad\quad (5.41) & (0.1) & (0.35) & (0.12) & \end{array}$$

$$\begin{array}{lll} u_t = & -0.31u_{t-1} & +\,0.80\varepsilon_{t-1} + \varepsilon_t, \\ \text{s.e.} & (0.62) & (0.91) \end{array}$$

with an adjusted R^2 of 0.97 and Durbin-Watson (DW) statistics of 1.27. We find that the inverse price elasticity is 0.5, i.e. if the supply of cod increases by 1%, the world price drops by 0.5%. The Durbin-Watson statistic is a bit small, indicating that we may not have fully succeeded in solving the issue of autocorrelation. As a further robustness check we have estimated the same model in different specifications (see Appendix). These additional estimations support an inverse price elasticity around 0.5.

[22] Due to constraints in the fishing process, it seems unlikely that fishermen can substitute saithe for cod – at least not in the same way consumers do. If they could, using saithe in the demand function would be problematic (as it may measure a supply effect). The size of the coefficient and robustness checks in the Appendix suggest that this is not a serious problem.

From (5.2), the inverse price elasticity is given by $(H/P)(dP/dH) = -bH_t / P_t$, which should equal 0.5. Using the inflation-corrected average price per kilogram of cod between 1997 and 2007 of 12.59 NOK, and the annual average landing of cod of 527,815 tonnes, we find $b = 1.19 \cdot 10^{-8}$. Substituting this value of b, together with the price and quantity data, into (5.2), we find $a = 18.88$.[23] Hence, the price of a kilogram of cod is given in our model as:

$$P_t = 18.88 - 1.19 \cdot 10^{-8} H_t. \tag{5.28}$$

5.4.3 The biological model

Because our biological data is given in years, while our biological model (5.4) and (5.5) is continuous, we use the following model to approximate the growth function of cod:

$$X_t - X_{t-1} - H_t = a_0 X_{t-1} + a_0 b_0 X_{t-1}^2 + \varepsilon_t, \tag{5.29}$$

where the error term ε_t is expected to follow an autoregressive process. Here, a_0 yields the point estimate for *r*, and b_0 is our point estimate for $1/K$. The results are presented in Table 5.4.

Table 5.4. Regression output for the biological model. The standard errors are presented in parentheses. The data covers the period 1946-2007.

r	0.54 (0.06)	0.55 (0.09)	0.56 (0.08)
K^{-1} (in 10^{-7} tonnes)	1.78 (0.2)	1.85 (0.34)	1.87 (0.31)
AR(1)		0.25 (0.15)	0.31 (0.14)
AR(2)			-0.18 (0.14)
Durbin Watson	1.50	1.89	2.02
Adjusted R^2	0.01	0.03	0.05
AIC	28.39	28.37	28.38
Total observations	61	60	59

The model with the lowest AIC gives a carrying capacity (K) of 5.41 million tonnes and an intrinsic growth rate (r) of 0.55. These estimates are similar to the results obtained by Kugarajh et al. (2006). Hence, equation (5.5) reads as $G(X) = 0.55X\left(1 - \frac{X}{5.41}\right)$. The biomass that supports a maximum sustainable yield (MSY) thus equals $5.41/2 = 2.7$ million tonnes of cod, and the associated MSY is equal to 743,000 tonnes.

[23] Using not average prices and landings, but the price in a specific year gives slightly different values for a and b around our estimate. If one is particularly interested in a specific year, it would obviously better to use these year-specific estimates.

5.5 Using the empirical results for an optimal policy

We can now use the estimates obtained in section 5.4 to derive the optimal quota sizes per vessel of each type, the optimal amount of biomass, and the associated producer surpluses as indicated in section 5.3. We start by calculating the optimal scale of operation (as measured in tonnage-days) for each of the three vessel types (see equation (5.16)), and also the optimal number of fishing days (by dividing (5.16) by the average BRT of the boat type). We find that the optimal number of days fishing cod for the three boat types varies between 62 for longliners and 238 for trawlers; see Table 5.5. This is more than the time the boats spend currently catching cod (see Table 5.1) and consistent with intuition because the current situation is most likely characterized by overcapacity.

We now turn to the question how to set an optimal total allowable catch. First, we present results for the situation where the fleet composition is flexible, and there is no upper limit on the number of boats that can be used (as derived in section 5.3.2). We start by calculating the average costs of harvesting one kilogram of cod for each of the three vessel types, as given by (5.20). At a given level of remaining biomass equal to X, the costs equal $C_T = 1{,}327{,}706 / X^{0.58}$ in case of trawlers, $C_{LL} = 955 / X^{0.22}$ in case of longliners, and $C_{FT} = 23{,}557 / X^{0.38}$ in case of factory trawlers. Multiplying these numbers by TAC_z gives the total costs of all boats belonging to one fleet type; see (5.19).

Table 5.5. Optimal number of days fishing cod for different boat types, as measured by days and tonnage-days.

	Trawlers	Longliners	Factory Trawlers
Optimal tonnage days	667122	134514	1070196
Optimal days	238	62	138

Furthermore, we find that trawlers have always lower average harvesting costs than factory trawlers (when both are operating at their efficient scales), for all levels of biomass between 0 and *K*.[24]

[24] In our analysis we ignore the fact that factory trawlers create added value by processing the fish on board. That means that our regression results underestimate the benefits (or overestimate the costs) from using factory trawlers that produce frozen fish fillet rather than raw fish. Therefore, it seems unfair to compare them with the other boats. Hence, factory trawlers will be omitted from the rest of the analysis, except in the scenario where a diverse fleet is preferred by society.

Similarly, trawlers have lower average harvesting costs than longliners, as long as biomass is 530,000 tonnes or higher. In the analysis that follows, it is shown that if biomass is below this level, it will be desirable to stop fishing altogether. Therefore, we can conclude that – if fishing takes place – trawlers are always the cost-minimizing option; $LC1 = T$.

Next, we use our model to determine the optimal long run equilibrium values for biomass, TAC, and fleet profits for the various management objectives; see Table 5.6. Three scenarios will be compared. First, the government minimizes fleet costs and allocates all quotas to trawlers (potentially through an ITQ mechanism). Second, the government not only cares about financial welfare but pursues environmental objectives too, and hence allocates all quotas to longliners as they are more environmentally-friendly than trawlers. Third, we also solve for the case where the government has other objectives too (like cultural and social considerations), embodied by assuming that society prefers a diverse fleet as it operates currently.[25]

We obtain the following results. First, we find that discounting has a negligible impact on optimal long run policies – Table 5.6 shows that the optimal biomass levels for a discount rate of zero percent are less than two percent smaller than for a discount rate of ten percent. Second, independent of the fleet composition, we find that the optimal biomass is always larger than the maximum sustainable yield stock of 2.7 million tones of cod (see section 5.4.3).

Table 5.6. Optimal steady state biomass and harvest levels for several harvesting scenarios for maximizing rents (using only trawlers), environmental concerns (using only longliners), and cultural diversity (using trawlers, factory trawlers, and longliners)

Management objective	Discount rate	Biomass (mln tonnes)	Harvests (mln tonnes)	Profits (bln NOK)
Maximizing rents	0%	3.94	0.59	4.86
Environmental concerns	0%	4.35	0.47	2.84
Fleet diversity	0%	4.12	0.54	3.97
Maximizing rents	10%	3.89	0.60	4.86
Environmental concerns	10%	4.34	0.47	2.84
Fleet diversity	10%	4.09	0.55	3.96

[25] Since our dataset comprises almost all boats that are engaged in the cod fishery, we derive the parameters from our dataset. That is $\theta_T = 0.478$, $\theta_{FT} = 0.185$, $\theta_{LL} = 0.337$.

That means that search costs and a lower price that can be obtained for higher catches are more important than discounting (even when using a discount rate of 10%), as the optimum is always on the right hand side of the logistic growth function. Third, we find that the optimal biomass is smallest in the case fleet profits are maximized and the cheapest boats – trawlers – are used (and hence the amount of cod caught is largest). The remaining biomass is largest in case of environmental concerns, when the government only allows cod harvesting to take place by longliners.[26] Our results are similar to results obtained by Armstrong (1999), who found a TAC of 650,000 tonnes to be optimal and Armstrong and Sumaila (2000) who found an optimal TAC of 450,000 tonnes for NEA cod.

So we find that the optimal biomass levels are much higher than the biomass levels that we are currently experiencing; see Figure 5.1. This raises the question how the transition path looks like – as given by the HCR (10); see Figure 5.2.

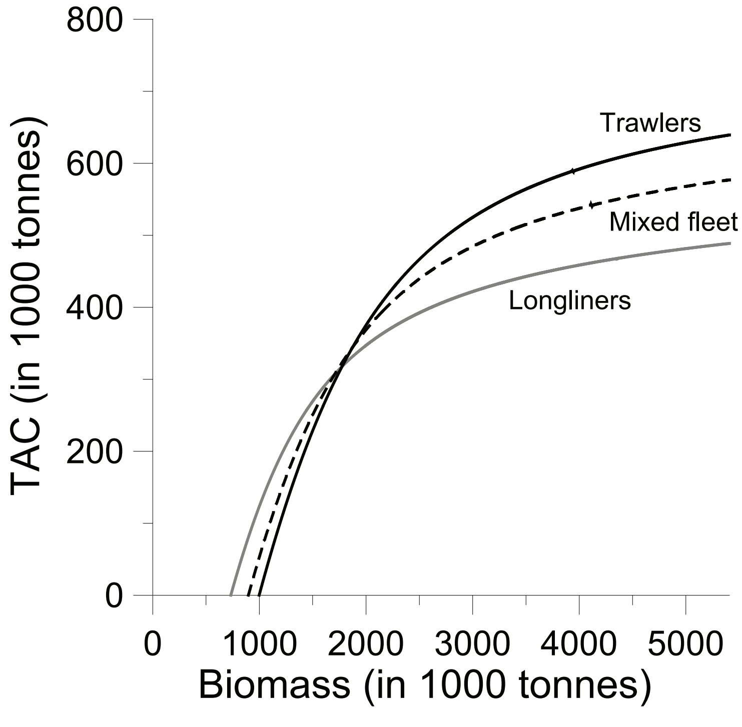

Figure 5.2. The optimal harvest control rule for using only trawlers, longliners, or a mixed fleet.

[26] Recall that we assume the government to maximize producer surplus (with or without constraints on the type of boats used). If the government not only cares about producer surplus but also about consumer surplus (and hence aims to maximize social welfare as the sum of consumer and producer surplus), the optimal steady state biomass is 3.14 mln tonnes with an associated TAC of 0.72 mln tonnes (trawlers), 3.33 mln and 0.70 mln tonnes (longliners), and 3.20 and 0.72 mln tonnes (mixed fleet).

First, we find that the optimal HCR is concave: a one percent decrease in the remaining biomass calls for a more than proportional decrease in the quantity harvested. Second, we find that the minimum biomass level, below which all fishing activities should be ceased, is higher for trawlers than for longliners – as trawlers are getting relative inefficient at low biomass levels (see the stock-output elasticity in Table 5.2). Third, we find that at higher biomass levels, it would be optimal to harvest more with a fleet of trawlers compared to longliners.[27] The minimum biomass levels we find are similar to the ones found by Arnason et al. (2004), who start fishing at 1.3 million tonnes of biomass, and Kugarajh et al. (2006) who found it optimal to start fishing at around one million tonnes of biomass. In these studies, harvesting was never higher than around 500,000 tonnes (Arnason et al., 2004) and 750,000 tonnes (Kugarajh et al., 2006).

If the situation is characterized by fleet lock-in, it is not necessarily possible to follow the cost-minimizing policy identified in Table 5.6. If the optimal TAC of 590,000 tonnes needs to be harvested using trawlers operating at their efficient scale, 206 vessels are needed. Let us analyze the case where only 90 trawlers are available.[28] Plugging in all estimated parameters into (5.20) we find that it is cheapest to let trawlers operate 365 days a year, before using longliners if biomass levels are above 600,000 tonnes (cf. (5.23)); see table 5.7.

Table 5.7. Optimal steady state biomass and harvest levels for the case of cost-minimization harvesting if the fleet is characterized by a lock-in.

Discount rate	Biomass (mln tonnes)	Harvests (mln tonnes)	Profits (bln NOK)	# Trawlers (number of vessels)	# Longliners (number of vessels)
0%	4.42	0.44	4.34	90	97
10%	4.40	0.44	4.34	90	119

Therefore, if the manager aims to maximize the rents from cod harvesting, it should force all 90 trawlers to operate the whole year. In addition, it is then optimal to also have 97 longliners active in the cod fishing industry, each of which operates at its efficient scale (see the third line in equation (5.23) and Table 5.7). Surprisingly, the optimal amount of biomass is then larger than when

[27] For the HCR, we find that maximizing producer surplus leads to somewhat smoother harvesting activity than maximizing the sum of profits and consumer surplus (as harvesting is continued at lower biomass levels, but not as aggressive at higher biomass levels). Then, the minimum biomass levels would be around 1.5 million tonnes.

[28] We do not know the exact number of available vessels. As a benchmark, the number of trawlers holding a license for catching cod has decreased from 102 in 2000 to 41 in 2009. However, in reality the TAC is also harvested by Russian vessels and smaller coastal vessels. Therefore the current situation can still be characterized by overcapacity – in spite of a substantial reduction of the Norwegian fleet over the past 10 years.

using just trawlers or longliners (and hence the TAC is smaller, given that the optimal biomass lies on the right hand side of the logistic growth curve); compare Tables 5.6 and 5.7. The explanation for this counter-intuitive finding is that using additional longliners will negatively affect the efficiency of trawlers through the stock effect. At lower biomass levels, trawlers are getting less efficient than longliners (see the estimates of the stock-output elasticities (α_z) presented in Table 5.2, and equation (5.9)). This indirect cost explains why it is optimal to use less longliners – given that the trawler fleet is already operating at maximum capacity.

5.6 Discussion and conclusions

In this chapter we developed a management plan to determine the optimal steady state biomass, total allowable catch, and the associated welfare for three different management objectives for the Norwegian Northeast Arctic cod fishery. Our model allows the decision maker to determine the optimal allocation of a total allowable catch over the various types of vessels currently used in the fishery (trawlers, longliners and factory trawlers). Having derived the associated cost functions of catching cod, the information can subsequently be used to determine the optimal steady state level of biomass, as well as the harvesting trajectory towards it (the so-called harvest control rule). All equations of the model have been estimated using detailed data of the NEA cod fishery, while addressing the many statistical difficulties associated with them.

Our analysis shows that fleet structure is important for optimal policy as it determines not only how many boats optimally harvest a given TAC, but also the size of the overall TAC and the associated optimal biomass levels. Taking the cost structure of the industry into account affects the optimal biomass levels substantially, as the steady state stock is 3.94 million tonnes in the case the government aims to maximize long-run financial welfare (implying that the fishing fleet should consist of trawlers only, because they are most efficient in harvesting) while it is 4.35 million tonnes if the government aims to maximize long-run welfare while limiting the fleet structure to consist of longliners only – as they are indicated to impose least damages to the marine ecosystem. These results can be used by the decision maker to decide how to set an optimal TAC, to choose which vessels are allowed to participate in the fishery, and assess the costs of deviating from the least-cost approach. If there is a maximum number of vessels available in the cod fishery (as analyzed in section 5.3.4), cost-minimization may require using longliners in addition to trawlers. Interestingly, in such a case optimal total biomass is even higher than when only trawlers or longliners are used. This happens because harvesting of one vessel affects the productivity of all other vessels through the stock-effect.

Some model assumptions deserve special attention, as they may have potential policy implications. First, we assume cod growth to be represented by a simple logistic growth model. Adding more biological realism may alter our results; see Eikeset et al. (2010c) and chapter 6 for a study that uses a more complex biological model to determine an optimal HCR. While the way in which the cost structure of different vessel types affect the management plan would carry over to a model allowing for more biological complexity, the specific optima – such as the size of optimal biomass or the TAC – will probably be different. Second, we focus our analysis on cod harvesting ignoring all economic and ecological interactions with other fish species (Nøstbakken, 2006; Salvanes and Squires, 1995; Squires et al., 1998). An interesting further avenue would be to investigate how the economies of scale that we identified in this paper relate to economics of scope (i.e. the possibility to catch other fish species). Third, we assume that each fleet can be represented by a typical boat. In reality, a fleet comprises many boats that differ in age, size, productivity, and costs. This is not accounted for in our model, giving rise to potential inefficiencies when quotas are distributed over boats. These inefficiencies could be eliminated by an ITQ mechanism (within the fleet constraints that we have outlined), even though the costs and disadvantages of such a mechanism can easily outweigh the potential benefits (Sumaila, 2010); see also chapter 7. Fourth, while our results give optimal quotas in tonnage-days, in practice it would be desirable to hand out the actual quotas – transferable or not – as catch shares to remove the incentive to substitute controlled for uncontrolled capital.

The study presented here is novel, as it provides an optimal management plan that is flexible and can be adapted to various policy objectives concerning the utilization of the fleet going beyond cost-minimization. However, it can only be considered a first step towards optimal management of natural resources that recognizes the full array of preferences society holds regarding how these resources should be exploited.

Appendix 5.A: Deriving an analytical harvest control rule

Deriving an HCR is fairly straightforward if the discount rate is assumed to be equal to 0 $(\delta = 0)$; see Sandal and Steinshamn (1997; 2001). The current-value Hamiltonian of the optimal control problem in section 5.3.1 can be written in general terms as follows:

$$\mathscr{H} = \Pi\left(X, TAC\right) + \varphi\left(G(X) - TAC\right), \tag{5.A1}$$

and the associated first-order conditions are

$$\mathscr{H}_{TAC} = 0 \quad \Rightarrow \quad \varphi = \Pi_{TAC}(X, TAC), \tag{5.A2}$$

$$\dot{\varphi} = \delta\varphi - \mathscr{H}_X \quad \Rightarrow \dot{\varphi} = -\Pi_X\left(X, TAC\right) + \varphi\left(\delta - G_X\left(X\right)\right), \tag{5.A3}$$

$$\dot{X} = \mathscr{H}_{\varphi}. \tag{5.A4}$$

We aim to derive the optimal harvest control rule which is, by definition, a function of the stock of biomass, $TAC(X)$. Substituting this generic expression of the HCR into (5.A1) and taking the first derivative with respect to time, we have

$$\dot{\mathscr{H}} = \left(\frac{\partial \mathscr{H}}{\partial X} + \frac{\partial \mathscr{H}}{\partial TAC}\frac{dTAC}{dX}\right)\dot{X} + \mathscr{H}_{\varphi}\dot{\varphi} = \delta\varphi\dot{X} \tag{5.A5}$$

where the latter equality holds because of (5.A2) and (5.A3); see also Sandal and Steinsham (2001). That means that the Hamiltonian is constant over time if $\delta = 0$. In that case, maximizing the Hamiltonian then boils down to just choosing X to maximize the instantaneous profit flow in steady state (Sandal and Steinshamn, 2001):

$$\mathscr{H} = \max_X \Pi\big|_{TAC=G(X)} = \Pi^*(X) \tag{5.A6}$$

Substituting (5.A6) and (5.A2) into (5.A1), we obtain the following equality:

$$\Pi^*(X) = \Pi(X, TAC) + \Pi_{TAC}(X, TAC)\left[G(X) - TAC\right]. \tag{5.A7}$$

Using the instantaneous profit function (5.1), the growth function (5.5) and (5.A7), we find the optimal harvest control rule as presented in (5.10).

Appendix 5.B: Model validations

Model validation of the production and cost functions

As a robustness check, we have split total costs into variable and fixed costs, by assuming that variable costs contain expenses made for fuel, salt and packing, social costs, provisions, wages, and a category "other costs". We found that the average cost per trawler fishing cod is 36,823 NOK, per factory trawler 158,452 NOK, and per longliner 47,285 NOK. The fixed costs that are assumed to comprise vessel insurance, other insurance, vessel maintenance, gear and equipment maintenance, and vessel depreciation are 1.6 million NOK for trawlers, 3.57 million NOK per factory trawler, and 0.75 million NOK per longliner. These guesstimates are comparable to what has been estimated in our regression analysis, even though the fixed costs in the regression analysis are somewhat lower, especially for longliners. This may be due to the fact that some costs that were assumed to be fixed here (e.g. maintenance) are in reality partially dependent on effort. Since our regression analysis is able to capture this effect, while the ad hoc composition here is not, we are confident that our regression results are the more accurate ones.

As a further robustness check we have estimated the relationship of costs and harvests jointly without using effort as a variable, by combining equation (5.24) and (5.26). The equation

$$\log\left(TC_{it}\right) = \log(a_0) + a_1 \log(X_t) + a_2 \log(h_{it}) + \upsilon_{iz} + \varepsilon_{it} \qquad (5.A8)$$

is again estimated with fixed effects. The results are presented in Table 5.A1. We find a stock-output elasticity smaller than one and an effort-output elasticity smaller than zero, which is consistent with our earlier findings. In order to judge the quality of each model, we will assess their forecasting ability. Table 5.A2 shows the forecasting ability of the production functions (5.24) and cost functions that are estimate directly (5.26) compared with the model (5.A8). All models have been estimated in the period 1990-1995, while the period 1996-2000 has been forecasted. The bias proportion indicates how the mean of the predicted time series differs from the original one, while the variance proportion tells us how the variance of the two series differs. We find that the models (5.24), and (5.26) perform better than model (5.A8), indicated by the lower bias and variance proportion and consequently the higher covariance proportion.

Table 5.A1. The coefficients of model (5.A8), estimating the relationship between costs, total biomass and harvests.

	Trawlers	Factory Trawlers	Longliners
a_0	13.67 (1.04)	12.75 (1.83)	8.92 (1.37)
a_1	-0.44 (0.06)	-0.39 (0.08)	-0.21 (0.06)
a_2	0.81 (0.03)	0.82 (0.05)	0.81 (0.05)
Durbin Watson stat.	2.26	1.64	2.64
Adjusted R^2	0.97	0.83	0.79
Total observations	437	169	309
Number of boats	107	25	85

Table 5.A2. The forecast ability of models (5.24) and (5.26) estimated directly and model (5.A8) estimated indirectly.

	Trawlers			Longliners			Factory Trawlers		
Model	Production	Cost	Jointly	Production	Cost	Jointly	Production	Cost	Jointly
Equation	(5.24)	(5.26)	(5.A8)	(5.24)	(5.26)	(5.A8)	(5.24)	(5.26)	(5.A8)
Bias proportion	0.344	0.258	0.078	0.36	0.089	0.52	0.455	0.248	0.476
Variance proportion	0.013	0.019	0.32	0.12	0.026	0.059	0.012	0.006	0.02
Covariance Proportion	0.643	0.723	0.602	0.52	0.885	0.421	0.533	0.746	0.504

Model validations of demand function

We have estimated the inverse price elasticity under various different specifications to evaluate the robustness of our results. First, we have estimated (5.27) as an ARIMA(0,1,0) process. This delivers a slightly lower inverse price elasticity of -0.40 as given by

$$\Delta\log(P_t) = 0.0 - 0.40\Delta\log(H_t) + 1.72\Delta\log(\text{Inc}_t) + 0.62\Delta\log(S_t) + \varepsilon_t$$

s.e. $(0.06)(0.14)$ (2.34) (0.15),

with an adj. R^2=0.36 and DW=1.92. Instruments for $\Delta\log(H_t)$ are $\Delta\log(B_{t-1}), \Delta\log(B_{t-2})$.

Furthermore, the same model has been estimated without income as a variable, giving an elasticity of -0.42 and the following results:

$$\Delta\log(P_t) = 0.03 - 0.42\Delta\log(H_t) + 0.59\Delta\log(S_t) + \varepsilon_t$$

s.e. (0.03) (0.13) (0.15),

with an adj. R^2=0.36 and DW=1.89. Instruments for $\Delta\log(H_t)$ are $\Delta\log(B_{t-1}), \Delta\log(B_{t-2})$.

Omitting the price of saithe gives the same results:

$\Delta\log(P_t) = 0.08 - 0.42\Delta\log(H_t) + \varepsilon_t$

s.e. $\quad (0.03) \; (0.17)$,

with an adj. R^2=-0.02 and DW=1.62. Instruments for $\Delta\log(H_t)$ are $\Delta\log(B_{t-1}), \Delta\log(B_{t-2})$.

Alternatively, we have re-estimated model (5.27) without landing as independent variable, but with export quantity directly.

$\log(P_t) = -19.33 - 0.61\log(Q_t) + 1.76\log(\text{Inc}_t) + 0.73\log(\text{S}_t) + u_t$

s.e. $\quad (3.29) \; (0.07) \quad (0.22) \quad (0.09)$,

$u_t = -0.16u_{t-1} + 0.09\varepsilon_{t-1} + \varepsilon_t$

s.e. $\; (1.53) \quad (1.50)$

with an adjusted R^2=0.98, DW= 1.90. Instruments for $\log(Q_t)$ are $\log(B_{t-1}), \log(B_{t-2})$.

The estimated elasticity is a bit higher now (0.61), which is not unexpected, because exports quantities have a much more direct impact on prices than landings. All in all, we can conclude that the estimated inverse price elasticity of -0.50 seems reasonable.

Data sources

Equations 5.24-5.26: Data for harvests, costs and effort has been obtained by the Directories of Fisheries, Bergen, while biomass comes from ICES (2009a). The cost data has been deflated with the Producer Price index for Norway taking from the OECD, (2008) using the year 2000 as a benchmark. The OECD data has been accessed via www.SourceOECD.org/database/OECDStat.

Equation 5.28: Landings and biomass are taken from ICES (2009a). Export prices for cod and saithe are inferred from export values and export quantities; see Timmer and Richter (2009) for more information on the method. For each export commodity i ("Atlantic cod, fresh or chilled", "Atlantic cod, frozen", "Atlantic cod, salted, or in brine", "Cod, dried, unsalted", "Cod, salted, and dried" a price is calculated by dividing the total value in a given year by the total quantity: $P_{it} = V_{it} / Q_{it}$. A weighted export price is obtained by multiplying each price by its value and dividing it by the value of all exports given by $P_t = \frac{\sum_{i=1}^{5} P_{it}V_{it}}{\sum_{i=1}^{5} V_{it}}$. The data for saithe is given by "Saithe, dried, salted or in brine". This data was accessed with Fish Stat Plus (FAO, data from "FAO Yearbook of Fishery Statistics – Commodities"; the data was collected originally by Statistics

Norway). This is annual data for the period 1976-2006. European income is proxied by real European GDP (Maddison, 2010). The data has been corrected for inflation, and has been converted from US Dollar into Norwegian Kroner using exchange rates from the OECD (2010).

Equation 5.28: The KG price for cod is given by the off-boat sales prices ("Førstehåndspris") as given by the Directories of Fisheries, Bergen (Fiskedirektoratet, 2007). To make the price data comparable with the costs data we have used, again, the producer price index from the OECD. The baseline year was 2000 (as before). The average KG price is the average price between 1997 and 2007.

Equation 5.29: Landings and biomass are taken from ICES (2009a).

6

The economic repercussions of fisheries-induced evolution

Abstract

Human-induced changes in life-history traits have been observed for many harvested populations, with a component of those changes being attributed to an evolutionary (i.e., genetic) response. Most notably, fish stocks that experience high fishing mortality show a tendency to mature earlier and at a smaller size. Some have suggested that fisheries-induced evolution could affect the fishery's yield and therefore have economic repercussions for society. Yet, this has not been formally investigated. We use data from 1932 to 2005 to develop a bio-economic model specifically for Northeast Arctic cod that allows us to compare the economic yield in scenarios with and without evolution of key life-history traits. We also compare a "business as usual" scenario where fishing continues at its current pace, with a scenario in which harvest is controlled through an optimal control rule. Our model predicts that fisheries-induced evolution decreases economic yield if fishing mortality rates continue at their current high levels. We also find that maximum economic yield is achieved at a considerably lower fishing mortality than what the stock has historically experienced. At this lower mortality, fisheries-induced evolution is less pronounced and actually increases the spawning stock biomass and economic yield. Overall, we find that evolutionary and non-evolutionary models recommend similar harvesting rates and the overriding message is that higher economic yield can be obtained by lower harvest rates irrespective of whether evolution occurs or not.

This chapter is a modified version of: Eikeset, A.M., A.P. Richter, E.S. Dunlop, E. Nævdal, U. Dieckmann, and N.C. Stenseth, 2010. *The economic repercussions of fisheries-induced evolution.* Manuscript. Valuable comments and advice on earlier drafts of this manuscript were provided by P. Sandberg, J. A. Hutchings and C. T. Marshall, K. Enberg, C. Jørgensen, L. Nøstbakken, T. Schweder, A. Skonhoft, C. Armstrong, C. Brinch S. I. Steinshamn, J. Grasman and D. P. van Soest.

6.1 Introduction

Today, management decisions in fisheries throughout the world are based on the assumption that evolutionary changes caused by harvesting do not occur (Jørgensen et al., 2007). However, experimental studies have demonstrated that the pace of evolution can be rapid, occurring in only a few generations (Conover and Munch, 2002; Reznick and Ghalambor, 2005). Additional evidence from field-based studies strongly suggests that fisheries-induced evolution has occurred in several commercially important stocks (Barot et al., 2004; Darimont et al., 2009; Grift et al., 2007; Grift et al., 2003; Olsen et al., 2005; Olsen et al., 2004; Sharpe and Hendry, 2009), although there is debate as to the relative contribution of environmental factors in driving the observed changes (Andersen and Brander, 2009b; Koons, 2009; Marshall and McAdam, 2007). However, the bulk of life-history theory, experiments, and field-based studies would at least strongly suggest that fishing is capable of inducing genetic adaptations, especially when it preferentially removes individuals with certain characteristics such as large body size (Carlson et al., 2007; Dunlop et al., 2009b; Hutchings, 2009; Hutchings and Fraser, 2008; Jørgensen et al., 2007). Even if fishing is not size-selective, high fishing mortality may be sufficient to induce genetic change (Roff, 1992; Sharpe and Hendry, 2009). The resultant selection pressure imposed by most fisheries favors individuals that mature at younger ages and invest more in reproduction because individuals with those genotypes are more likely to survive and reproduce (Andersen and Brander, 2009a; Dunlop et al., 2009c; Eikeset et al., 2010a; Enberg et al., 2009). The direction of fisheries-induced evolution of growth depends on the size-selectivity of the fishing pressure: if both immature and mature fish are exposed to fishing mortality, growth rates may increase (Dunlop et al., 2009c; Eikeset et al., 2010a). In contrast, size-selective harvesting of large, mature individuals leads to a decreased growth rate (Andersen and Brander, 2009a; Conover and Munch, 2002; Dunlop et al., 2009c).

Genetic adaptations to fishing may, in principle, be beneficial for the state of the stock because individuals invest more in reproduction and growth (Andersen and Brander, 2009a; Dunlop et al., 2009c; Eikeset et al., 2010a; Enberg et al., 2010). However, adapting to the fishing pressure may also bear a cost of evolution through increased natural mortality (Jørgensen and Fiksen, 2010). An additional cost of evolution may occur if genetic changes are difficult to reverse (Conover et al., 2009; Dunlop et al., 2009c; Enberg et al., 2009; Stenseth and Dunlop, 2009). This is sometimes referred to as Darwinian debt that must be paid back by future generations (Dieckmann et al., 2009). Studies have suggested that fisheries-induced evolution might reduce yield (Hard et al., 2008; Hutchings, 2009; Jørgensen et al., 2007; Stenseth and Dunlop, 2009; Sutherland, 1990). However, to our knowledge, no study has explicitly investigated the economic implications of evolutionary change in wild populations.

It is difficult to predict how genetic changes at the individual-level affect population-level properties such as the spawning stock biomass (SSB). For example, an individual's increased reproductive investment leads to larger gonads, but at the expense of slower post-maturation growth. Maturing earlier may also reduce fecundity because individuals are smaller when they reproduce (Marshall et al., 2004). Furthermore, the adaptive response of a population to fishing could actually allow it to withstand higher mortality rates than if it otherwise did not adapt (Eikeset et al., 2010a), perhaps also permitting higher yields. Therefore, evolution has the potential to have both positive and negative effects on stock properties such as total biomass and yield, making the total economic effect ambiguous. Furthermore, other so-called ecological effects such as the release of density-dependence when population biomass is fished down could be more important drivers of phenotypic change, and could effectively swamp out any underlying effects evolution has on yield.

The Northeast Arctic (NEA) cod is currently the world's largest stock of Atlantic cod (*Gadus morhua*). The stock's fishery is an important economic resource for Norway, Russia, and the European Union with an annual catch by Norway of about 3.5 billion NOK in 2008 – see also chapter 5. Traditionally, the fishery harvested primarily adult cod at the spawning grounds along the Norwegian coast (Godø, 2003). From the 1930s onwards, the stock experienced a shift in fishing pressure when industrial trawlers were introduced in the stock's feeding grounds in the Barents Sea (Garrod, 1967). When the trawling fishery began, fishing pressure substantially increased for both adult and juvenile populations, the latter being previously little affected by harvesting as the traditional fishery had mostly been in the spawning grounds (Godø, 2003). Evolutionary changes in this stock have been predicted to be a factor in explaining observed decline in age and length at maturation (Eikeset et al., 2010a; Heino et al., 2002b), although the extent of evolutionary change predicted has varied among studies (Eikeset et al., 2010a; Heino et al., 2002b; Jørgensen et al., 2009).

After the 1930s when fishing mortality was intensified, there has been a steady decline in age and length at maturation, with the age at maturation declining from age 9 to 6 years and the length at maturation declining from about 80cm to 60cm (Eikeset et al., 2010a). Our biological model was specifically developed to investigate the ecological and evolutionary effects of exploitation on these changes in maturation. There are mainly two hypotheses on the importance of ecological versus evolutionary effects in explaining this phenomenon. The first hypothesis claims that these changes have a genetic component where evolution favors those individuals that mature early. The intuitive explanation is that traits can only be passed on if a fish reproduces before getting caught. Therefore, fisheries-induced evolution will occur and it is caused by genetic adaptations to fishing mortality. The second hypothesis states that these changes can be explained by phenotypic plasticity. This occurs as high fishing mortality results in less competition over resources. As a result, fish can grow faster and reach maturation earlier.

Our biological model allows for both hypotheses to occur. Changes in life-history traits may be driven by both ecological processes, like phenotypic plasticity and density-dependence, and through genetic processes. Eikeset et al. (2010a) suggest that fisheries-induced evolution has occurred in NEA cod, but the genetic changes are smaller than what has been suggested in previous studies on Atlantic cod (Edeline et al., 2007; Heino et al., 2002b; Olsen et al., 2004).

Here, we develop a bio-economic model to investigate the economic repercussions of fisheries-induced evolution, specifically testing how genetic change affects economic gain in fleet profits and total welfare (the latter being the sum of fleet profits and consumer surplus). Our model incorporates feedbacks between the state of the resource (i.e. SSB), the economic gains, and management of the fishery (i.e., fishing mortality, Fig. 6.1).

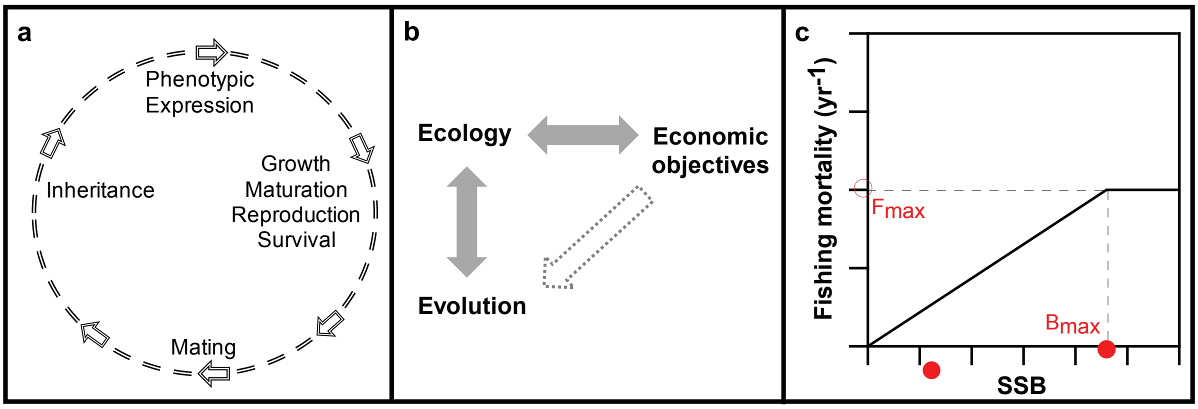

Figure 6.1. The individual-based eco-genetic model and the coupled dynamics between ecology, evolution and economics. **a**, The model framework for the biological component, describing an individual's life cycle. **b,** Ecological dynamics and economic objectives mutually influence each other. Additionally, fishing mortality may induce evolutionary changes. These changes affect the eco-evolutionary dynamics and have economic repercussions. **c,** The shape of an HCR depending on two parameters: Above a certain level of spawning stock biomass, B_{max} a maximum fishing mortality F_{max} is allowed. Between B_{max} and a biomass level of zero, fishing mortality linearly decreases from F_{max} to zero. These parameters can take the shape of the HCR that was implemented for the NEA cod fishery in 2004.

The biological component of our bio-economic model is specifically developed for NEA cod to capture trends in the age and length at maturation from 1932 until 2005; see panel a in Fig. 6.1 and Eikeset et al. (2010a). The biological model is built upon the individual-based eco-genetic model described by Dunlop et al. (2009c) to study fisheries-induced evolution. It describes four evolving life-history traits capturing key components of growth, maturation, and reproduction (Dunlop et al., 2009c): two traits for maturation tendency given by a linear probabilistic maturation reaction norm (Dieckmann and Heino, 2007; Heino et al., 2002b), one trait describing the intrinsic somatic growth capacity, and one trait as a measure of reproductive investment, the gonado-somatic index. The assumptions of our model arise from basic quantitative genetics theory and commonly observed empirical relationships for fish stocks (Dunlop et al., 2009c; Eikeset et al., 2010a). Resource limitation and competition with conspecifics are accounted for through density-dependent growth and newborn mortality (Eikeset et al., 2010a). The model structure accounts for both the demographic and evolutionary effects of fishing on growth, maturation, and reproduction, and allows us to distinguish genetic change from phenotypic plasticity in the modeled populations (Dunlop et al., 2009c; Eikeset et al., 2010a). For more details about the bio-economic model structure, parameter values and data sources, see Appendix. Model limitations and simplifying assumptions are discussed further in Eikeset et al. (2010a) and Dunlop et al. (2009c).

To evaluate whether fisheries-induced evolution requires a unique management plan, we created a non-evolutionary version of the biological model where the genetic traits could not evolve; for more information see Eikeset et al. (2010a) and Appendix. This framework allows us to mimic stock properties to investigate the relative importance of evolutionary change. Hence, we compare a non-evolutionary model, in which changes in populations are driven only by phenotypic plasticity, with an evolutionary model that allows, in addition, for genetic adaptations. Comparisons between non-evolutionary versus evolutionary eco-genetic models have been done previously to study historical trends in NEA cod (Eikeset et al., 2010a) and recovery potential in Atlantic cod (Enberg et al., 2009).

6.2 The bioeconomic model and results

Our economic model has been developed in chapter 5 of this thesis and consists of a Cobb-Douglas production and cost function estimated specifically for the Norwegian cod trawler fleet.

We used these estimations to derive a mechanism that mimics the use of individual quotas in the most efficient way. Additionally, we incorporated a demand function, also estimated from data to account for total catch affecting the landings price. For more information see Appendix and chapter 5. Using our bio-economic model, we compare the characteristics from (i) a scenario of high fishing mortality that mimics the historic level of harvesting with (ii) optimal harvest scenarios. We investigate how the emerging harvest properties differ between the evolutionary and the non-evolutionary model. Furthermore, we ask whether any substantial losses arise from overlooking fisheries-induced evolutionary changes. For each of the fishing scenarios, we compare the trajectory of key life-history traits and economic yield (Fig. 6.1b).

In the first fishing scenario, we analyze stock development according to the observed fishing mortality between 1932 and 2005. Then, from 2006 onwards we continue with "business as usual" with a mean fishing mortality of 0.68 yr^{-1} (Fig. 6.2a). Our optimal management plan is built on the harvest control rule (HCR) implemented for the NEA cod fishery in 2004 (ICES, 2008b, 2009b); see Fig. 6.1c. This HCR allows a specific fishing mortality for a given SSB. The maximum fishing mortality F_{max} is allowed above a certain SSB level, given by the parameter B_{max}. Below B_{max} fishing mortality decreases linearly to the origin. We ran model simulations, searching over a large grid size for the combination of these two parameters (F_{max} and B_{max}) that delivered the best results for our economic objectives (to either maximize profits or total welfare). In the main results discounting is neglected because it does not change the overall results, and we only discuss results for the objective total welfare, noting that maximizing profits recommends a very similar management plan compared to maximizing total welfare; but see the supplementary results in the Appendix.

When fishing mortality is high (Fig. 6.2a), the evolutionary model predicts cod maturing about half a year earlier than in the non-evolutionary model (Fig. 6.2b) due to the underlying genetic adaptations in the evolving population. Also, the predicted total welfare is lower in the model where fisheries-induced evolution occurs, although not by a large amount (Fig. 6.2c). This higher cost of evolution occurs because the total biomass and catch are slightly lower in the evolutionary model than in the non-evolutionary model (See Appendix, Table 6.A2 and Fig. 6.A2).

Given that fisheries-induced evolution has already occurred in the past, what would be the best way to avoid undesired effects in the future? Furthermore, can genetic changes – if undesired – be reversed? To address these questions, we have approximated an optimal management plan for the period 2006-2100 (Fig. 6.2d-f). We find that the optimal fishing mortality in both the evolutionary and non-evolutionary model is substantially lower than it has been historically (indicated by a black arrow in Fig. 6.2d). For both models the predicted age at maturation does not decline as much when lower fishing mortality is implemented from 2006.

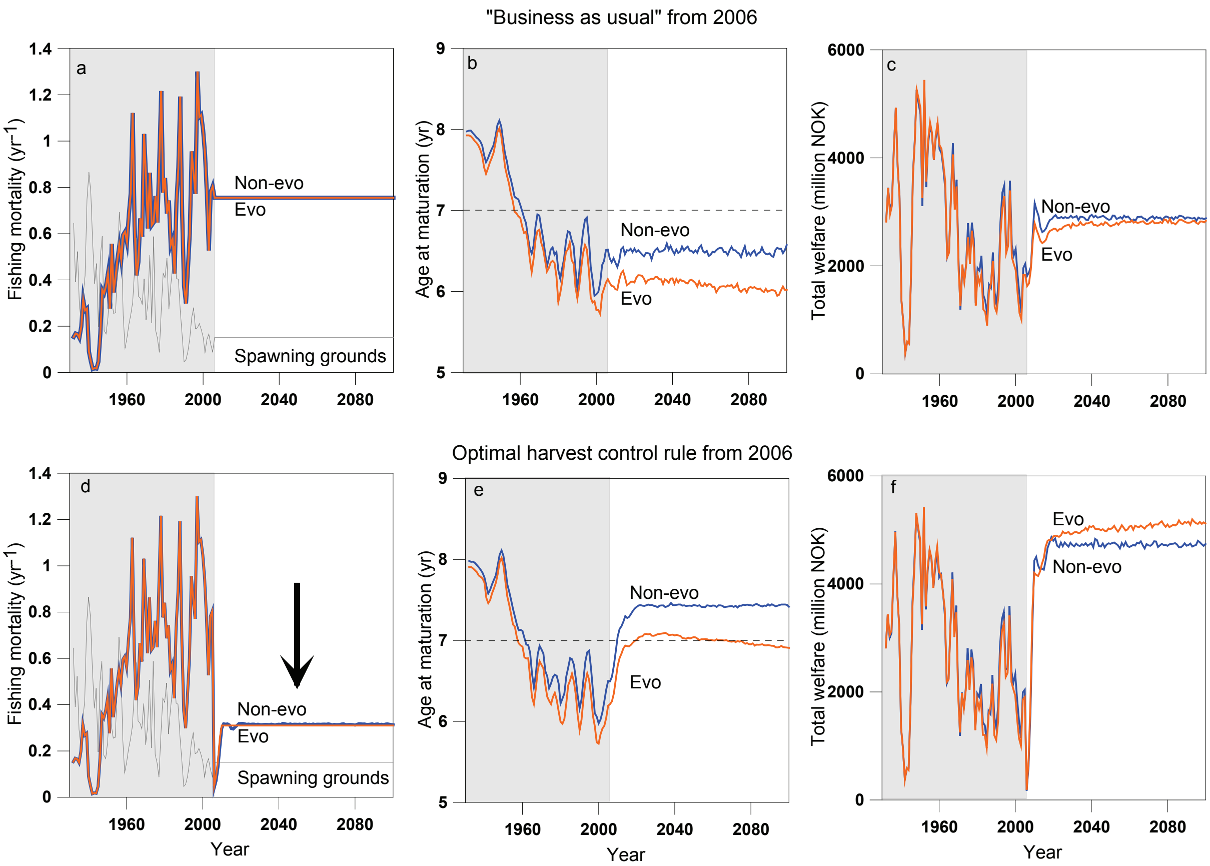

Figure 6.2. The grey shaded area has occurred in the past with observed fishing mortality from 1932-2005. From 2006 on, we have analyzed two different scenarios: one continues with the average fishing mortality that has been observed between 1946-2005, called "business as usual". The other scenario has implemented an optimal HCR from 2006 by maximizing welfare. For each scenario, the emerging properties from an evolutionary model (grey) are compared with a non-evolutionary model (black). The grey line at the bottom in **a** and **d** shows fishing mortality at the spawning grounds which are beyond the control of the manager (not included in deriving the HCR). In **a,** fishing mortality stays as high as it was in the past. In **b**, the predicted age at maturation is lower in the evolutionary model than in the non-evolutionary model. Both models predict age at maturation to fall between age 6 and 7 in 2005 (dashed line), which is what has been observed from data. **c,** The evolutionary model delivers slightly lower welfare than the non-evolutionary model (Table 6.A2). **d,** If each population is managed optimally from 2006, this would demand a much lower fishing mortality (illustrated by a black arrow) in both the evolutionary and non-evolutionary model. **e,** In the evolutionary model the age at maturation is lower, and continues to decrease, while the non-evolutionary model has stabilized around the age of 7 (dashed line). **f,** Although age at maturation is lower in the evolutionary model, total welfare is slightly higher and shows an increasing trend in the evolutionary model. The higher level of welfare in both models is caused by higher spawning stock biomass and catch (see Table 6.A2). Note that the evolutionary and non-evolutionary models are not directly comparable in this figure because they do not have the same starting point in 2006 in respect to stock characteristics. For a direct comparison, see Fig. 6.3, where both models start from 1932. Emerging properties are shown from an average of 15 independent model runs.

Interestingly, the evolutionary model still predicts a lower age at maturation (Fig. 6.2e) but due to a higher SSB, the evolutionary model performs slightly better in terms of welfare (Fig 6.2f and Table 6.A2). Therefore, if fishing pressure is sufficiently low, evolutionary change can increase the SSB, total biomass, catch, and welfare, but if fishing pressure remains at historically high levels, evolution may lead to undesired economic repercussions (Fig. 6.2c and Table 6.A2).

In most stocks, the degree of evolutionary change and its impact are not known, which makes it difficult to manage them from an evolutionary perspective. If evolutionary effects are present, but ignored by managers, how costly will it be to overlook fisheries-induced evolution? The encouraging answer is that a derived optimal HCR in an evolutionary model is not very different than what a non-evolutionary model recommends. For both the evolving and the non-evolving model optimal fishing mortality is equally low (Fig. 3a see also Appendix).

Low fishing mortality avoids the large decline in age at maturation (Fig. 6.3b) that is observed for high fishing mortality (Fig. 6.2b). For both an evolutionary and non-evolutionary model, the same low fishing mortality is required for ecological and economic sustainability. Comparing the overall performance, an optimal HCR in an evolutionary model predicts higher total biomass, higher SSB, and higher catch (Table 6.A3, Fig. 6.A3). As a result, fisheries-induced evolution leads to higher total welfare (Fig. 6.3c).

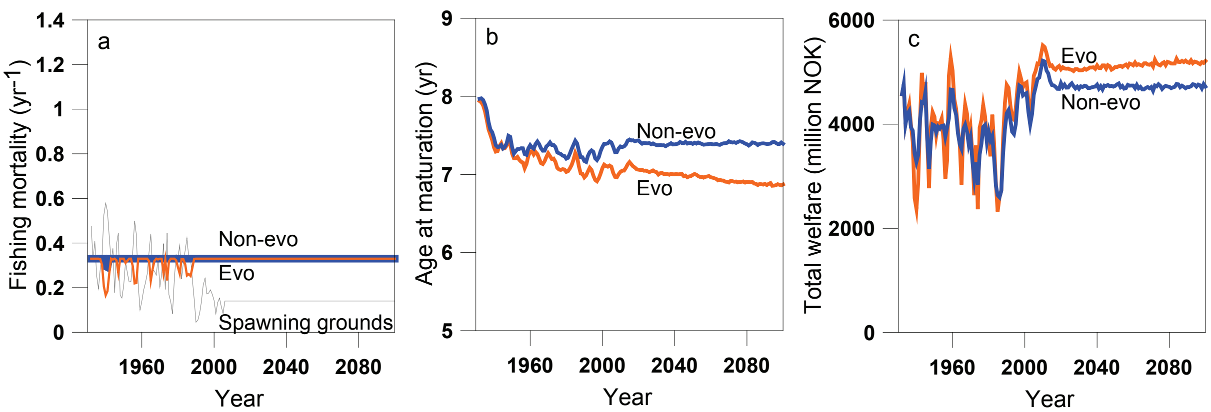

Figure 6.3. The hypothetical case that optimal HCRs had been in use since 1932 in an evolutionary model (grey line) and a non-evolutionary model (black line). The fishing mortality at the spawning grounds (grey) is beyond the control of the manager. **A**, The same low fishing mortality is both optimal in an evolutionary (grey) and non-evolutionary (black) model. **b**, The evolutionary model predicts lower age at maturation than the non-evolutionary model. **c**, Total welfare is higher in the evolving population, caused by higher spawning stock biomass and catch (Table 6.A3). Emerging properties are shown from an average of 15 independent model runs.

Therefore, if the fishing mortality is set correctly according to what would be optimal in a non-evolutionary model, biomass, catch, and welfare are even higher in a model that allows for genetic adaptations. Hence, in this case, ignoring evolutionary effects would not undermine management objectives. If a population in an evolutionary model is managed according to what would be optimal in a non-evolutionary model, total welfare is almost as high, as if fisheries-induced evolution was taken into account when deriving the optimal HCR (Table 6.A3). Hence, using an optimal management plan derived from a non-evolutionary model in an evolutionary model does not result in substantially reduced economic yield and SSB.

This does not necessarily imply that evolution can be ignored. Beneath the surface – and likely to go unnoticed – the qualities and properties of the stock are changing as a result of evolution. Although the age at maturation declines in the non-evolutionary model (solely a product of phenotypic plasticity), the decline is even more severe when evolution takes place. In the evolving population, not only is age at maturation lower but there is also a lower length at maturation, higher intrinsic growth capacity, higher reproductive investment and higher gonad weight caused by the increased reproductive investment (Fig. 6.4a-c and Eikeset et al. (2010a)). These changes have various implications. First, age-truncated populations, consisting of younger and smaller spawners, may lead to a destabilized population growth rate (Anderson et al., 2008). Second, such juvenation may also reduce resilience to climate (Ottersen, 2008; Ottersen et al., 2006). Third, the type of evolutionary changes predicted here may be difficult to reverse in some cases (Dunlop et al., 2009c; Enberg et al., 2009). Fourth, evolutionary changes could alter the stock's migration patterns (Dunlop et al., 2009a; Jørgensen et al., 2008; Theriault et al., 2008) and, fifth, may have wider ecosystem effects that have yet to be fully explored (Jørgensen et al., 2007).

Another result of evolution is a higher ratio between SSB and total biomass from age 3 (Fig. 6.4d). In fisheries management it is very common to use SSB as an indicator for stock viability. We should be aware that with changing maturation schedules, assuming stability in the SSB may mask a decreasing total biomass. Fisheries-induced evolution may have effects on recruitment and the accuracy of recruitment predictions (Enberg et al., 2010). This may increase the risk of stock collapse if biomass levels approach the limit reference points. It has been suggested that using SSB as a measure for stock reproductive potential may lead to overly optimistic assessments of stock status (Marshall et al., 2006). Thus, biological reference points should be set in the light of eco-evolutionary dynamics (Hutchings, 2009).

The biological component of our bio-economic model has several limitations and simplifications (Dunlop et al., 2009a; Dunlop et al., 2009c; Eikeset et al., 2010a; Enberg et al., 2009). A few assumptions merit special attention here. First, we assume an initial 1:1 sex ratio although it has been shown that the sex ratio in this stock has fluctuated over time (Marshall et al., 2006). Second, we assume no sexual selection, though it is possible that sexual selection may influence the evolutionary changes in life-history traits (Hutchings and Rowe, 2008a, 2008b; Urbach and Cotton, 2008).

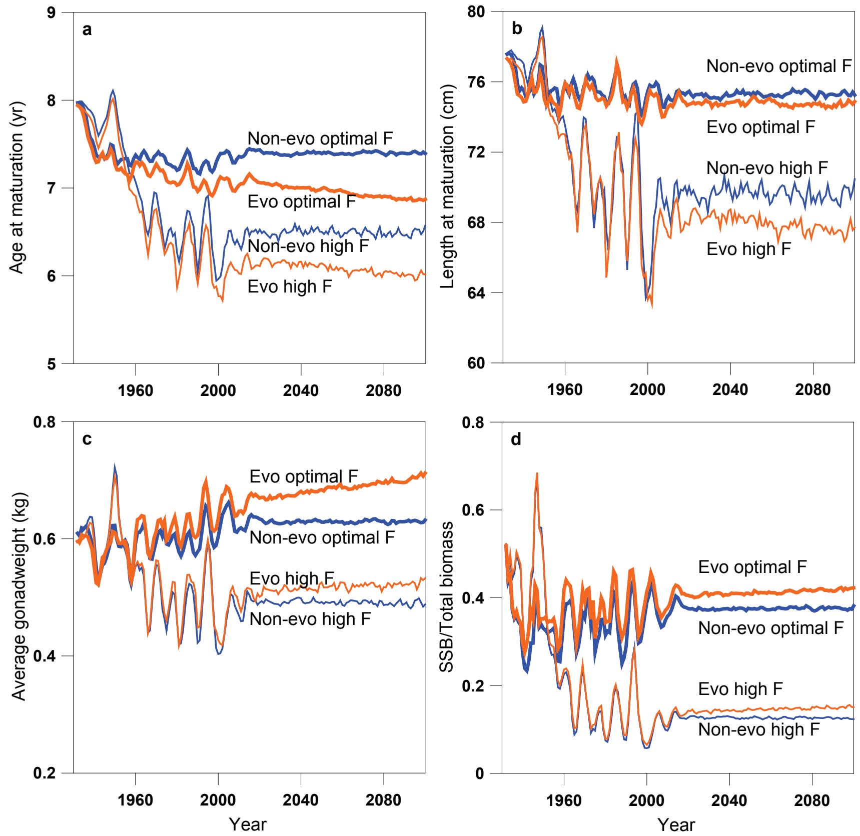

Figure 6.4. Life-history trait changes shown by predicted **a,** mean age and **b,** mean length at maturation, **c,** mean gonad weight and **d**, annual ratio of SSB and total biomass from age 3 for an evolutionary model (grey) and a non-evolutionary model (black). Age at maturation for these scenarios is also shown in Fig. 6.2b and 6.3b. The scenario of high fishing mortality (business as usual) is shown by thin lines, while the optimal scenarios are shown by the thick lines corresponding to Fig. 6.2 and 6.3. High fishing mortality leads to lower **a,** age at maturation and **b,** length at maturation. These effects are stronger for the evolutionary model (grey) compared to the non-evolutionary model (black). **c,** As the other properties, the predicted average gonad weight is higher for low fishing mortality, but the evolutionary model has higher gonad weight than a non-evolutionary model. **d,** Due to age-truncation, the SSB to biomass ratio is lower for both the evolutionary and non-evolutionary model when fishing is high (thin lines). For both fishing mortalities, the evolutionary model (grey), predicts a higher SSB and biomass ratio than the non-evolutionary model (black). Emerging properties are shown from an average of 15 independent model runs.

Third, we do not include genetic correlations between the life-history traits describing maturation tendency, growth capacity and reproductive investment (Eikeset et al., 2010a). Fourth, we limit our analysis to the feeding grounds, while considering the fishing pressure in the spawning ground fishery to be beyond the control of a resource manager. It is an evident extension to allow for optimization of both the Barents Sea and Lofoten islands fisheries at the same time. However, in reality, it is often the case that managers' jurisdiction contains only a certain part of an ecosystem, and analyzing such a case may be even more relevant for policy makers.

6.3 Conclusions

Summing up, our results quantify the costs and benefits of fisheries-induced evolution. Depending on harvesting pressure, it can reduce or increase the economic revenue generated by a fishery. In any case, low fishing mortality is the key for successful management. However, fish stocks are typically far from being managed ecologically optimally. Furthermore, there may very well be cases where the biological circumstances are such that a population requires a different management plan than what a non-evolutionary model would suggest. We also find evidence that fisheries-induced evolution could alter management reference points through its effects on life-history traits and population properties such as SSB. Changes in the stock-recruitment relationship and SSB are crucial for stock assessment models and management decisions. If such changes go undetected, this can result in persistent harvesting above sustainable biomass levels, or harvesting at levels below the economic optimum, which are both undesirable for different reasons. The interplay between ecology, evolution and economics underscores that a Darwinian approach to fisheries science is necessary to enhance the long-term sustainability and profitability of valuable fisheries.

Appendix 6.A: Model description

Our bio-economic model consists of two sub-models: "the biological model" which is a description of the life-cycle of NEA cod, and "the economic model" describing details such as cost and demand for the NEA cod fishery. These two components are linked together through an annual feedback of spawning stock biomass (SSB). This is fed into the economic model where ultimately the total allowable catch is determined by an optimal harvest control rule (HCR). The derived total allowable catch (TAC) feeds back into the biological model and affects the stock size. Each of the sub-models have been specifically estimated and calibrated for the NEA cod fishery using data from the time period 1932-2005 (Table 6.A1). In this chapter we focus on the fishery in the cod's feeding grounds, hence, we keep harvest rates in the spawning grounds at the historical levels between 1932 until 2005, and at a constant rate after 2006. Hence, we consider the spawning ground fishery to be beyond the control of the manager, and consequently, the HCR (Fig. 6.1).

Biological model

The biological model is an individual-based, eco-genetic model similar to the one developed by Dunlop et al. (2009c). It describes an individual's life cycle through annual processes for maturation, somatic growth, reproduction and mortality, and has been calibrated specifically for the NEA cod (Eikeset et al., 2010a). Our main focus is to analyze how changes that occur at the individual level lead to emerging properties at the population level, as for example, changes in SSB and catch. Our individual-based model follows the fate of about 50,000 super-individuals (Huse et al., 2004; Scheffer et al., 1995). All model results, such as SSB and catch, are given for a population that has been scaled up by a factor of 100,000.

Genetic structure

In the initial population, each genetic trait had initial values based on empirical data (Table 6.A1). The genetic traits are assumed to be normally distributed with variances based on the coefficient of genetic variation which has been determined in Eikeset et al. (2010a). Based on quantitative genetics (Mousseau and Roff, 1987), each trait has a heritability that is the ratio between their additive genetic variance (V_G) and phenotypic variance (V_P). The heritability is assumed to be 0.2 for all genetic traits in the initial population (Mousseau and Roff, 1987). Phenotypic variance is the sum of genetic and environmental variance components $(V_P = V_G + V_E)$, and we used this to calculate the environmental variance for each trait in the initial population. This environmental variance was then subsequently kept constant. The phenotypic expression of each individual life-history trait was then determined by drawing randomly from a normal distribution with means equal to the respective genetic trait with the corresponding environmental variances. In our model, we studied the changes in four life-history traits: maturation tendency by probabilistic maturation reaction norm (PMRN) (i) slope s_{P} and (ii) intercept i_{P}; (iii) growth capacity g_{P} and (iv) reproductive investment (gonado-somatic index GSI_{P}). The genetic traits were passed on to offspring by drawing random values from a normal distribution with means equal to the midparental value and variances equal to half of the variance for a given genetic trait in the initial population (Roughgarden, 1979). After the initial year, genetic means, heritabilities and the trait distributions could change freely as determined by the processes of maturation, somatic growth, reproduction, natural mortality and harvesting mortality. These processes were applied sequentially in each year to all individuals.

We made two versions of our model, an evolutionary and a non-evolutionary, each modeling their respective population of individuals: we did this in order to compare a population that has the propensity to evolve, with a population that does not evolve. In the evolving population, the coefficient of genetic variance has been determined by matching trends in age and length at maturation. For the non-evolving population, which is only driven by ecological processes, the coefficient of genetic variance is equal to zero (Eikeset et al., 2010a).

Maturation, growth, reproduction and natural mortality

Each year, the probability that an immature individual will mature depends on the individual's probabilistic maturation reaction norm (Dieckmann and Heino, 2007; Heino et al., 2002b), $p_{\mathrm{m}}(a,l) = [1 + \exp(-(l - l_{\mathrm{P50}}(a)) / \Delta l)]^{-1}$. The length at age $l_{\mathrm{P50}}(a)$ is where the maturation probability $p_{\mathrm{m}}(a,l)$ is 50% with a phenotypic intercept i_{P} and slope s_{P}, $l_{\mathrm{P50}}(a) = i_{\mathrm{P}} + s_{\mathrm{P}} a$. The width of the maturation envelope w_{P} covers the lower envelope bound (25%) to the upper bound (75%) from where the probability to mature at a given age, is determined by the change length, or growth, defined by Δl (Dunlop et al., 2009c; Eikeset et al., 2010a).

Table 6.A1. Parameter values and data sources for the bio-economic model.

Parameters	Value	Source
Biological model		
Initial mean PMRN† slope, $\bar{s}_{\mathrm{G}}$	0.15 cm yr^{-1}	1
Initial mean PMRN† intercept, $\bar{i}_{\mathrm{G}}$	77.4 cm	1
Initial mean PMRN† width, $\bar{w}_{\mathrm{G}}$	12.88 cm	1
Initial mean reproductive investment, $\overline{GSI}_{\mathrm{G}}$	0.15	2
Reproductive investment conversion factor, γ	0.60241	3
Allometric proportionality constant, k	3.2 10^{-6} kg cm^{-j}	4
Allometric exponent, j	3.24	4
Weight-specific oocyte density, D	4.45 10^{6} kg^{-1}	5
Initial mean growth capacity, $\bar{g}_{\mathrm{G}}$	11.08 cm	6
Coefficient of genetic variation in PMRN slope, CV_{s}	10 %	7
Coefficient of genetic variation in PMRN intercept, CV_{i}	2 %	7
Coeff. of genetic variation in reproductive investment, CV_{GSI}	12 %	7
Coefficient of genetic variation in genetic growth, CV_{g}	4 %	7
Immature fishing probability in spawning-ground pre-1932	0.38	8
Immature fishing probability in feeding-ground pre-1932	0.09	8
Minimum-size limit on feeding grounds	45 cm	9
Economic model		
Intercept of the demand function, a	18.88 NOK KG^{-1}	10
Slope of the demand function, b	1.19257 10^{-8} NOK KG^{-2}	10
Stock-output elasticity α	0.58	10
Effort-output elasticity β	0.85	10
Catchability coefficient q	*exp(-7.39) tonnagedays*$^{-1}$	10
Fixed costs per boat c_f	*1.55* 10^{6} *NOK*	10
Variable costs per boat c_v	*131.6NOK tonnagedays*$^{-1}$	10

Sources: 1 = Heino et al. (2002b) and M. Heino (unpublished); 2 =Kjesbu et al. (1998); 3 = Gunderson et al. (1988) and Lester et al. (2004); 4 = survey data from 1999–2007 (IMR, O. R. Kjesbu, pers. comm); 5 = Thorsen and Kjesbu (2001); 6 = survey data from 1932–2005 (M. Heino, unpublished); 7 = model calibration Eikeset et al. (2010a); 8 = Godø (2003) and M. Heino (unpublished); 9 =Bjordal et al. (2004); 10 = Richter et al. (2010); see chapter 5.

For the immature individuals, the body length in a given year depends on the length in the previous year and the growth increment in that year, $l_t = l_{t-1} + g_{\mathrm{P,D},t-1}$. To reflect density-dependence in growth brought about by changes in abundance, and consequently competition and resource availability, we used an estimated relationship of phenotypic growth $g_{\mathrm{P,D},t} = g_{\mathrm{P},t}\exp(bB_{3+,t})$ depending on total stock biomass $B_{3+,t}$ from age 3 and older in year t.This density-dependent growth model was estimated using data from 1978-2009 on annual growth increments and biomass from survey and stock assessment (Eikeset et al., 2010a; ICES, 2008a).

Mature individuals do not only allocate resources growth, but also to reproduction, depending on the reproductive investment. This is given by the phenotypic gonado-somatic index GSI_{P} and a conversion factor, γ, needed to account for the higher energy content of gonadic tissue relative to somatic tissue (Gunderson and Dygert, 1988; Lester et al., 2004). Consequently, the length of a mature individual is given by $l_t = 3(l_{t-1} + g_{\mathrm{P,D},t-1}) / (3 + \gamma GSI_{\mathrm{P},t-1})$. An individual's fecundity f is determined by its length l and gonado-somatic index phenotype GSI_{P} and given by $f = kl^j GSI_{\mathrm{P}} D$, where D is the weight-specific packing density of oocytes (Thorsen and Kjesbu, 2001), and k and j are allometric constants relating body length to body mass. In our model, sex was assigned randomly at birth at a 1:1 primary sex ratio. Atlantic cod are batch spawners and so may mate with several different partners (Kjesbu et al., 1998; McEvoy and McEvoy, 1992). We therefore assumed mating to be random with replacement. The density-dependent newborn mortality was modeled by using an estimated Beverton-Holt stock-recruitment relationship for 3-year olds (Eikeset et al., 2010a). The estimated stock-recruitment relationship depended on the SSB and sea surface temperature to reflect climate impact. The sea surface temperature stretches from the Kola meridian transect (33°50' E, 70°50' N to 72°50' N) and has been shown to be a good indicator for recruitment (Bochkov, 1982; Hjermann et al., 2007; Ottersen et al., 2006; Tereshchenko, 1996). Annual temperature data from 1932-2005 was fed into the modeled stock-recruitment relationship. Prior to 1932 we used the average from 1932-1950 and after 2006-2100 we used the average from 1995-2005. Our model does not include cannibalism despite it has been shown to be important for the natural mortality in the young age-classes (Hjermann et al., 2007; Yaragina et al., 2009). By back-calculating we then found the number of 1-year olds as in a VPA-analysis by assuming an annual total natural mortality rate equal to 0.2 (ICES, 2009a). The individuals can die from natural or fishing mortality. The natural mortality has been summed up by two sources: background natural mortality and mortality from a growth-survival trade-off (Eikeset et al., 2010a). The growth-survival trade-off accounts for the mortality increase when growth increases, as a result of, for example, risky foraging behavior (Biro and Post, 2008). Due to annual spawning migration out of the feeding ground at about ¼ of the year, the harvest probability of mature fish on the feeding grounds was $1-(1-p_0)^{3/4}$, where p_0 is the harvest probability for the immature fish.

Economic model

To calculate the welfare effects of harvesting, we rely on the analysis developed in the previous chapter. We consider the situation where harvesting takes place at minimal costs and only trawlers are used. Therefore, the number of trawlers are given in 5.18, the number of tonnage-days per trawler are given in 5.16. Furthermore the demand function is given $P_t = a - bH_t$, where P_t is the price, H_t is the total allowable catch, and a and b are parameters as estimated in 5.28.

The objective function

Each year, the NEA cod fishery generates economic profits. An economic objective that is often followed in bioeconomic models is to maximize these aggregated profits over T years. The maximum economic yield requires us to maximize $\sum_{t=0}^{T} \Pi_t \left(1/(1+\delta)\right)^t$, where δ is the discount rate. From society's point of view, it is desirable to take into account that consumers and fish processors benefit from buying cheap fish. Therefore, one may also take the consumer surplus into account. Consumer surplus is given by $CS_t = 0.5\left(a - P_t\right)H_t$. Total welfare is given by the sum of profits and consumer surplus. Maximizing total welfare requires us to maximize $\sum_{t=0}^{T}\left(CS_t + \Pi_t\right)\left(1/(1+\delta)\right)^t$.

The harvest control rules

The harvest control rule (HCR) implemented for the NEA cod fishery in 2004 translates precautionary reference points into a management plan (ICES, 2008b, 2009a). Below these reference points the stock is at risk of being harvested unsustainably. The implemented HCR for the NEA cod in 2004 consists of two parameters (Bogstad et al., 2005; Kovalev and Bogstad, 2005): a maximum fishing mortality F_{pa} is followed if the biomass level is above the precautionary biomass level B_{pa}; below this biomass level the fishing mortality decreases linearly to the origin, i.e. fishing mortality is zero at a biomass level of zero.

Here we generalize a HCR with two parameters (Fig. 6.1c) to compare with the implemented management plan. If SSB is between zero and B_{max}, the instantaneous fishing mortality for the given year is $F_{max} SSB / B_{max}$. If SSB is larger than B_{max}, the fishing mortality is equal to the F_{max}. The current HCR is therefore recovered as a special case when $B_{max}=B_{pa}$ and $F_{max}=F_{pa}$. In our model, we vary the parameters in the HCR over a wide range of values, not constraining them to existing precautionary reference points. Due to the complexity of our biological model, we cannot deliver an exact optimum. Instead, we search for the combination of parameter values B_{max} and F_{max} to approximate the maximum of our objective functions (maximize profit and maximize welfare). The grid size for the parameters gave a grid of 4141 different HCRs. B_{max} from 0-800 thousand tonnes in steps of 20, and instantaneous fishing mortality F_{max} from 0.2-1.2 yr^{-1} in steps of 0.01 yr^{-1}. Our model is individual-based, and for some of these HCRs, fishing could make the abundance very low. To avoid stochastic effects at low abundances, we therefore set a threshold below which the population was classified as extinct (at 20 modeled mature "super-individuals") (Eikeset et al., 2010a; Scheffer et al., 1995). In total the computation time was 2 days for 7 independent model runs at a cluster with 5776 CPUs at the Research Computing Services at the University of Oslo.

Historic fishing pressure

The harvest pressure in the feeding ground increased steadily from the 1930s to the middle of the 1960s and remained high until mid- 2000. In one of the fishing scenarios, we assume a fixed fishing mortality in the feeding ground (0.68 $year^{-1}$) being maintained into the future. This is an average of historic fishing mortality between 1946-2005 and is higher than what is suggested to be precautionary for the NEA cod (0.4 $year^{-1}$) (ICES, 2009a); we coin this scenario "business as usual".

Appendix 6.B: Objective functions and discount rates

Maximizing welfare

In the main results we focus on the objective maximizing welfare (Fig. 6.2-6.4). The fishing scenarios are i) observed fishing mortality between 1932-2006, then from 2006 onwards "business as usual" with annual fishing mortality set to 0.68 (Fig. 6.2a-c), ii) observed fishing mortality between 1932 and 2006 followed by optimal fishing mortality from 2006 (Fig. 6.2d-f), and iii) optimal fishing mortality from 1932-2100 (Fig. 6.3).

For each scenario, we compare the results from the evolutionary and non-evolutionary model. Table 6.A2 and 6.A3 report the parameters $B_{max,}$ and F_{max} for the optimal HCR. In the "business as usual" case, these parameters are omitted because fishing mortality is not derived from an HCR. Furthermore, we analyze the emerging harvesting properties, given in averages over the course of fishing (Table 6.A2-6.A7).

Table 6.A2. Mean values corresponding to Fig. 6.2. Observed fishing mortality from 1932-2005, followed by a constant rate from 2006-2100. In the optimal harvest control rule (HCR), total welfare has been maximized from 2006-2100: for this case, values shown are averages for 2006-2100. Averages of fishing mortality (F), catch (TAC), spawning stock biomass (SSB), profit, and welfare are shown with standard deviation in parentheses.

Model	F_{max}	B_{max}	F	TAC	SSB	Profit	Welfare
Historic							
Evolutionary	-	-	0.68	360 (95)	267 (365)	1755 (519)	2705 (874)
Non-evolutionary	-	-	0.68	370 (93)	260 (356)	1821 (505)	2808 (836)
Optimal HCR							
Evolutionary	0.31	440	0.3	474 (76)	938 (155)	3149 (441)	4837 (767)
Non-evolutionary	0.32	580	0.31	446 (71)	798 (113)	3099 (446)	4565 (716)

Units: F_{max} (inst. rate); B_{max}, TAC, SSB (1000 tonnes); Profit and total welfare (million NOK).

Business as usual and optimal management from 2006

Table 6.A2 shows harvesting properties for a scenario of historic fishing mortality, followed by "business as usual", and an optimal HCR from 2006 on. In the evolutionary model, lower spawning stock biomass (SSB), total allowable catch (TAC) and economic yield can be observed, though these differences are insignificant. If management is optimal, fishing mortality is much lower than what has been done historically (average fishing mortality from 2006-2100 is 0.31 $year^{-1}$ in the non-evolutionary model and 0.30 $year^{-1}$ in the evolutionary model). The evolutionary model delivers slightly higher SSB, TAC and welfare. Figure 6.A2 shows that in the "business as

usual" case, total biomass and TAC tend to be slightly lower in the evolutionary model, though not significantly so. When the optimal HCR is used, total biomass and TAC are somewhat higher in the evolutionary model.

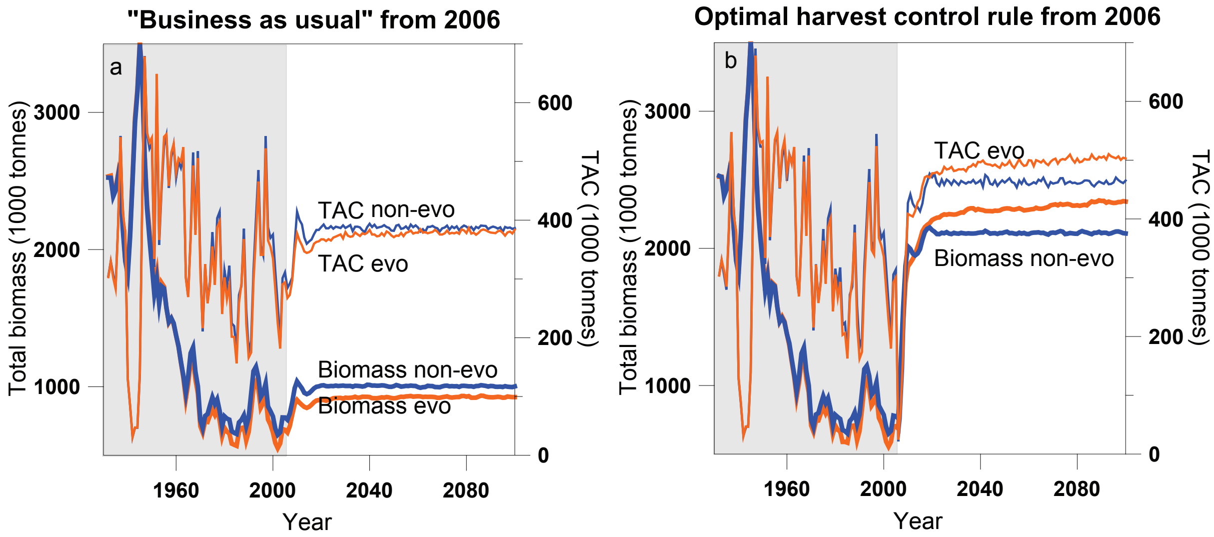

Figure 6.A2. Supplementary information for Fig. 6.2. Emerging total biomass from age of 3 and older (thick line) and total allowable catch (TAC, thin line), compared for the evolutionary model (grey) and the non-evolutionary model (black). **a,** Historic fishing mortality from 1932-2005 (grey area), continued by a high constant harvest rate from 2006-2100 (white area). Total biomass and TAC tend to be slightly lower in the evolutionary model **b,** When the optimal HCR is used, total biomass and TAC is higher in the evolutionary model. Predictions are averages over 15 independent model runs.

Optimal management from 1932

For the optimal HCR derived over the time-period 1932-2100, the evolutionary and non-evolutionary models have the same starting point in 1932. Hence, we can compare these two models and ask how optimal management differs in an evolutionary model compared to a non-evolutionary one. Therefore, we derive an "ecologically optimal HCR" from a non-evolutionary model and an "evolutionarily optimal HCR" from an evolutionary model. We find that the maximum fishing mortality F_{max} is marginally higher in the evolutionary model compared to the non-evolutionary models (0.33 vs. 0.32 year^{-1}, Table 6.A3). The parameter B_{max} differs, but note that the SSB almost never drops below this precautionary buffer (not shown). Therefore, the mean emerging fishing mortality from 1932-2100 is 0.32 year^{-1} for both the non-evolutionary and the evolutionary model. Note that the evolutionary model allows for a slightly higher TAC, while maintaining higher SSB. As a result, total welfare is higher for the evolutionary model. Profits, however, are not significantly higher in the evolutionary model. This happens because higher catches depress the price (see section below). Fig. 6.A3 shows that total biomass from age 3 and older with corresponding catch are slightly higher in the evolutionary model compared to the non-evolutionary one.

Table 6.A3. Mean values corresponding to Fig. 6.3. Optimal harvest control rule (HCR) for maximizing welfare from 1932-2100. Averages of fishing mortality (F), catch (TAC), spawning stock biomass (SSB), profit, and welfare are shown with standard deviation in parentheses.

Model	F_{max}	B_{max}	F	TAC	SSB	Profit	Welfare
Evolutionary	0.33	380	0.32	465 (72)	826 (138)	2970 (487)	4631 (755)
Non-evolutionary	0.32	120	0.32	439 (48)	702 (123)	2927 (427)	4372 (551)
Evolutionary [¥]			0.33	468 (60)	804 (169)	2977 (453)	4635 (678)

Units: F_{max} (inst. rate); B_{max}, TAC, SSB (1000 tonnes); Profit and total welfare (million NOK).

[¥] Evolutionary model with HCR derived from a non-evolutionary model; using an "ecologically optimal HCR".

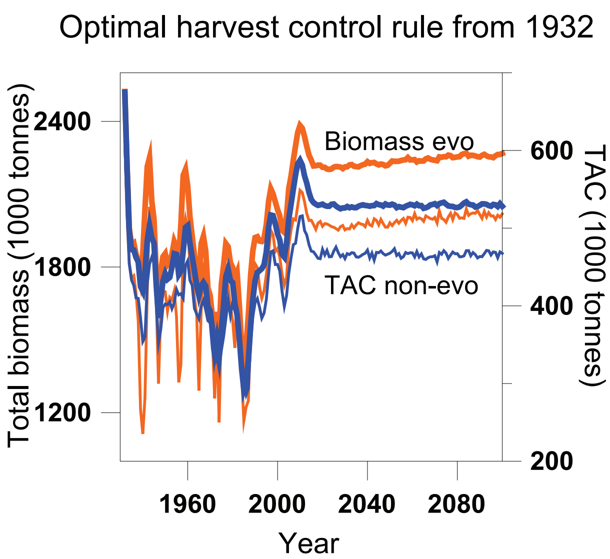

Figure 6.A3. Supplementary information for Fig. 6.3. Emerging mean total biomass from age 3 and older (thick line) and total allowable catch (TAC, thin line). Optimal harvest control rule (HCR) for maximizing welfare from 1932-2100. Total biomass and catch are higher in the evolutionary model (grey), compared to the non-evolutionary model (black). Note that the decline in total biomass prior to 2006 is also driven by the historic and variable spawning ground fishing mortality. Predictions are averages over 15 independent model runs.

In the evolutionary model, total biomass tends to increase over time (though not significantly), which could be a symptom of changing stock-properties due to genetic change. Furthermore, we investigate how costly it is to overlook evolutionary effects. If a manager is unaware that genetic change occurs, how large will the welfare losses be? To answer this question we use an "ecologically optimal HCR" derived in a non-evolutionary model in an evolutionary model. Given that the non-evolutionary and evolutionary models recommended very similar HCRs, the welfare losses are negligible. An evolutionary model that is managed with an "ecologically optimal HCR" still outperforms the non-evolutionary model. Using an "evolutionary optimal HCR" in the evolutionary model does not increase any of the emerging harvesting properties significantly (Table 6.A3).

Changing the objectives to maximizing profit

We investigated the characteristics from alternative HCRs for different objectives and discount rates. The objective maximizing profit from 2006-2100 leads to slightly lower mean fishing mortality for the evolutionary and the non-evolutionary model (0.29 year^{-1}) than maximizing welfare (0.32 year^{-1}). Overall, changing objectives did not alter our results concerning the ranking of the evolutionary and the non-evolutionary model: the evolutionary model allows for a slightly higher TAC, while it tends to maintain higher SSB. Therefore, profits and total welfare are still higher for the evolutionary model, though not very significantly (Table 6.A4 and 6.A5).

Table 6.A4. Optimal harvest control rule (HCR) for maximizing profits from 2006-2100. Values shown are averages for 2006-2100 for fishing mortality (F), catch (TAC), spawning stock biomass (SSB), profit, and welfare with standard deviation in parentheses.

Model	F_{max}	B_{max}	F	TAC	SSB	Profit	Welfare
Evolutionary	0.29	140	0.29	465 (67)	1002 (197)	3149 (384)	4795 (717)
Non-evolutionary	0.29	280	0.29	438 (62)	872 (140)	3112 (398)	4549 (655)

Units: F_{max} (inst. rate); B_{max}, TAC, SSB (1000 tonnes); Profit and total welfare (million NOK).

Table 6.A5. Optimal harvest control rule (HCR) for maximizing profits from 1932-2100. Values shown are averages for 1932-2100 for fishing mortality (F), catch (TAC), spawning stock biomass (SSB), profit, and welfare with standard deviation in parentheses.

Model	F_{max}	B_{max}	F	TAC	SSB	Profit	Welfare
Evolutionary	0.31	60	0.31	463 (60)	872 (182)	2983 (455)	4637 (685)
Non-evolutionary	0.29	200	0.29	427 (46)	826 (141)	2940 (438)	4359 (561)
Evolutionary¥				455 (59)	944 (199)	2982 (463)	4616 (695)

Units: F_{max} (inst. rate); B_{max}, TAC, SSB (1000 tonnes); Profit and total welfare (million NOK).

¥ Evolutionary model with HCR derived from a non-evolutionary model; using "ecologically optimal HCR".

Changing discount rates for maximizing welfare

As expected, higher discount rates (2 and 4%) lead to slightly higher fishing mortality. Discounting does, however, not influence optimal policy substantially. This is in line with earlier findings from chapter 5 and Sandal and Steinshamn (1997). Discounting did also not change the relative difference between results obtained from the evolutionary model compared to the non-evolutionary model; see tables 6.A6 and 6.A7.

Table 6.A6. Optimal harvest control rule (HCR) for maximizing welfare from 2006-2100 with different discount rates (dr), 0, 2 and 4%. Values shown are averages for 2006-2100 for fishing mortality (F), catch (TAC), spawning stock biomass (SSB), profit, and welfare with standard deviation in parentheses.

Model	dr (%)	F_{max}	B_{max}	F	TAC	SSB	Profit	Welfare
Evolutionary	0	0.31	440	0.3	474 (76)	938 (155)	3149 (441)	4837 (767)
	2	0.41	760	0.32	479 (81)	852 (126)	3128 (464)	4822 (796)
	4	0.35	440	0.34	483 (75)	790 (121)	3121 (427)	4811 (744)
Non-evolutionary	0	0.32	580	0.31	446 (71)	798 (113)	3099 (446)	4565 (716)
	2	0.33	460	0.32	450 (67)	743 (104)	3092 (418)	4560 (678)
	4	0.33	460	0.32	450 (67)	743 (104)	3092 (418)	4560 (678)

Units: F_{max} (inst. rate); B_{max}, TAC, SSB (1000 tonnes); Profit and total welfare (million NOK).

Table 6.A7. Optimal harvest control rule (HCR) for maximizing welfare from 1932-2100 for different discount rates (dr), 0, 2 and 4%. Values shown are averages for 1932-2100 for fishing mortality (F), catch (TAC), spawning stock biomass (SSB), profit, and welfare with standard deviation in parentheses.

Model	dr (%)	F_{max}	B_{max}	F	TAC	SSB	Profit	Welfare
Evolutionary	0	0.33	380	0.32	465 (72)	826 (138)	2970 (487)	4631 (755)
	2	0.33	20	0.33	468 (60)	802 (170)	2977 (455)	4634 (682)
	4	0.35	120	0.35	470 (60)	733 (155)	2959 (450)	4604 (668)
Non-evolutionary	0	0.32	120	0.32	439 (48)	702 (123)	2927 (427)	4372 (551)
	2	0.34	120	0.34	442 (48)	670 (121)	2916 (424)	4363 (549)
	4	0.36	180	0.36	445 (51)	621 (114)	2896 (425)	4348 (554)

Units: F_{max} (inst. rate); B_{max}, TAC, SSB (1000 tonnes); Profit and total welfare (million NOK)

7

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

Abstract

In this chapter we discuss the potential failure of simple management models. Analyzing 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.

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7.1 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 exploitation of marine resources – is a widespread observable fact (Hilborn et al., 2003; Jackson et al., 2001; 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). 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 analyzing 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) the resource system (e.g. a coastal fishery), (ii) the resource units (e.g. fish stock), (iii) the users (e.g. fishermen), and (iv) the governance system (e.g. the specific laws and social norms in place); see Figure 7.1. 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 Figure 7.1). It is crucial to recognize the occurrence of feedbacks and, to the extent possible, to identify particular feedback structures within and between the specific systems (Berkes et al., 2003; Berkes et al., 1998). If management strategies are based on results derived from analyzing one of the subsystems in isolation, the outcome may be very different from 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 backdoor. By this, we mean that models that focus 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 as follows: First, in sections 7.2 and 7.3, we highlight the fundamental complexity of the social and natural environment relevant for fisheries management. In section 7.4, 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. Finally, in section 7.5, we summarize and 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.

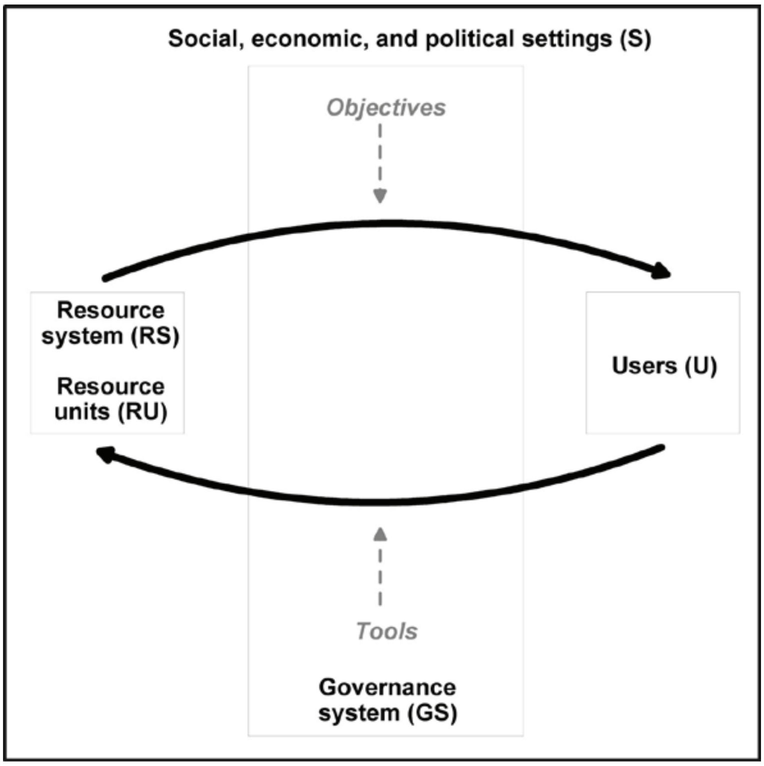


Figure 7.1. An adaptation of Ostrom's (2009) framework of core subsystems for analyzing social-ecological systems in our marine fisheries context. Here, we emphasize the feedback loop between the resource system and resource units in their interactions with the users (black arrows) in the form of objectives, and the resulting management tools applied to the resource system and/or units (grey arrows).

7.2 Stakeholder participation and the social environment

A stakeholder can be defined as any member of society who has direct (primary stakeholders) or indirect (secondary stakeholders) interests, or stakes, in 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 dialogue process, objectives can be cognitively broken down and made more compatible (Follett, 1955). For example, 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 much more resources – financial funds as well as knowledge – than others (Esteban and Ray, 2006; Mikalsen and Jentoft, 2008). In many cases it is impossible to distinguish an active debate among stakeholders from lobbyism. Often stakeholders are willing to spend a substantial amount of money and time on influencing political decisions. This form of "rent seeking" activity (Bergland et al., 2002; Johnson and Libecap, 1982; Krueger, 1974) is 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 decision-making process. 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).

Fisheries managers, on their side, should also be accountable and bear the full responsibility of their decisions (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 fulfill 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 amounts of uncertainty (especially in regard to tradeoffs 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 dialogue 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, 2008; Hatchard, 2005; Prince, 2003); 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 (Bowles, 2008; Deci et al., 1999, 2001; Frey and Jegen, 2001; Frey et al., 1996; Frey and Stutzer, 2006; Gintis et al., 2005; Vollan, 2008), see also chapter 2 of this thesis. 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 (unintended mortality of non-targeted organisms caught in fishing gear). He may also 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. 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 fears to be fined. Once the authorities start monitoring an individual frequently he may respond 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 – see chapter 2 of this thesis. 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; Bowles, 2008; Hatcher et al., 2000; Sutinen and Kuperan, 1999) 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; van Soest and Vyrastekova, 2008)

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., 1994; Ostrom et al., 1992; Vyrastekova and van Soest, 2003). However, 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.

7.3 Uncertainty and the biological environment

Worldwide marine fish stocks are declining (FAO, 2008; Worm et al., 2006; Worm et al., 2009; Worm and Myers, 2004) leading to changes in ecosystem structure and functioning. After over-exploitation 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; Pauly et al., 2002; Pauly and Palomares, 2005). Habitat loss from trawl (the fishing net usually towed behind a fishing vessel) activity and bycatch 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 et al., 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. Exploitation, for example, 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% larger than what occurs naturally (Darimont et al., 2009). In commercial fish populations, changes in life-history traits, exemplified by maturation at earlier ages and smaller size, is higher 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 (Dieckmann and Heino, 2007; Dunlop et al., 2009c; Heino, 1998; Heino et al., 2002a, 2002b; Marshall and McAdam, 2007; Olsen et al., 2004; Stenseth and Dunlop, 2009), see also chapter 6 of this thesis. 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 (Conover et al., 2009; Enberg et al., 2009; Law and Grey, 1989). The extent to which fisheries-induced evolution occurs and how important it is compared to other factors is being debated (Andersen and Brander, 2009a; Browman et al., 2008; Conover and Munch, 2007; Hilborn, 2006; Jørgensen et al., 2007). 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 through the backdoor.

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 to analyze data for many fish stocks (Ånes et al., 2007; Bogaards et al., 2009; Eikeset et al., 2010b; Lindegren et al., 2009; Millar, 2002; Millar and Methot, 2002; Swain et al., 2009).

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 changes in ecosystems, such as an 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.

7.4 Fisheries management

Many different tools for fisheries 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 (Caddy and Seijo, 2005; Dankel et al., 2008; Degnbol et al., 2006; Grafton et al., 2000; Jentoft, 2006; 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.

7.4.1 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 a 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.

7.4.1.1 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 in practice.

There is also the danger that unintended consequences of economic incentives will sneak in through the backdoor. 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.

7.4.1.2 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 dialogue 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 is 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 (Armitage et al., 2009; Chuenpagdee and Jentoft, 2007; Jentoft, 2000a, 2000b, 2005; Jentoft, 2006; Jentoft and Chuenpagdee, 2009; Jentoft and Mikalsen, 2004; Pinkerton and John, 2008).

7.4.1.3 Community-based management

Community-based management takes 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 (Baland and Platteau, 1996; McCay and Acheson, 1987; Ostrom, 1990; Ostrom et al., 2002). Chapters 3 and 4 of this thesis develop theoretical models that shed light on the mechanisms that may facilitate – but also threaten – successful community-based management. 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, 2000a), 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 in 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; see 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 to lower harvesting costs (Swallow, 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 resource 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 (Agrawal, 2001; Baland and Platteau, 1996; Ostrom, 1990; Ostrom et al., 2002; Ostrom et al., 2007; Schlager et al., 1994; Schlager and Ostrom, 1992, 1999), but when fish species are highly migratory and foreign fishermen are difficult to exclude, the prospects for community governance are rather bleak.

7.4.2 Management tools

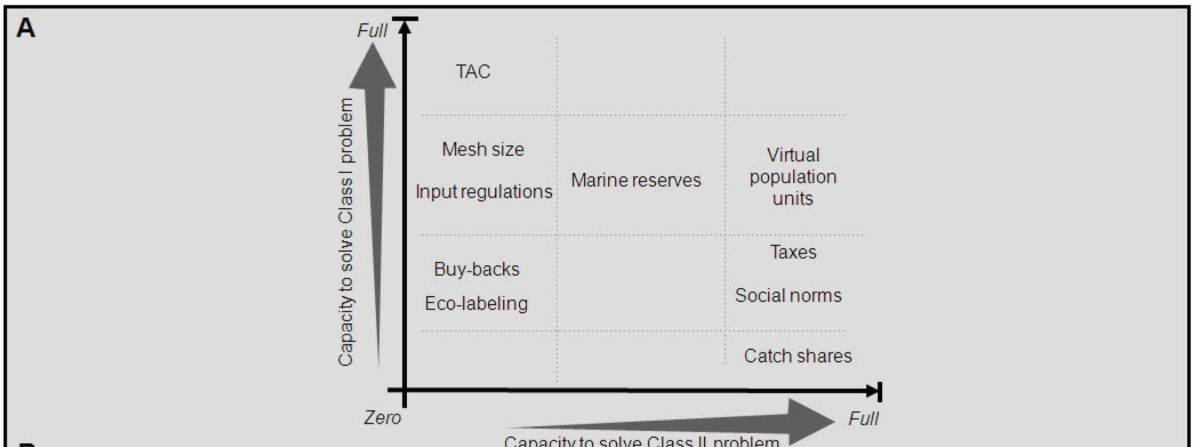
Fisheries management can rely on a variety of tools (see Figure 7.2 for a graphical and tabular exposition of the tools we discuss). Good overviews can be found in Rettig (1995), Kahn (2005, Ch.10), and van Kooten and Bulte (2000. pp. 94). We will distinguish between tools that are based on a command 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 panel A in Figure 7.2). 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 (panel B in Figure 7.2)? 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 sections.

7.4.2.1 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-centered instrument is a cap on the **total allowable catch (TAC)**. 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 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 1980's, when the year's catch was taken in three to five 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, Pauly et al. 2002, Pauly and Palomares 2005).



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 from incumbent fishermen
Buybacks		Very costly and often ineffective		In anticipation of a buyback program users may increase capacity, leading to a situation that is worse than open access
Eco-labeling	Creates 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 depend 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

Figure 7.2. 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.

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 (Arnason et al., 2004; Sandal and Steinshamn, 1997), see also chapters 5 and 6 of this thesis. 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 et al., 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" is also 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 in the long run profits could be more than doubled, simply by changing the mesh size (Diekert et al., 2010a).

Another output-centered management approach that has sparked considerable interest 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 (Lubchenco et al., 2003; Sumaila and Alder, 2001), particularly in warm water ecosystems (shallow water coral reefs) compared to 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 over-ride 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 instruments; 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, p. 855–56) "…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).

7.4.2.2 Tools based on financial incentives

Taxes increase the cost of catching a fish and essentially affect 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 (Brown, 2000; Johnson and Libecap, 1982; Scott, 1979). 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 draw on financial incentives are **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.

7.4.2.3 Tradable permits

Tradable permits are a special case of market instruments. Individuals are endowed with harvesting rights, such as a catch quota, which 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. Firstly, most people are very sensitive to financial incentives, making market instruments very effective. Secondly, 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 (Grafton et al., 2006; Hannesson, 2004). 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 (Grafton, 2000; Wilson, 1982). In fact, property rights have several relevant dimensions, as pointed out by Schlager and Ostrom (1992). 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 rents. 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 fulfils 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 **individual transferable quotas** (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 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 to 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; see chapter 5.

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. ITQs 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 11000 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 heterogonous 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 fishery collapses, 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 and chapter 2, 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. ITQs 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., 2010b).

7.5 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 Figure 7.2). 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 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 decision, 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 and 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 impossible 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 backdoor. 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 of 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.

8

Discussion

8.1 Introduction

This research was motivated by three observations. First, worldwide several renewable natural resources, as for example fish stocks, grazing lands or forests are severely depleted, while some are in relatively good shape. Second, some of these resources are conserved *because* of active government intervention, some are conserved *in absence* of any government intervention, while other resources are depleted *in spite* of active government intervention. Third, sometimes government-intervention can be counter-effective and speeding-up, rather than slowing down resource degradation (Cárdenas et al., 2000; Frey et al., 1996; Gintis et al., 2005; Vollan, 2008). The explanation for this ambiguity is that most decisions by resource users do not only depend on formal institutions (such as governmental law and the corresponding fines), but also on informal institutions (such as social norms or moral preferences). The main message of this thesis is that acknowledging complexity in resource management is important. This holds for social complexity (chapters 2-5), but also biological complexity (chapter 6), or both (chapter 7). Yet, most economic models rely on simple assumptions. For example, most economic models describe human behavior with the stylized avatar of the homo economicus. These models assume that each individual has all available information that is necessary to make an optimal decision and consequently maximizes her expected utility. Furthermore, it is assumed that the collective action of many individuals will lead to a stable situation – a steady state – that can be identified. Additionally, it is usually assumed that individuals are solely motivated by financial motives. Empirical evidence regularly shows that these assumptions are not met in practice; see for instance Henrich et al. (2001).

This discussion analyzes how problematic these shortcomings are and how they can be remedied. Moreover, these insights will be linked to the models used in this thesis. First, it will be discussed whether human behavior can be accurately described by optimization models. Second, it is addressed whether the objective function of homo economicus is defined too narrowly – as it includes only self-regarding motives.[29] In the third sub-section, several modeling choices that were made in this thesis are critically reflected. Furthermore, it is discussed why fisheries economics had so little impact on actual management practices in the real world. The discussion ends with some concluding remarks.

8.2 Model techniques and the problems with utility maximization

The critic that utility maximization is not always well suited to describe human behavior goes back at least to Herbert Simon (1956).[30] It is well known that individuals often do not have the capacities to make optimal choices, for example because of cognitive, informational, or time constraints. If one wants to take these ideas into consideration, it is appropriate to use a model that takes learning processes into account. Evolutionary game theory is a framework that can be used to model a learning process. The replicator equation (presented in chapter 1) assumes that the more successful strategy grows over time and can therefore represent a process of learning, imitation, trial-and error, but also just a mental evaluation of different strategies of agents unaware of each other. Not used in this thesis is the framework of Bayesian updating, which is commonly used for modeling learning processes, see for example Charness and Levin (2005) and Brekke et al. (2009).

Often overlooked is the fact that even if humans were able to maximize utility, this would not necessarily converge to a unique equilibrium at the group level (Elster, 2009; Green and Shapiro, 1996).[31] This is a valid concern, but it can be remedied fairly easily. After all, the notion of equilibrium implies that there must be a dynamic process, which can in principle be analyzed and understood.

Finally, rational actor models always maximize utility in a static environment. In reality, the environment is not constant but changes in response to actions by all individuals. Ecosystem changes and government regulations have both in common that they typically i) respond slowly (or delayed) and ii) respond to the actions of all agents, while individual actions have a negligible impact. In these cases, rational actor models are not very useful, as they cannot capture frequency-dependence, where the payoff of one individual depends on the actions of all other agents –

[29] Sometimes the additional assumption is imposed that individuals are only motivated by financial incentives.
[30] An example in this thesis with utility maximization is equation 3.7.
[31] This was for example assumed in equations 3.11 and 4.32.

and both the payoff and frequency of strategies endogenously coevolve. In such a case it is more appropriate to use a modeling framework that can analyze feedbacks between individual behavior and the environment affecting all individuals as a coevolutionary process. Adaptive dynamics used in chapter 4 of this thesis is such a framework. In adaptive dynamics, decisions are optimal, but the environment is not static, but changing in response to individual actions. This implies that individuals exhibit optimizing behavior, but due to repercussions from the environment, optimal actions are not constant but change endogenously.

The evolutionary models described above are all analytically tractable. In addition, agent-based models (ABMs)[32] are welcome alternatives because they allow for almost endless complexity and robustness checks. Since ABMs are not based on equations, but on rules, they are not analytically tractable. The research pursued in this thesis suggests that ABMs and equation-based models are highly complementary. Analytical work gives valuable insights to understand the basic properties of a system, while sensitivity, stability, robustness, and realistic extensions can be better analyzed with an ABM. It is therefore the combination of the two that is especially appealing, as used in chapter 4. The analytical work in this chapter raised some typical questions: how important are the functional forms that were used? What happens if individuals cannot monitor each other perfectly? How sensitive are the results if individuals make mistakes? All of these questions could be answered by pointing at the agent-based simulations. Likewise, skepticism towards the results coming out of the ABM due to its "black box" character and its complexity vanished, since the analytical work told us what drives the results.

A second example where analytical work and individual-based simulations complement each other is given in chapters 5 and 6 of this thesis. While chapter 5 determines an analytical harvest control rule that provides valuable insights about the relative importance of the demand effect and the fleet structure, the biological model in that chapter is very simple. With these insights from chapter 5, we have developed a highly complex bio-economic model in chapter 6 that relied on an individual-based biological model and lacked analytical rigor to focus on very complex biological questions.

Therefore, the combination of equation-based models and individual-based models serves a dual purpose. First, the agent-based models validate the robustness and stability of the analytical work. Second, the analytical work helps explaining what drives the results obtained from an ABM. Therefore, equation-based models with their analytical rigor, and ABMs with their

[32] also referred to as individual-based models (IBMs)

bells and whistles cover each other's back and help analyzing complex problems that have high real-world relevance.

8.3 (Lacking) moral motivation in economic models

Most economic models assume that humans only care about themselves – referred to as homo economicus. This thesis has argued several times that this is not a very good description of human behavior, because the objectives of most individuals are much broader. These other objectives, such as pro-social considerations are not just random deviations from homo economicus, but they tend to be very consistent over time and consistent between other individuals – even though they may differ between individuals from a different cultural background (Boyd and Richerson, 2005; Gintis et al., 2005; Henrich and Henrich, 2007). One could modify the objective function to facilitate a closer match with behavior of socially responsible citizens, which Nyborg (2000) aptly called homo politicus. In a similar vein, one could include pro-social considerations, such as equity preferences (Fehr and Fischbacher, 2002). These suggestions do not question the rational actor model as such, but only their restrictive objective function – which could be changed easily. For example, in chapter 4, our model included a utility component that was affected by approval or disapproval of other agents. It is of course surprising that empirical evidence strongly suggests that humans are often motivated by non-financial incentives and financial incentives can even undermine motivation – as discussed extensively in chapter 2 – while most economic models assume that individuals are *only* motivated by financial incentives. For non-economists it is often hard to grasp that empirical evidence is not necessary to justify using models of homo economicus – while empirical evidence is sometimes considered to be insufficient to justify a deviation from homo economicus. Therefore, it is sometimes argued that these modeling choices are made for ideological reasons, and economists have a hidden agenda of promoting the role model of self-interested behavior (Foley, 2004). Be that as it may, two reasons are usually mentioned why we should continue using models of homo economicus. First, even if it is true that humans have other-regarding and non-financial preferences, we know very little about how these preferences are shaped. Since we know so little about them, preferences become a tautology: agents act in a certain way because they prefer to do so. Second, the average person is probably equipped with a moral compass, but the models we are using are not meant to describe average behavior. Instead, we are using them for advising policy makers. These policies should be designed for people who are self-interested, even if they do not represent the majority of our population. Otherwise these individuals would take advantage of the others.

The first point has been addressed by Stigler and Becker (1977), according to whom economists should just assume preferences to be stable, rather than spending effort on analyzing preferences with their "endless degrees of freedom. Further, Stigler and Becker (1977, p.89) argue that

> *[w]e have partly translated 'unstable tastes' into variables in the household production functions for commodities. The great advantage, however, of relying only on changes in the arguments entering household production functions is that all changes in behavior are explained by changes in prices and incomes, precisely the variables that organize and give power to economic analysis.*

This argument may be criticized for the following reason. If a process is perfectly understood, using a model is not necessary. But if a process is not fully understood, and we only have a vague idea – or several hypotheses – that can explain the process, then models are really useful. Therefore, suggesting to model something that we know (how do changes in prices affect behavior), rather than attempt to model what we do not know (how do preferences affect behavior), is the exact opposite of what scientific work requires. However, in moving forward one must also be careful and compare, test, and validate different models and their assumptions. This is not always easy or possible, especially if the models are very theoretical, such as in chapters 3 and 4 of this thesis. In that case, it is desirable to build models that are very flexible and allow for example for pro-social and anti-social punishment (as in chapter 4). Even if Stigler and Becker do not have a convincing argument, the idea that preferences are stable over time is worth thinking about. Admittedly, empirical evidence points into a different direction. Voors et al. (2010), for example, have shown that individuals that have been exposed to conflicts, exhibit more altruistic, but also more risky behavior. This does not necessarily imply a preference change, as decisions may be context-dependent. This point has been discussed in chapter 2; see for the situation where it is not clear whether a preference or the perceived context has changed Figure 2.2. In biology, organisms frequently make phenotypic expressions, such as growth, dependent on cues from the environment. Intuitively, it seems that humans with more cognitive capacities make their behavior even more contingent on the environmental context. To my knowledge, no study has so far disentangled preference changes from context-dependency.

The second point, stating that models of homo economicus are useful because we need to design policies for selfish individuals rather than for committed citizens (homo politicus) seems to be a valid justification. Since we cannot rely on moral virtues and social norms only, often laws are required to prevent selfish individuals to take advantage of other members of society. Designing policies in a way to constrain homo economicus in her activities is crucial and should always be an essential part of economic policy analysis. Having said that, the costs of doing so may be very high. The nature of this dilemma is reflected in Horace's question "quid leges sine moribus vanae proficiunt?" – what good are laws when there are no morals? Crafting laws that are tailor-made for homo economicus, while the far majority of the population consists of home politicus can be problematic, or at least inconsistent. Again this insight is far from new and a particularly nice illustration has been given by David Hume (1826):

> *Political writers have established it as a maxim, that, in contriving any system of government, and fixing the several checks and controls of the constitution, every man ought to be supposed a knave, and to have no other end, in all his actions, than private interest. By this interest we must govern him, and, by means of it, make him, notwithstanding his insatiable avarice and ambition, cooperate to public good […]. It is, therefore, a just political maxim, that every man must be supposed a knave; though, at the same time, it appears somewhat strange, that a maxim should be true in politics which is false in fact.*

Even if designing public policy for homo economicus is a perfectly legitimate political choice, it is important to understand that it comes at a cost. Chapters 2 and 7 of this thesis have shown that there are many cases where laws that are tailor-made for homo economicus cause a collateral damage on homo politicus by undermining existing moral values and social citizenship. Consequently, this collateral damage leads to an outcome that it suboptimal – an efficiency loss, which can be prohibitively high. Note that economic models that rely only on homo economicus are not able to assess this efficiency loss – in spite of the fact that efficiency is usually highly emphasized in assessing these models. Typically, the attractiveness of several policy tools is *only* evaluated in terms of economic efficiency (in a world of homo economicus), while the efficiency effects in reality may be fundamentally different. Efficiency calculations that rely only on homo economicus may still serve as important baseline cases, but we should be aware that any policy recommendations coming from them may be inaccurate, or even harmful.

To sum up, the main challenge for policy makers is to design policies that are fair and successfully restrict homo economicus in her incredibly high plasticity, while keeping the efficiency loss coming from alienating homo politicus limited. The challenge for the economic science is to build flexible models that can provide valuable advice to achieve this.

8.4 Modeling assumptions and roads not taken

The previous sub-section has dealt with the necessity to build models that go beyond the assumption that individuals are only self-regarding and motivated by financial motives. In this subsection, I will briefly reflect on several modeling choices that have been made in this thesis, and point out several alternatives.

In chapter 3 good spirits were assumed to be contagious. This implies that individuals can be convinced at no cost to become cooperators. In theoretical models of cooperation (and Hollywood movies), the good guys always need some form of obstacle to make the story interesting. In chapter 3, a strong temptation to defect formed that counter-force. In chapter 4, it was assumed that social sanctions are costly. Again, this assumption was made to make the scientific problem more interesting, not so much because of empirical considerations. The cost of sanctions is one of the ingredients in models of cooperation where the bridge to the real-world is not very clear. While severe penalties or excluding individuals may bear some costs, simply disapproving of someone's behavior seems to be very cheap and effective (Masclet et al., 2003). Some studies even show that sanctioning someone is making us feel good (de Quervain et al., 2004). This seems to be even more the case if social sanctions are based on "sweet revenge" (Knutson, 2004), or if someone, who sanctions a bad guy is rewarded, esteemed, or cherished by other individuals (Kendal et al., 2006). On the other hand, anecdotal evidence abounds where sanctioning is rather stressful, and hence costly, especially if the other person retaliates.

From a theoretical perspective, costly punishment is a puzzle in itself, as it classifies as a (second-order) public good. Cooperative individuals who do not discipline their peers are better off than individuals who cooperate *and* sanction defectors. Additionally, Dreber et al. (2008) have pointed out that punishment may escalate conflicts, rather than moderate them. As a result, the possibility to punish may actually lower average payoff. Chapter 4 touches upon this issue, by finding that a sanctioning mechanism evolves towards the social optimum if time lags are such that individuals adjust their behavior quickly. This mechanism therefore can solve the second-order free rider problem, and shows that sanctioning can be socially optimal after all.

In the models used in chapter 3 and 4 of this thesis, spatial aspects are neglected. Agents interact with each other at random, which is most likely not met in practice. Allowing for a spatial dimension usually favors cooperation, because cooperative clusters can emerge (Noailly et al., 2007; Nowak, 2006, Ch. 9). Therefore, making models spatially explicit typically helps cooperation a hand. It would be very interesting to investigate how spatial aspects influence norms of cooperation. Chapter 4 has shown that it is important for the community to be able to monitor each other. An interesting question would be to investigate how social control is affected by spatial considerations. Are social norms of cooperation likely to evolve if everyone knows more or less what all other community members are doing? Or is it better if each individual knows exactly what the neighbors are doing, while having no idea what individuals living three streets away do? These spatial considerations are very exciting, but unfortunately beyond the scope of this thesis.

In the last sub-section it was explained why it is important not to restrict models to the stereotype of homo economicus. In chapter 4 an enforcement mechanism evolves through a sanctioning preference. Several scholars have suggested that it may be better to model moral consideration not as moral preferences, but as moral constraints; see for example Baland and Platteau (1996). While preferences and constraints are of course different sides of the same coin, the implications could be very different. Thinking about moral values as constraints is intriguing, as it may explain why some individuals are unwilling to update – and hence tighten – their moral constraint by ignoring, rejecting or denying information about, for example, climate change. A formal model along these lines has been developed by Rabin (1995). An interesting avenue would be to see how these findings correspond to approaches that rely on preferences, such as chapter 4 or Bowles and Hwang (2008). This may have implications when it comes to, for example, crowding-out – as discussed in chapter 2.

8.5 Fisheries

Chapter 7 has sketched a gloomy picture of the global fisheries, characterized by declining fish stocks, overcapacity of fishing fleets, and consequently low profits. These depressing results have laid bare the failure and limits of current management practices – and questions also the underlying scientific work. Fisheries economics absorbs a large part of the intellectual effort in the field of resource economics, while the practical relevance of this work is negligible (Wilen, 2000). In the previous chapter several ways have been addressed to make management more successful. Here, I will reflect on why fisheries management is yet so difficult. Often it is argued that political roadblocks are the main reason why current fisheries management is not successful; see for instance Heal (2007). Indeed, political objectives and constraints are much higher in fisheries than in other parts of environmental policy. This is especially problematic if these objectives are not clearly defined. But according to James Wilen (2006), political limitations are not the only reason why fisheries management failed. Wilen (2006) argues that scientific advice often overlooks that excessive fishing mortality is as symptom, rather than the root of a problem and the key message should therefore be to get the institutions right, rather than fighting the symptom of overfishing. This point is well made, even though it often makes a lot of sense to fight symptoms if the underlying problem cannot be solved immediately. According to Wilen the solution is the creation of property rights and, more specific, the establishment of individual transferable quotas (ITQs). In a different paper, Wilen (2000) concludes the resource economists' "most important policy achievement must surely be its influence on ITQs on the agenda as a viable policy instrument. " This thesis has shown many examples, where resource management is very successful without relying on private property rights. Additionally, any policy intervention, especially when it relies on market mechanism poses a danger of crowding-out stewardships motives. Therefore, Wilen's reasoning that good institutions *imply* private property rights, which *implies* individual transferable quotas, seems a bit hasty. An ITQ is a powerful tool that may work extremely well in many fisheries, but it has failed and will fail in other cases as well – see chapter 7 of this thesis. The overriding message coming from that chapter is that magic bullets and panaceas do not exist. Using the words of Amaryta Sen (2009)

> [*t*]*here are, however, good evidential reasons to think that none of these grand institutional formulae typically deliver what their visionary advocates hope, and that their actual success in generating good social realizations is thoroughly contingent on varying social, economic political, and cultural circumstances.*

Admittedly, we all have a fascination for magic bullets and panaceas. Who does not find Weitzman's (2002) conclusion that "pure ecological uncertainty unambiguously favors fees over quotas." much more appealing than Richter's (2011), who finds that "policy descriptions that work well in one setting may lead to management failure in a different context."? Nonetheless, it is our responsibilities to present solutions that work in practice and not only in stylized economic models. Models should have the aim to represent empirical phenomena and be useful – not serve aesthetical appeal.

David Bromley (2009) goes even further and sketches fisheries economists as a science that can be characterized by "chronic conceptual confusion" using incoherent models. Essentially, Bromley's point of view is orthogonal to the one taken by Wilen, as Bromley says that fisheries economists have not understood the problem and the establishments of an individual fishing quota (IFQ) should be answered by "comprehensive incredulity". Bromley further argues – again in contrast to Wilen – that a low fishing mortality (through a strict TAC) is the key, while there is no necessity to think about ways to maximize economic rents in models of fisheries management. The main difference in opinion between Wilen and Bromley boils down to the question whether we should focus on the class I or the class II problem of fisheries – as analyzed in chapter 7. Remember that the class I problem refers to an excessive fishing mortality (which could be solved by a strict TAC – as suggested by Bromley), while the class II problem refers to wasteful competition among fishermen (which can be solved with an ITQ – as suggested by Wilen). While an ITQ will improve the efficiency of exploitation, it will not necessarily protect the stock. A strict TAC, however, will preserve the stock, but a situation in which all fishermen make zero profit can neither be the aim of fisheries management, nor is it a viable strategy – simply because we can do better than that. Economic and biological sustainability must go hand in hand.

Furthermore, fisheries management is more difficult than other parts of public policy because of inherent conflicts over objectives (Hilborn, 2007). Even if objectives are clearly defined, they do often not find their way into economic models. One of the key objectives of many policy makers is secure and stable employment in the fisheries sector; see for instance Nakken et al. (1996). The idea of having employment as an explicit objective – an argument in a social welfare function – must sound very odd to most economists. However, this does not imply that employment effects of different policies cannot be addressed and assessed. Chapter 5 provided optimal policy for a flexible set of political choices, concerning the utilization of the fleet. So there is also a responsibility on the scientific side to pay closer attention to the questions policy makers have, rather than telling them what we find interesting. There are different opinions on this issue, such as the one from Robert Hahn (2000), according to whom

> *economists need to get more comfortable with the idea of being lobbyists for efficiency or advocates for policies in which they believe. This comfort level is increasing slowly. Moreover, economists are finding ways to institutionalize their power in certain policy settings.*

While I do understand the frustration of many scientists whose well-intended advice is not followed on political grounds, this is inherent part of democracy. Elected politicians have the responsibility and the legitimacy to make policy, while we scientists do not. It is the responsibility of the scientific community to inform and advise policy makers about potential problems and suggest solutions by providing tools and instruments for policy makers, rather than to answer inherently political questions based on own personal beliefs and preferences.

8.6 Concluding remarks

This discussion has reflected on how analytical evolutionary and agent-based models can remedy shortcomings of standard economic models that overemphasize steady state solutions. Furthermore, it is argued that relying only on models that assume individuals to be motivated only by material incentives is dangerous when government policy is designed. This does neither mean that we should stop using these standard models, nor does it imply that the models that were used in this thesis are always better. Scientific work requires diversity in theories, hypotheses, but also in models. These different models will provide good grist for empirical mills that help us understanding which models apply in which situations. With a diversity in models and policy tools we are well equipped to stop the current overexploitation of many of our renewable resources. But momentum is vital, because currently the transition from depleting to carefully managing some of our most valuable assets is not happening fast enough.

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Summary

Social-ecological systems are inherently complex. As a consequence, the way humans use and manage a natural renewable resource, such as a fish stock, is affected in various ways. First, this thesis investigates several mechanisms that can explain the success of self-governance of small communities without any government intervention. Second, this thesis identifies several key factors that make successful community governance more likely, while it identifies other factors – or driving forces – that may undermine or jeopardize self-governance. Finally, this thesis looks at the case where self-governance is not sufficient to effectively manage a resource, and government intervention is needed. This thesis investigates how formal institutions should be designed in the light of social and biological complexity. Since human behavior is contingent on the social environment, any attempt to change human behavior – as intended by government intervention – depends also on the social context. Government policies that fail to take this into account may alienate users and crowd out social norms of cooperation, resulting in a wasteful loss of social capital or, even worse, lead to a result that is in contrast to what was originally intended. Furthermore, it is immanent that policy descriptions that work well in one setting may lead to management failure in a different context. The overriding message of this thesis is that understanding biological and social complexity is important to achieve any management objectives one has in mind. After all, we cannot manage what we do not understand.

Chapter 2 links insights from economic experiments to theoretical models of cooperation in order to understand the circumstances under which government intervention may be countereffective, as it undermines the motivation to obey rules voluntarily. Main policy lessons are that formal institutions work best if they support local social norms in place. If this is not possible, legitimacy can be increased by involving users in the process of policy design. While legitimacy is important, an institution that is perceived to be unfair can be even worse. Financial incentives, a special class of policy tools, can be problematic for two reasons. First, they are framed in a way that they tend to make an appeal to the individual as a self-interested individual, rather than as a good citizen. Second, they are addictive – even if individuals respond to the fi-

nancial incentive as intended, any compliance immediately stops once the financial incentive is removed.

Chapter 3 addresses the question why social norms of cooperation do not necessarily erode gradually, but often suddenly collapse. In our model, socially minded individuals try to convince non-cooperators to restrain harvesting effort to obtain a higher profit for everybody – cooperation is thus contagious. With rising levels of cooperation, the temptation to defect increases, as the profits from being the one wolf in a herd of sheep are very high. Surprisingly, we find that if individuals face some constraints in their economic activities, collapse of a social-ecological system occurs under various circumstances.

Chapter 4 goes one step further and investigates how a moral system of social sanctions that may stabilize cooperation evolves over time. We start with a situation where individuals neither have a certain moral preference, nor have exact information about the nature of the social dilemma the community faces. We find that the sanctioning system automatically evolves towards the socially optimal level if two conditions are met. First, economic decisions are revised frequently, while the moral preference to sanction peers is very persistent and changes only slowly. Second, individuals base their decision whom to punish and reward on own behavior, where own behavior is the moral yardstick that separates good from bad behavior.

In the second part of the thesis we pay special attention to the design of formal institutions in fisheries. **Chapter 5** investigates how Northeast Arctic (NEA) cod could be optimally managed for various objectives, including maximizing fleet profits, minimizing the impact of harvesting on the marine ecosystem, or having a diverse fleet to maintain the current structure of coastal communities. We develop a management plan that allows the regulator to choose the fleet structure that maximizes her objectives. For each policy choice we give the optimal total allowable catch per year, and the corresponding long-run level of remaining biomass. We also give management advice in case the fleet cannot be adjusted in the short run and suggest what to do if the biomass levels are far from their long-run optimal level. In addition, the management plan also allows the regulator to compare the long-run welfare levels between the various management options to determine the costs and benefits associated with each.

Chapter 6 deals with the possibility that harvesting pressure affects the genetic composition of Northeast Arctic cod. Fish stocks that experience high fishing mortality show a tendency to mature earlier and at a smaller size. Some have suggested that these changes in life history traits could affect the fishery's yield, as fish is evolving towards smaller sizes. Therefore, fisheries-induced evolution may have economic repercussions for society. Our model for Northeast Arctic cod predicts that fisheries-induced evolution decreases economic yield if fishing mortality rates continue at historically high levels. We also find that maximum economic yield is achieved at a considerably lower fishing mortality than what the stock has historically experienced. At this lower mortality, fisheries-induced evolution is less pronounced and actually increases the economic yield. Overall, we find that evolutionary and non-evolutionary models recommend similar harvesting rates and the overriding message is that higher economic yield can be obtained at lower harvest rates irrespective of whether evolution occurs or not.

Chapter 7 shows how regulations in fisheries policy should be designed in the light of social and ecological complexities. Analyzing 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. This chapter examines 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.

Chapter 8 provides a general discussion that critically reflects on the methods used in this thesis and puts them into perspective. We should use models that give us correct answers to questions we ask. Models that look good on paper, but have little relevance in reality, tend to be bad advisors. This message is even more important when dealing with natural resources that are endangered, threatened, or under pressure. After all, we cannot protect what we do not understand.

Samenvatting

Sociaal-ecologische systemen zijn intrinsiek complex. Als gevolg van deze complexiteit zijn er verschillende mechanismen die het menselijk gebruik en beheer van natuurlijke hulpbronnen zoals een visbestand, beïnvloeden. Dit proefschrift tracht de vraag te beantwoorden welke factoren of omstandigheden ertoe leiden dat kleine gemeenschappen natuurlijke hulpbronnen succesvol kunnen beheren – zonder ingrijpen van de overheid. Een aantal belangrijke factoren wordt geïdentificeerd, die succesvol zelfbestuur faciliteren of juist ondermijnen. Ook wordt onderzocht hoe een overheid moet ingrijpen als zelfbestuur niet voldoende is om een hulpbron effectief te beheren. Dit proefschrift kijkt specifiek naar de vraag in hoeverre formele instituties rekening moeten houden met de aanwezigheid van sociale en biologische complexiteit. Aangezien menselijk gedrag afhankelijk is van de natuurlijke en sociale omgeving, zal elke poging om menselijk gedrag te veranderen – bijvoorbeeld door een overheidsingreep – beïnvloed worden door de sociale context – en op zijn beurt kan het gedrag deze context weer veranderen. Overheidsmaatregelen die hiermee geen rekening houden, kunnen gebruikers vervreemden en positieve normen en waarden zoals de neiging tot samenwerking, teniet doen. Dit kan leiden tot een verlies van sociaal kapitaal of – nog erger – tot een resultaat dat haaks staat op wat oorspronkelijk de intentie van de overheidsingreep was. Bovendien is het duidelijk dat beleid dat goed in een bepaalde situatie werkt in een andere context kan falen. De belangrijkste conclusie van dit proefschrift is dan ook dat het noodzakelijk is biologische en sociale complexiteit te begrijpen om management doelstellingen te realiseren. We kunnen immers niet besturen wat we niet begrijpen.

Hoofdstuk 2 gebruikt inzichten uit economische experimenten en theoretische modellen over de evolutie van coöperatie om de omstandigheden te begrijpen waaronder ingrepen van de overheid averechts blijken te werken omdat de motivatie om vrijwillig regels te volgen ondermijnd wordt. De voornaamste boodschap voor beleidsmakers is dat overheidsregels het beste werken als ze ondersteuning vinden van bestaande normen en waarden. Indien dit niet mogelijk is, kan legitimiteit verhoogd worden door gebruikers te betrekken bij het maken van beleid. Ontbrekende legitimiteit van een overheidsingreep kan ongunstige consequenties hebben, maar regels die als oneerlijk ervaren worden zijn wellicht nog erger. Financiële prikkels als beleidsinstrument zijn problematisch om twee redenen. Ten eerste hebben ze de neiging om

iemand als egoïstisch individu aan te spreken en niet zo zeer als gecommitteerde verantwoordelijke burger. Ten tweede zijn ze "verslavend" – zelfs als individuen zoals bedoeld reageren op de financiële prikkel, zal dit gedrag onmiddellijk stoppen zodra de financiële prikkel vervalt.

Hoofdstuk 3 behandelt de vraag waarom sociale normen niet noodzakelijkerwijs langzaam en geleidelijk veranderen maar vaak plotseling instorten. Een model wordt ontwikkeld waarin coöperatieve individuen proberen egoïstische individuen ervan te overtuigen om minder te gaan oogsten om zo een hogere winst voor iedereen te realiseren. Samenwerking is dus besmettelijk. Als meer mensen samenwerken, stijgt echter de verleiding om egoïstisch te zijn. Het is natuurlijk voordelig de enige wolf in een kudde schapen te zijn. Verrassend genoeg vind ik dat het systeem plotseling kan kantelen als individuen niet ongelimiteerd kunnen oogsten, door bijvoorbeeld maar beperkt tijd te hebben. Verschillende omstandigheden zoals instroom van nieuwe mensen kunnen er dan toe leiden dat coöperatie maar ook de omvang van de hulpbron plotseling ineen kan storten.

Hoofdstuk 4 gaat een stap verder en kijkt hoe sociale sancties zoals waardering of afkeuring van bepaald gedrag een coöperatieve houding kunnen ondersteunen en stabiliseren. Verder wordt geanalyseerd hoe deze sancties ontstaan en hoe ze in de loop der tijd veranderen. We beginnen met een situatie waarin mensen geen bepaalde morele voorkeur hebben en ook niet weten welk gedrag het beste zou kunnen zijn voor de maatschappij, en komen tot de conclusie dat het morele systeem automatisch streeft naar het sociaal-optimale niveau als aan twee voorwaarden wordt voldaan. Ten eerste worden economische beslissingen vaak herzien, terwijl morele waarden en normen slechts langzaam veranderen. Ten tweede baseren mensen hun sanctie beslissing op eigen gedrag, waarbij het eigen gedrag de morele maatstaf is die goed van slecht gedrag scheidt.

In het tweede deel van het proefschrift besteed ik bijzondere aandacht aan het ontwerpen van formele instituties in de visserij. **Hoofdstuk 5** onderzoekt hoe "Northeast Arctic (NEA) Cod" – kabeljauw die vooral in de Barentszzee voorkomt – optimaal kan worden beheerd onder drie verschillende doelstellingen: pure winstmaximalisatie, winstmaximalisatie onder de voorwaarde dat geen boten gebruikt worden die averechtse effecten op het ecosysteem hebben, en onder de voorwaarde dat om culturele redenen een diverse vloot wordt gebruikt. We ontwikkelen een beleidsplan dat de toezichthouder op iedere vlootstructuur kan toepassen die haar doelstellingen maximaliseert. Voor elke doelstelling geven we de optimale totale vangsthoeveelheid (“total allowable catch”) en de bijbehorende biomassa op lange termijn. Bovendien geven we een beleidsadvies voor situaties waarin de boten van een vloot niet op korte termijn aangepast kunnen worden of waarin de biomassa ver van het lange termijn optimum verwijderd ligt. Onze analyse maakt het tevens mogelijk om kosten en baten van verschillende opties met elkaar te vergelijken.

Hoofdstuk 6 onderzoekt de mogelijkheid dat visserij onomkeerbare evolutie veroorzaakt doordat de genetische samenstelling van NEA kabeljauw wordt beïnvloed. Vissoorten die een bijzondere hoge visserijdruk ervaren tonen een neiging om op jongere leeftijd en kleinere lengte volwassen te worden – wat ten koste van de lichaamsgroei kan gaan. Deze veranderingen zouden de productiviteit en dus het rendement van de visserij kunnen beïnvloeden. Daarom kan visserij-geïnduceerde evolutie economische gevolgen hebben voor de samenleving. Ons model voor NEA kabeljauw voorspelt dat de visserij-geïnduceerde evolutie het rendement in de visserij verlagen als de visserijdruk zo hoog blijft als in het verleden. Als er minder vis gevangen wordt, is het effect van visserij-geïnduceerde evolutie zwakker en kan de economische opbrengst zelfs stijgen. Wij vinden dan ook dat de maximale economische opbrengst wordt bereikt als de visserijdruk aanzienlijk lager is dan in het verleden. Verder vinden we dat evolutionaire en niet-evolutionaire modellen over het algemeen vergelijkbare vangststrategieëen aanbevelen. De kern van de boodschap is dat lagere vangsthoeveelheden tot hogere economische opbrengst leiden – ongeacht of evolutionaire verandering optreedt of niet.

Hoofdstuk 7 bekijkt hoe regelgeving in het visserijbeleid rekening moet houden met sociale en ecologische complexiteit. Het analyseren van enkele componenten van een complex adaptief systeem is vaak misleidend omdat het daadwerkelijke gedrag bepaald wordt door interacties met andere componenten, en dus het hele systeem. We moeten rekening houden met de fundamentele complexiteit van de sociale en natuurlijke omgeving om onaangename verrassingen te vermijden. Dit hoofdstuk onderzoekt welke concrete beleidsinstrumenten in de visserij beschikbaar zijn en welke voordelen en nadelen deze hebben. Wij constateren dat het succes van bepaalde instrumenten niet alleen afhangt van de inherente eigenschappen, maar ook van de manier waarop deze instrumenten worden toegepast.

Hoofdstuk 8 bediscussieert kritisch de gebruikte methoden van dit proefschrift en vergelijkt ze met de aanwezige literatuur. Het is belangrijk om modellen te gebruiken die juiste antwoorden geven op de vragen die we stellen. Elegante modellen die niet toe te passen zijn op werkelijk problemen zien er wellicht leuk uit, maar zijn over het algemeen een wankele basis voor beleidsadvies. Dit is des te belangrijker wanneer het gaat om natuurlijke hulpbronnen die bedreigd worden. We kunnen immers niet beschermen wat we niet doorgronden.

Epilogue

January 2011, Cafeteria of the Georg Sverdrups hus, University of Oslo, Norway

Excuse me, can I sit down here? Or is this place taken? Thanks, very kind, why is it so incredibly busy here? Yes, I think you are right, all students came back from holiday and now everyone meets up for a coffee. I must say, the coffee is fantastic here, much better than the one that we have in the Netherlands. Have you ever been in Holland? Yes, I agree, Amsterdam is a fascinating city. I was living in Wageningen – it is a small town in the middle of the country. I did my PhD there, but now it is finished and I am continuing my research here in Oslo. Oh, you are also thinking about doing a PhD? Well, doing a PhD can be fantastic, but at the same time I have to warn you. I mean, as a PhD student you are extremely vulnerable. I have seen incredibly smart PhD students struggling and suffering, simply because they were at the wrong place at the wrong time. But fortunately I was very lucky. Above all, I had great supervisors. My daily supervisor Johan always found the right balance between letting me explore my own opportunities and making sure that I did not get lost. Johan always put me back on the ground when I came to the office wearing shades, because I thought my future was so bright, but he also built me up when I was on my knees, crawling on the Boulevard of broken dreams. And then it is of course only thanks to my second supervisor Daan that I was working on such a cool project in the first place. Daan is an excellent economist and it was really fun to work with him. I hope to continue working with him in the future. Also the other project members have helped making this project such a success. It was great to have Jan as a fellow PhD student on my side to share experiences and ideas. Also Jana's input was very useful in putting the project on the right track. And I guess all of this would not have happened if I had not enrolled in this resource economics course that Erwin was teaching when I was a Master student. Thanks to him I discovered that doing research on the interface of natural resources and institutions can be extremely exciting.

My arrival in Wageningen feels like such a long time ago. But I still remember when I showed up at my department for the first time – I was kind of scared, well basically I was a little boy. Marjolein helped me getting started, and sharing one office with her was so much fun! I am also thankful to Jaap, who has been a very supportive group leader and cared a lot about us PhD students. I also enjoyed being together with my fellow PhD students at Biometris, Maaike, Santosh, Apri, Yiannis, Sabine, Tahira, and Robert. Thanks to Patricia, Elisa, Sanne, Simon, Evert-Jan, Martin, Maarten, Onno Lia, but also all others colleagues, Biometris was a very nice working environment. Well, life in Wageningen is a bit on the quiet side, but I always enjoyed living there, also because of all the friends that I had. I also had a good start, living in a house with cool people. I had a fantastic time together with Herman, Sofieke, Renskita Coen, Ben, JJ, Maya, but also all others who have been there. I became close friends with one of my housemates, Maartje, with whom I lived later at a different place where we always had Sushi competitions. I have never admitted this to her, but her Sushi were so much better than mine. Always! It was also a lot of fun to have Tanya in that house towards the end of my PhD. Here in Oslo I will definitely miss my good friend Vasilis, with whom I often discussed the challenges and opportunities both of us were facing in the Netherlands and abroad. I was also very happy that my friend Kristina moved from Ijmuiden to Wageningen – it was very good to have her around! I will also miss the lunches and conversations with Marleen. I always enjoyed hanging out with my friends from the aquatic ecology group, Andrea, Nika, Darya and Mascha, who always accepted me, even though I was as an economist a bit of an invasive species. And often I had a fantastic time with my good friends Lucie and Marion. If all else fails, I have the backup-plan to open a bar with Marion in Wageningen. We want to call it "Bayesian Lounge". And I am incredibly happy that I met Judith. She is just awesome. I wish she was here now!

Why I ended up in Oslo? Well, basically we have got to blame Mia for all this. She convinced me to come here for a summer to work on the cod and then I fell in love with Norway. Mia is a great biologist and it is fantastic to work with her. She can be my wingman anytime! And she also became a wonderful friend. It has also been a pleasure to work with Nils-Christian – I think we have only scratched the surface so far and I am looking forward to see how far we can get! It was also great that Kjell-Arne, Karine and Ragnhild gave me a helping hand to find my way among the economists in Oslo. I guess working at CEES was also such a nice experience because all my colleagues and friends there were so nice and kind! First of all Jan, with whom I discovered that it can be extremely dangerous to open a semi-frozen bottle of Pepsi Max. Did you know that it explodes? Yes I know, everyone is talking about these Mintos, but what we experienced was much more dangerous! It was also great to have Florian around at CEES, my fellow-economist among all these bird-huggers and tree-watchers. I also found some very good friends in Oslo. It was "grisebra" to live with Anette, she is incredibly funny, and such a good musician! You should check out her website: www.anetteaskvik.com. And I will never forget the nights out in Grünerløkka with Silje! Together we were setting the city on fire many times and dancing (almost) like stars. I also enjoyed the evenings with Svenn in Kafé Tamara – and of course the dinners and conversations with Håkon! Thanks to all my friends here in Oslo, this is not just a place where I work now, but it feels already a little bit like my home.

Did I mention that the coffee tastes really good here? Have you ever been in Vienna? The coffeehouses there are just marvelous! I spent quite some time there while I was at IIASA, a research institute, where I did parts of my PhD. Thanks to Ulf I had the possibility to visit IIASA many times. At IIASA, it was very good that Åke helped me using Adaptive Dynamics. I also enjoyed all stimulating discussions with Karl around the evolution of cooperation. It all started when I visited the Summer Program for Young Scientists. It was a fantastic summer, also thanks to all the time I spent with Dolly, Heidi, Thomas, Michiel, Neala, Gergely, Fabian, Simona, Dietmar, Marta, Tyler, Carolyn, Miyuki, and Vivek. When I came back to IIASA after that summer I was always looking forward to see my friends Bartek, Varia, Fiona, Davnah, and Jacob. And you know what the best thing is? One of my dearest friends in Vienna, Barbara, has also moved to Oslo. It is "leiwand" to have her here in my cercle d'amis! Just before I finished my PhD I spent an unforgettable summer with Thomas and Lauren in Burlington. I hope to see them back very soon, maybe in Oslo even. I still have not visited Alex in Sydney, but one day I will! Alex is one of my oldest friends, and he also made the great pictures on the cover of my thesis.

But you know what the disadvantage is? While doing my PhD I met so many great people and found so many new friends, but sometimes I really miss my family and my old friends. They are so important to me, but I spend so little time with them. Without their support I could never have written my thesis. Sometimes I wish all of them would live in the same city as I do. Then I could visit them without travelling all day, or I would give a birthday party and everyone would just come. Sorry for telling you all this! I got a bit carried away. Thanks, it is nice of you to say that you enjoyed the story. You wonder whether I wish I had done anything differently? No, not really. If I look back, all my decisions made a lot sense in the particular context they were made. No, that does not mean that all of my choices were good. Not at all! If I could start again, I would do a million things differently. Basically I would do everything differently. But let's not worry! It' too late now, it will always be too late, fortunately.

Det vil helst gå godt. (Max Manus)

About the author

Andries P. Richter was born on June 24, 1978 in Bremen, Germany. He studied economics at the University of Groningen, The Netherlands. During his studies, Andries worked as a research assistant and as a teaching assistant for the courses *Microeconomics I*, *Macroeconomics I,* and *International Economics* at the Department of Economics. There, he also enrolled in the Research Master program with the profile "Economics and Econometrics". After graduation in 2006, Andries started his PhD research at the Mathematical Department of Wageningen University. In Wageningen, Andries was a teacher for the course *Mathematics T* on Linear Algebra each fall. As part of his PhD, he joined the Young Scientists Summer Program at the International Institute for Applied Systems Analysis (IIASA) in Laxenburg, Austria in 2007. His research there was awarded with the Mikhalevich scholarship, which facilitated another stay at IIASA in summer 2008. Andries visited the Centre for Ecological and Evolutionary Synthesis (CEES) at the University of Oslo, Norway, in Summer 2009 and spring 2010 to work on the applied case of the Barents Sea ecosystem. Andries finished his PhD in Wageningen at the end of 2010. Afterwards, he joined the CEES as a Post-doc researcher to work on social complexity in resource management.

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Research Skills

- Young Scientists Summer Program, IIASA, Austria
- Mikhalevich Award, IIASA, Austria

Oral Presentations

- The evolution of social norms for renewable resource harvesting, 16th Annual Conference of EAERE, 27 June 2008, Gothenburg, Sweden
- The evolution of social norms for renewable resource exploitation, 10th International BIOECON Conference on "The Effectiveness and Efficiency of Biodiversity Conservation Instruments", 29 September 2008, Cambridge, UK
- The evolution of social norms for renewable resource exploitation, SENSE / EPCEM Symposium, 10 October 2008, Wageningen The Netherlands
- Temptation versus norm compliance in a common-pool resource game when behavior is contagious, 17th Annual Conference of EAERE, 27 June 2009, Amsterdam, The Netherlands
- When do social sanctions work? The invisible hand of social norms in overcoming a social dilemma, Fourth World Congress of Environmental and Resource Economists, 30 June 2010, Montreal, Canada

SENSE Coordinator PhD Education and Research

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