Harvest control rules in modern fisheries management

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Abstract

Harvest control rules have become an important tool in modern fisheries management, and are increasingly adopted to provide continuity in management practices, to deal with uncertainty and ecosystem considerations, and to relieve management decisions from short-term political pressure. We provide the conceptual and institutional background for harvest control rules, a discussion of the structure of fisheries management, and brief introductions to harvest control rules in a selection of present day cases. The cases demonstrate that harvest control rules take different forms in different settings, yet cover only a subset of the full policy space. We conclude with views on harvest control rules in future fisheries management, both in terms of ideal and realistic developments. One major challenge for future fisheries management is closing the gap between ideas and practice.

Introduction

Harvest control rules (HCRs) allude to a long-standing desire for, and often perceived necessity of, controlling harvests of marine fishes. While concerns over degradation of coastal fisheries were present already in pre-industrial times (Garcia et al., 2014), ocean fish resources were held to be practically inexhaustible until modern times. However, the development of industrial-scale fishing, accelerating in the interwar years, quickly demonstrated that marine resources had limits. The collapse of Pacific sardine (*Sardinops sagax*) in the 1940s stands as a prominent example, and the establishment of regional commissions in the Pacific (halibut in 1932, salmon in 1937) demonstrated a rising, international concern (Garcia et al., 2014). After World War II, public awareness of environmental problems – particularly those of resource exhaustion – grew, and institutions developed to address some of these problems. Early fisheries regulations consisted of restrictions only on entry, gear, and season length (inputs), while subsequent regulations have increasingly relied on catch (output) limits. As understanding and knowledge of marine ecosystems have grown, and as economists and social scientists have unraveled the economic, political and social realities of fisheries, the provision of modern regulatory advice has come to rely on comprehensive and complex undertakings that transpire in an integrated science-policy setting. Harvest control rules lie at the heart of these undertakings. Arriving at present day harvest control rules has been a long process, involving legal and institutional developments, theoretical and conceptual understanding, and managerial practice and experience. Further, harvest control

rules take different forms in different settings; this variation reflects differences in governance needs and frameworks, but also the unique attributes, histories, and management needs of each fish stock and fishery. With this essay, we portray harvest control rules as a manifestation of rule-based management, show-case present use in a number of fisheries, and discuss challenges for future developments.

A harvest control rule has been described as providing "the scientific basis for the tactics employed in a fishery and depends on explicit or perceived management objectives and the data available on which to base scientific management advice" (Punt, 2010, p. 583), and as "an algorithm and a tactical management tool that translates biological information […] into management information such as a [total allowable catch limit]" (Eikeset et al., 2013, p. 173). (Punt, 2010, is perhaps the authoritative review, but see also Deroba and Bence, 2008.) Harvest control rules have emerged alongside increasing demands on fisheries management to observe the precautionary approach and objectives of sustainable development, and in response to criticism of complex and incomprehensible fisheries models used for management advice. Thus, we offer an account of the conceptual and institutional development underlying harvest control rules, a discussion of the structure of modern fisheries management and further discussion of current arrangements in different fisheries, highlighting cases that span a considerable section of the policy space. Ultimately, we offer reflections on challenges and the future potential of harvest control rules, in particular with regard to climate change. Along the way, we show how placing a wider scope of the political domain in HCR schemes can relieve fisheries management decisions from political pressure motivated by short-term rent capture and gains, and how this pressure reenters via ad hoc provisions in real-world implementations of HCRs.

Harvest control rules are implemented as 'top-down' management measures where a central regulatory body typically decides the total harvest and sets further restrictions like spatial and seasonal limits. Central regulations are often considered appropriate in situations with externalities; that is, when all costs associated with an economic activity are not borne by the active agents but rather that costs are also borne by third parties. Externalities often arise with exploitation of public goods like fish stocks or clean air (Gordon, 1954; Hardin, 1968). Public and total costs of 'bottom-up' approaches to regulations of public goods, which are based on property rights and a higher level of self-governance, are often lower because users have incentives to take account of externalities and information asymmetries and efficiently internalize them (Ostrom, 1990; Grafton et al., 2006). With 'top-down' fisheries management, monitoring, enforcement, and transaction costs are in general higher (Lane and Stephenson, 2000). Higher costs notwithstanding, if fishers accept HCRs as important tools to secure longevity of fish stocks and hence future fishing opportunities, fishers will more likely comply with fishing regulations and further provide information relevant to management. As we will discuss, HCRs can facilitate a fisheries governance system where regulators and fishers work together to decide on overall harvest. But to provide fishers with incentives for such compliance, some kind of rights-based management system that allow fishers to decide on where and how to fish is probably necessary (Grafton et al., 2006).

Modern fisheries management should, according to international ocean law, adhere to the precautionary approach and take ecosystem considerations into account (Hoel and VanderZwaag, 2014; see also FAO, 1996, 2003). Fisheries management is also subject to substantial domestic legislation. For example, fisheries management in the US should, among other things, prevent overfishing, use the best available science, consider economic efficiency and fishing community resilience, and minimize bycatch, with particular emphasis on bycatch of endangered species (Holland, 2010). Further, modern fisheries management is subject to substantial uncertainty, in terms of a fluctuating environment, political vagaries, and a host of other aspects. For example, uncertainty related to current and future regulations may be an issue, particularly for fisheries on migratory and straddling fish stocks. The introduction of HCRs into modern fisheries management represents a conceptual shift from model-based to rule-based management; a management framework that is ideal for a wide range of decision problems under uncertainty (see Walters and Hilborn, 1976, and Sandal and Steinshamn, 1997, for fisheries related discussions). The shift has arguably led to a more complex management framework and difficult policy choices, but also has substantial benefits in the long run. The primary benefit is that it forces political decisions to explicitly reflect long-term preferences. The shift has also stimulated conceptual developments of fisheries systems that are less reliant on complex models (Schnute and Richards, 2001). Rule-based management can address many of the involved uncertainties and has arguably simplified management operations.

To demonstrate how the HCR framework is implemented in modern fisheries management, we discuss six cases in some detail: Northeast Arctic cod (*Gadhus morhua*), North Sea cod, Norwegian spring spawning herring (*Clupea harengus*), Icelandic capelin (*Mallotus villosus*), Pacific sardine, and Alaska groundfish. All but the latter case deal with single species fisheries while Alaska groundfish fisheries are discussed together because their management is and has been very much similar. We chose these cases because they demonstrate some of the variability of management schemes that are possible within the HCR framework. The different fisheries also have different histories and take part within, and have been shaped by, different institutional frames. Among our cases, explicit reliance on environmental conditions only showed up once (Pacific sardine), ecological considerations are incorporated in harvest control rules in one case (Alaska groundfish) and indirectly influencing management in two cases (Icelandic Capelin, Pacific sardine), while all cases are

subject to one or more provisions that restrict the direct feedback from indicators to management measures. Thus, our cases suggest that considerable distance remains between the theoretical concept and its current practical implementation.

HCRs have been heavily discussed both in the scientific literature and in policy documents. Punt (2010) provides a broad discussion of HCRs in fisheries management, highlighting the intimate connection between stock assessments and HCRs, among other things for reference points. Froese et al. (2011) discuss a set of generic HCRs for European fisheries, with references to the relevant legal and institutional framework. Their HCRs are based on six pillars: economic optimization, international fisheries agreements, the precautionary principle, relevant governance experience, ecosystem-based fisheries management, and known biological properties. Froese et al. (2011) also discuss the motivation and science behind the different rules, before ultimately discussing HCRs for North Sea herring and blue whiting (*Micromesistius poutassou*). Similarly, Eikeset et al. (2013) compare the current HCR for Northeast Arctic cod with alternative HCRs that are based on conceivable objectives like profit maximization and optimal yield. To mention a few other studies, Roel and Oliveira (2007) and Hegland and Wilson (2009) discuss HCRs for the Western horse mackerel (*Trachurus trachurus*); Enberg (2005) and Tjelmeland and Røttingen (2009) discuss possible and actual HCRs for the Norwegian spring-spawning herring. Further studies and discussions of implementation of HCRs are mentioned in Punt (2010), in ICES (2006), and in our discussion below.

Conceptual and institutional background

A driving force in the formation of the International Council for the Exploration of the Sea (ICES) in 1902 was the idea that marine research could increase catches in fisheries and support their regulation (Rozwadowski, 2002). Almost half a century later, when ICES restored their activities after World War II, the problem of describing fish stock dynamics was perceived as a major impediment to support fisheries regulations. In response, Beverton and Holt (1957) developed their groundbreaking model of fish stock dynamics, which enabled fisheries scientist to study how fishing pressure and regulations influence fish populations and their exploitation. The Beverton-Holt model and related advances in fisheries science became the standard tool for description and prediction of fish stocks and thus prepared the ground for the conceptual development of HCRs.

While the conceptual development of models that encompass mappings from potential regulatory decisions (fishing pressure) to population effects happened within the realm of biology, the first theoretical considerations of feedback solutions in fisheries models, which essentially translate stock measures or indicators into harvest advice, came from fisheries economists. Feedback solutions can be viewed as forerunners to modern HCRs and were implicit, perhaps as early as in Crutchfield and Zellner (1962; the dynamic solution of their model follows a most rapid approach path where the harvest level essentially depend on the stock level; see discussion in Wilen, 2000), but definitely from the 1970's, when for example Clark and Munro (1975) described the optimal harvest rate as depending on the stock level. From then on, standard fisheries economics models, subsequently known as bioeconomic models, provided optimal harvest rules as functions of the stock level. A crucial part of the advance in fisheries economics hinged on the representation of fisheries management problems in a capital-theoretic framework. Capital theory remains as the major framework for analysis of natural resource governance (Arrow et al., 2012; Fenichel and Abbott, 2014; Guerry et al., 2015). Conceptual similarities between bioeconomic feedback solutions and HCRs notwithstanding, modern HCRs are usually not derived from formal, integrated bioeconomic optimization models, but are rather simple, common-sense rules based on expert opinion.

In the late 1980s, The World Commission on Environment and Development established the concept of sustainable development on the international political agenda (WCED, 1987). The Commission identified the oceans as an important topic in this regard, and when the United Nations Conference on Environment and Development took place in Rio in 1992, oceans and fisheries attracted increasing attention; fisheries were about to become a major environmental issue. The Rio Declaration (UNCED, 1993) included prescriptions relevant for oceans and fisheries management: the precautionary principle, prescribing that 'lack of full scientific knowledge' should not 'be used as a reason for postponing cost-effective measures to prevent environmental degradation' (Principle 15); Agenda 21, an action plan that set objectives and timelines for the environment in the twenty-first century; the Biodiversity Convention; and the Framework Convention on Climate Change. Following up on the conference in Rio, the UN General Assembly decided that global action was needed to protect fish stocks on the high seas in particular, and initiated negotiations under UN auspices of an international agreement to this end.

Alongside the UN-led global developments, several regional developments took place. In 1990, after rising concern over the health of the North Sea in the 1980's, ICES established a working group on ecosystem effects of fishing (Rozwadowski, 2002). The ICES working group initiated the incorporation of ecosystem

considerations into European fishery management advice. In 1991, the concept of Ecologically Sustainable Development was introduced into Australian fisheries management. These developments strongly influenced work on precautionary and ecosystem approaches in FAO (Garcia et al., 2014).

Aiming to implement the provisions of the Law of the Sea Convention regarding straddling and highly migratory fish stocks, the UN Fish Stocks Agreement was negotiated in 1993–1995. The agreement, which entered into force in 2001, generally applies to areas beyond national jurisdiction, but certain principles, the precautionary approach in particular, also applies in the exclusive economic zones. The agreement elaborates on the provisions of the Law of the Sea Convention regarding fisheries management to make these provisions more precise, as for example the requirement to base management decisions on the best available scientific evidence.

The precautionary approach is operationalized in an annex to the UN Fish Stocks Agreement that requires states to establish limit and target reference points for exploited fish stocks. These reference points identify levels to be avoided and levels to be approached by relevant management measures. These provisions are the real start of harvest control rules in fisheries management that were agreed upon at a global level, with requirements to maintain stock size and exploitation rates at sustainable levels. At the time, similar approaches had already been adopted in specific cases such as by the Scientific Committee of the International Whaling Commission (see Punt, 2010). The precautionary approach was also reflected in other international instruments that developed simultaneously, such as the FAO Code of Conduct for Responsible Fisheries (1995).

In advance of the adoption of the UN Fish Stocks Agreement in 1995, the FAO developed guidelines for the implementation of a precautionary approach. The guidelines (FAO, 1996) point to the need to take inherent uncertainties into account, stating that "[m]anagement according to the precautionary approach exercises prudent foresight to avoid unacceptable or undesirable situations, taking into account that changes in fisheries systems are only slowly reversible, difficult to control, not well understood, and subject to change in the environment and human values" (p. 8). The guidelines address the need for management plans and the incorporation of harvest control rules in these plans. "Harvest control rules should specify what action is to be taken when specified deviations from the operational targets and constraints are observed" (Punt, 2006, p. 441).

The FAO guidelines were followed up at domestic levels of governance and in regional contexts over the subsequent years. In the Northeast Atlantic, for example, ICES elaborated on these efforts to translate them into practical management advice in the region (ICES, 1998); ICES (2006) provides a more detailed description of how the principles are intended to be implemented. Lassen et al. (2012) described the ICES-approach as follows: "The [precautionary approach-based] ICES fisheries advices was primarily risk-averse, i.e. ICES advice aimed at reducing the risk of something undesirable happening to the stocks rather than it advised on an optimal situation in the fisheries" (p. 6). Today, a large number of fishing nations and fishery management organizations have adopted the precautionary principle and the FAO guidelines (Punt, 2006), and harvest control rules are viewed as ideal for translating scientific results into fisheries management measures. Harvest control rules are at the same time used to pursue various objectives in fisheries management, for example, stability for the fishing industry and maintaining ecosystem services. These objectives may at times be conflicting, as we will discuss further below. Nevertheless, harvest control rules are a necessary element of transparent fisheries management to ensure that the way in which harvest varies with the status of the resource is clear to all stakeholders (Deroba and Bence, 2008).

For managers and policy makers, the application of the precautionary approach through harvest control rules is a way of committing themselves to a predefined, transparent course of action that they cannot easily deviate from. Like Odysseus, managers tie themselves to the mast to avoid prioritizing short-term gains at the expense of long-term sustainability. Compared to earlier regimes, harvest control rules have likely made it easier to defend unpopular decisions when decisions result from a predefined rule. However, times of crises have also lead to revisions of harvest control rules when revisions are considered necessary.

While developments in the UN, the UN Fish Stocks Agreement in particular, paved the way for institutionalization of HCRs and as such can be regarded as important and successful, it has been argued that they fail to provide an adequate framework for sustainable high-seas fishing (Hallwood, 2016). Further, the focus on sustainable development has – by shifting the focus from current social issues to a concern for future generations – weakened the status and importance of social objectives (Symes and Phillipson, 2009).

The structure of fisheries management

Harvest control rules reside at the intersection between domestic legislation and international law while being subject to conflicting objectives of conservation and exploitation (Lane and Stephenson, 1995; Guerry et al., 2015). At the same time, HCRs must be considered in the context of complex biological and economic systems. Partly because of this institutional and conceptual complexity, progress of fisheries management toward present day systems has been slow (Garcia and Charles, 2007; Moxnes, 2010).

Over the last decades, fisheries management has been subject to substantial criticism as overfishing has remained a global problem (Pauly et al., 1998; Dulvy et al., 2003; Hilborn et al., 2003; Clark, 2006; Worm et al., 2006; Link, 2010). Many key stocks are at low levels (Worm et al., 2009), and historically important

fisheries have suffered collapses up until fairly recently (Mullon et al., 2005; Schrank and Pontecorvo, 2007). Much of the criticism targets what often is referred to as model-based management, which alludes to analytic, mathematical virtual population formulations that are based on the Beverton-Holt framework and the sentiment that all knowledge necessary for fisheries management can be embedded into a single model. The criticism, the adoption of the precautionary principle and ecosystem perspectives in fisheries, and the perceived crisis in global fisheries created a need for more intuitive and acceptable management routines, and rule-based management – harvest control rules in particular – is, as we will argue below, a response to some of the criticism. Whether it will contribute towards solving or improving the situation in global fisheries is perhaps too early to say and we will return to this question below.

Model-based management has been criticized on several levels, in particular regarding uncertainty and errors in predictions (Peterman and Anderson, 1999, Garcia and Charles, 2007; Schrank and Pontecorvo, 2007), requirements for detailed, accurate, and vast data (Grafton et al., 2006; Kelly and Codling, 2006), and a tendency to create misperceptions about dynamics and feedbacks (Moxnes, 1998). Further criticism has addressed ad hoc assumptions and generally identification problems (Schnute, 1987, 1994; Schnute and Richards, 2001). Specifically, the issue of estimating stock-recruitment relationships has been, and remain, difficult (Needle, 2002; Subbey et al., 2014). Underlying many of these issues are models of fisheries systems that have become increasingly complex and incomprehensible to policy makers and stakeholders, if not fisheries scientists themselves (Moxnes, 1998; Rochet and Rice, 2009). "[…] the lure of mathematics and modeling created in fisheries scientists a false sense that they understood the systems they studied" (Rozwadowski, 2002, p. 247). Further, data requirements of modern fisheries management models typically surpass available observations (Schnute and Richards, 2001), leading to uncertainty about the relevance of models and their structure. As pointed out by Schnute (1994), this identification problem could be turned into an advantage because it "can help to delineate the limits of current knowledge and to establish rational priorities for future data collection" (p. 1686). But whether more and better data is the best way to improve fisheries management is not clear.
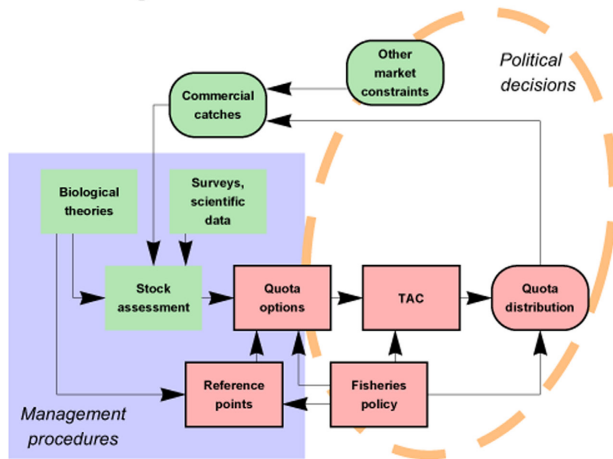
Uncertainty in fisheries models may lead to collection of more detailed data (Grafton et al., 2006), and with the advent of the precautionary principle and the ecosystem approach to fisheries, there is a conflict between demand for ecosystem knowledge and the apparent exponentially increasing costs associated with obtaining such knowledge (Degnbol, 2003). The costly collection of biological data may also limit resources for collection of economic data – data that is much needed, in general (see Gordon, 2015), and in particular for extending the ecosystem approach to include socio-economic considerations. More detailed biological data has the potential to reveal inaccuracies or errors in the existing model framework, or inspire development of more complex models, all of which result in more or new uncertainties. Ultimately, "more and better information about fish stocks or the environment in the face of uncertainty does not necessarily ensure better fisheries outcomes" (Grafton et al., 2006, p. 133).

The complexities, uncertainties, and increasing costs inherent in model-based fisheries management reduce trust in, and legitimacy of, the fisheries management system (Hauge et al., 2007; Rochet and Rice, 2009). This reduced trust can lead to misreporting, non-cooperation, and non-compliance in the fishing industry (Daw and Gray, 2005; Kelly and Codling, 2006), which again undermines and further erodes legitimacy of the management system (see Pascoe et al., 2014 and references therein). Fisheries scientists are well aware of the various issues discussed above (see, for example, the initial discussion in Link, 2010) and have risen to many of the inherent challenges (Patrick and Link, 2015). Adoption of harvest control rules is, as we will argue below, a part of this development.

The broader perspective of modern day fisheries management, in terms of ambition and need (ecosystem-based management), legal developments (for example, with respect to rare and endangered species), and interdisciplinary research (for example, use of ocean circulation models in estimates of recruitment) has increased the complexity of the management problem (Stringer et al., 2009; Dickey-Collas, 2014). For example, the possibility of performing precise analyses generally decreases as complexity increases (Zadeh, 1972; see Rochet and Rice, 2009, for a discussion related to fisheries management). Even if a complex system could be modeled precisely, the necessary effort or ability to express it may be prohibitive. The broader perspective not only increases relevant system complexity beyond a level where useful functional models can operate, it also increases the number of conflicting interests between and within a larger group of users and stakeholders. The motivation for introducing an ecosystem perspective also expands the scope of management, possibly including new, conflicting interests. Conflicting interests in fisheries management among the commercial realm (fisheries), unpriced ecosystem services (carbon sink), existence values (Davidson, 2013), and other interests put further demands on modern-day fisheries management (Brodziak et al., 2004). The total burden of these challenges makes it virtually impossible to build an analytical, functional fisheries model (Hilborn, 2011; Cowan et al., 2012), and, in the words of Schnute and Richards (2001), "implies the need for a new understanding of the role of modeling in fisheries management" (p. 16).

Since the 1970s, ICES – as a leading institution for fisheries science and fisheries management advice – developed a framework connecting scientific knowledge and assessment procedures to advice for fisheries management (Rozwadowski, 2002). For the purpose of our discussion of the transition from model-based to

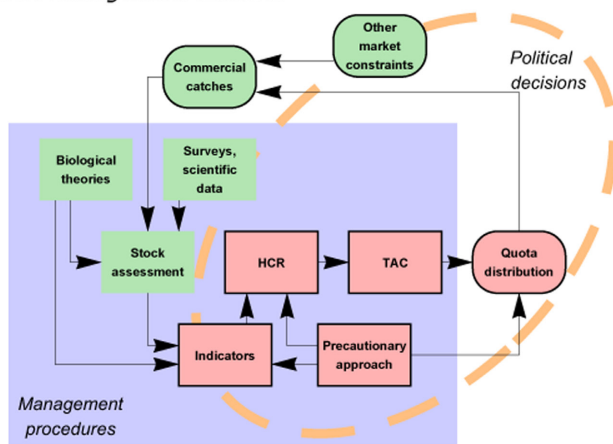Former management scheme:

HCR management scheme:

rule-based management, the ICES framework will serves to illustrate model-based management. Figure 1 (upper panel) provides a schematic illustration of the ICES framework. The blue box covers the domain of activities in ICES (science, surveying and data collection, reference points, stock assessments, policy advice). The dashed oval covers the policy domain (general policy, including acceptable risk levels, total catch limits, quota distribution). There is a principal difference between limit reference points, which reflect scientific thresholds, and precautionary reference points, which reflect policy risk preferences. This difference and the clear distinction between the realms of science and policy in the ICES framework has practical issues, for example with regard to handling and communication of uncertainty (Hauge et al., 2007). To be sure, ICES had a clear intention of keeping science and policy 'at arm's length' (Rozwadowski, 2002, p. 205). However, the use of reference points poses challenges to the separation of science and policy (Hauge et al., 2007). Also, fisheries science and policy are perceived to have conflicting objectives, making a separation difficult to maintain (Daw and Gray, 2005; Schrank and Pontecorvo, 2007).

As discussed above, the renewed or invigorated international attention to conservation issues, manifested in the precautionary principle and a general acceptance of sustainable development as a guiding doctrine, in concert with mounting criticism against model-based management, led to the adoption of harvest control rules – rule-based management – in fisheries. Rule-based management refers to a rather broad category of management models that for the purpose of our discussion needs to be delineated from model-based management. Schnute and Richards (2001) discuss two types of decision rules that correspond to the two modes of management. The model-based type relies on 'an estimation model that uses observed data to produce a control value,' while in the alternative mode, 'the control comes directly from data' (p. 15). How the control 'comes from data' represents a rule, but this terminology has the potential to confuse because such rules are routinely referred to as management models, perhaps particularly in the economic literature (see, for example, Sandal and Steinshamn, 1997). Although terminology and language use more generally may pose

real challenges to the interdisciplinary work required for fisheries management (Lane and Stephenson, 1995; Garcia and Charles, 2007), a further discussion of use and misuse of terminology is beyond our current scope. However, we touch upon the related issue of communicating scientific uncertainty in our concluding discussion.

HCRs, as instances of rule-based management, have not greatly altered the structure or information flows of the fisheries management scheme, but have changed the conceptual domains of science and policy (see lower panel of Figure 1). The domains of science and policy intersect under HCRs, which better reflects the interwovenness of science and policy (Hauge et al., 2007). Further, certain parts of the framework have changed in their content and role. With regard to the latter, indicators based on stock assessments, scientific theories and policy have taken the role of former reference points; that is, stock assessment models that earlier were the direct basis for management advice, now produce indicator values that serve as inputs for the HCR. The HCR reflects policy choices for translating the inputs into acceptable catch levels, typically in the form of limits. These changes have made policy and total catch decisions more clearly a part of the scientific domain, while the HCR and the indicators, which are an integral part of the HCR, have become part of the political domain. While this intersection of the scientific and political domains has led to a more complex management problem, the admittance of political influence on the HCR has forced the political domain to adopt a more long-term perspective and has relieved the total catch decision from vagaries of short-term political pressure. A political long-term perspective arguably also leads to more rationality in management. Ultimately, despite a more complex management problem in the larger perspective, for example in terms of deciding the HCR, HCRs have at the same time simplified management operations like setting a total allowable catch limit. A substantial part of this simplification stems from decisions being entrenched in the realms of science and policy.

Harvest control rules in practice

In the following, we discuss in some detail six different fisheries that are governed by HCRs. We provide some background on the fisheries, describe the HCRs, and discuss some related aspects (objectives, indicators, instruments, uncertainty). We also briefly discuss whether the HCRs are adhered to and how the fisheries developed after the HCRs were adopted. The fisheries are Northeast Arctic cod, North Sea cod, Norwegian spring spawning herring, Icelandic capelin, Pacific sardine, and Alaska groundfish. These fisheries were chosen to illustrate different implementations of HCRs that ranges from straightforward relationships between the estimated spawning stock biomass (SSB) and allowable catch (Northeast Arctic cod; Norwegian spring spawning herring) to complex rules that take into account environmental factors and ecosystem considerations (Pacific sardine, Alaska groundfish).

Before going into details of HCRs in practice, we explain some technical terms that may be foreign to readers outside the field. The ICES Advisory Committee on Fisheries Management (see ICES, 2001) established fisheries management reference points in 1991. Limit reference points indicate states considered undesirable; target reference points indicate states considered desirable. The biomass limit, denoted B_{lim}, indicates the spawning stock biomass level under which it is undesirable that the spawning stock declines because, for example, recruitment is likely impaired at low stock levels. The fishing mortality limit, denoted F_{lim}, is the fishing mortality that is associated with B_{lim} (in the sense that if F_{lim} was implemented over a longer period, biomass would be expected to approach B_{lim}). Maximum sustainable yield (MSY) is a classical target in fisheries management, and related biomass and fishing mortality reference points are denoted B_{MSY} and F_{MSY}. Further, there are precautionary reference points: the biomass precautionary limit, B_{pa}, dependent on assessment uncertainty, is the biomass level at which there is a 95% probability that the spawning stock biomass is above B_{lim}; the fishing mortality precautionary limit, F_{pa}, is the fishing mortality that is associated with B_{pa} (in the sense that if F_{pa} was implemented over a longer period, spawning stock biomass would with 95% probability be above B_{lim}; that is, at B_{pa}).

Northeast Arctic cod

The Northeast Arctic cod, the world's largest cod stock, lives and feeds in the Barents Sea and spawns along the northern Norwegian coast. The Joint Norwegian-Russian Fishery Commission manages the fisheries (both in the Barents Sea and on the spawning grounds). The origins of the Norwegian-Russian cooperation in marine research and fisheries date back to the start of the 1900s. However, it was not until 1975 that Norway and Russia reached an agreement regarding cooperation in fishing activities, which initiated the joint management of the most important fishing stocks in the Barents Sea: cod, haddock and capelin. A new treaty between Norway and Russia on maritime delimitation and cooperation in the Barents Sea and the Arctic Ocean, which came into force on 7 July 2011, stipulates that the Joint Norwegian-Russian Fisheries Commission shall continue to assess improvements to surveillance and control measures for the jointly managed fish stocks (The Cooperation Agreement between Norway and Russia of 1975 and the Agreement regarding joint relationships of 1976). That the fish stocks in the Barents Sea are shared between the two bordering nations is now, in other words, an incontestable fact.

Total allowable catch (TAC) of Northeast Arctic cod has been determined according to a HCR-system since 2004 (ICES, 2008; Eide et al., 2013). The current HCR was first agreed to between Russia and Norway in the autumn of 2002, and was first applied when setting quotas for 2004. Reference points and the Code of Conduct related to the precautionary approach were subsequently fine-tuned (in 2004 and 2009) based on experience and new knowledge. According to assessments by ICES, the HCR is in agreement with the precautionary principle and is intended to provide predictability for the fishing industry, via more stable total allowable catch levels, to create conditions for high yields in the long run, and to simplify the annual negotiations on harvest levels (Yaragina et al., 2011). Eikeset et al. (2013) found that the current HCR best approximated a profit maximizing harvest rule when compared with some alternative schemes (welfare maximization; yield maximization).

The HCR dictates a fishing mortality of 0.4 (F_{pa}) when the spawning stock is above the precautionary stock level of 450 thousand tonnes; at lower stock levels, the fishing mortality declines linearly to zero (see Eikeset et al., 2013, and references therein). The HCR has a number of provisions, however. The total allowable catch (TAC), for example, is a three-year average of the estimated catch from the fishing mortality scheme. Further, catches cannot change more than 10% from year to year, a provision that incidentally was set aside to increase the 2009 catches. If the HCR implies a fishing mortality below 0.3, catches are increased to correspond to a fishing mortality of 0.3. Finally, if the spawning stock is, has just been, or is predicted to be, below the precautionary level, no restrictions apply on the year-to-year variation in total catches.

Table 1 gives an overview of scientific advice, agreed TACs, and reported catches from 2000 to 2016, including estimates of unreported catches. After the implementation of the HCR (2004), particularly as the problem of unreported landings was solved (2007/2008), the overall compliance with the HCR has been good (ICES, 2014). There are at least two explanations for this compliance. First, the period after the

Table 1. Scientific recommendations, total quotas, and reported and unreported landings of Northeast Arctic cod 2000–2016[a]

Year	ICES advice[b]	Predicted catch corresponding to ICES primary advice	Agreed TAC	ICES landings	Unreported landings[c]
2000	Increase B above B_{pa} in 2001	110	390	415	-
2001	High probability of SSB >B_{pa} in 2003	263	395	426	-
2002	Reduce F to well below 0.25	181	395	535	90
2003	Reduce F to below F_{pa}	305	395	552	115
2004	Reduce F to below F_{pa}	398	486	606	117
2005	Take into account coastal cod and redfish bycatches. Apply catch rule.	485	485	641	166
2006	Take into account coastal cod and redfish bycatches. Apply amended catch rule.	471	471	538	67
2007	Take into account coastal cod and redfish bycatches. Apply F_{pa}	309	424	487	41
2008	Take into account coastal cod and redfish bycatches. Apply catch rule.	409	430	464	15
2009	Take into account coastal cod and redfish bycatches. Apply catch rule.	473	525	523	0
2010	Take into account coastal cod and redfish bycatches. Apply catch rule.	577.5	607	610	0
2011	Take into account coastal cod and redfish bycatches. Apply catch rule.	703	703	720	0
2012	Take into account coastal cod and redfish bycatches. Apply catch rule.	751	751	728	0
2013	Take into account coastal cod and S. marinus bycatches. Apply catch rule.	940	1000	966	0
2014	Take into account coastal cod and S. marinus bycatches. Apply catch rule.	993	993	986	0
2015	Take into account coastal cod and S. norvegicus bycatches. Apply catch rule.	894	894	-	-
2016	Take into account coastal cod and S. norvegicus bycatches. Apply catch rule.	805	-	-	-

[a]ICES (2015b)
[b]B is biomass, SSB is spawning stock biomass, pa is precautionary approach, and F is fishing mortality.
[c]Unreported landings are included in ICES landings
doi: 10.12952/journal.elementa.000114.t001

implementation has been characterized by a strong and sustainable cod stock, possibly caused by favorable temperatures, and the healthy stock has likely allowed relatively generous catches. Second, the close fit between ICES' primary advice and the agreed TACs is perhaps a consequence of more precaution on behalf of the Joint Norwegian-Russian Fisheries Commission.

While the current HCR for the Northeast Arctic Cod has no explicit recognition of ecosystem effects, the interaction of cod with capelin and herring in the Barents Sea is well established (Gjøsæter et al., 2009). Norway has taken a number of steps to implement an ecosystem-based management scheme for the Barents Sea (Olsen et al., 2007). For example, Barents Sea capelin stock assessment scientists have for a number of years estimated cod predation (Gjøsæter et al., 2015) and has developed at least the scientific basis for including the cod-capelin interaction in cod management. Given that capelin and cod are, typically, harvested by different fishing vessels and that Russia and Norway may value these stocks differently, it would require substantial political will to include the cod-capelin interaction in the HCR.

North Sea cod

Cod in the North Sea is found from Skagerrak in the east to the English Channel in the west and to Shetland in the north. Both the EU and Norway pursue fisheries, and regulation depends on bilateral agreements. From the EU, mainly fleets from the UK, the Netherlands, Germany, and Denmark participate in the fisheries.

The legal basis for regulation of fisheries in EU waters was established by the Common Fisheries Policy in 1983 (EU Council regulation no. 170/83), after which scientific advice was used to set TACs for the North Sea cod. In 1999, the EU and Norway established a management plan with the objective to keep the total stock biomass above 1,500 thousand tonnes and fishing mortality (F) at 0.65. Notably, the stock level had at that time been declining since the 1970s and was at an all-time low. Despite biological advice to reduce fishing effort and since 2001 to place a moratorium on the North Sea cod fishery, fishing continued in conflict with the reference points. In 2002, the legal basis for implementing recovery plans for fish stocks in EU waters was established (EU Council Regulation no. 2371/2002). The regulation calls for recovery plans for fish stocks outside of safe biological limits, with the long-term goal of bringing stocks within these limits. The first recovery plan for the North Sea cod was adapted in 2004 (EU Council Regulation 423/2004). The objective was to bring the stock level above B_{pa}. Further, a number of rules applied to the setting of TACs. Among the rules were that fishing mortality was not to exceed the precautionary rate of 0.65 when above B_{lim}, and less when below B_{lim}, and that year-to-year changes in TACs were not to exceed 15%. Another rule was that the spawning stock biomass was to increase 30% per year (understood to be in force until the objective of reaching B_{pa} was attained). All these rules may conflict with each other. Limited year-to-year changes in TACs were politically and economically motivated and turned out to have prevalence: finally in 2007, fishing mortality was reduced below 0.65 and the stock recovered from levels below B_{lim}.

In 2008, another long-term management plan was adopted (EU Council Regulation 1342/2008). The harvest control rules were quite simple: if the stock level is above 1,500 thousand tonnes, set fishing mortality to 0.4 (equal to F_{MSY}); if below B_{lim}, set fishing mortality to 0.2, and at intermediate stock levels, the fishing mortality is to be set as the linear combination of the two (see red lines in Figure 2). Whereas the recovery plan was in force while the stock was below B_{pa}, the long-term plan pursued maximum sustainable yield. The simple rules were, however, convoluted by restrictions on how fast the TAC was to be reduced and, after 2010, no more than 20% year-to-year changes in TACs were allowed. The gradual approach was termed a phase-in period that should last until 2014.

Figure 2 shows the development of the North Sea cod fishery against its reference points, from 1963 to 2013. Before the common fisheries policy in 1983, fishing mortality increased steadily while the stock level declined slowly from a high around 1970 (green dots). From 1983, TACs were set without formal rules but informed by scientific advice (yellow dots). Fishing mortality remained high while the stock level declined to historic lows. After 2000, under strong scientific advice to reduce fishing effort, the fishing mortality has declined slowly, and after the first recovery plan (2004) stayed below F_{lim} (blue dots). The years after 2000 saw strong reductions in TACs. Under the new management plan (2008), fishing mortality has continued to decline and has stayed below the precautionary rate (red dots). The stock has increased and, recently, the spawning stock level has almost reached safe biological limits (148.896 tonnes in 2015, ICES, 2015a).

The regulatory history of the North Sea cod fishery suggests that reducing TACs, however necessary, is difficult. The regulatory system's inability to properly manage the resource before the introduction of an HCR in form of the recovery plan is likely caused by how decisions were made. The EU Council that sets the TACs is composed of cabinet ministers for fisheries from EU member states. The ministers have to secure national interests and maintain public approval among their constituents. Reduced quotas, together with increased technical efficiency and progress, usually lead to overcapacity in national fishing fleets. Overcapacity creates national pressure for higher quotas, and ministers have difficulty in making decisions that prioritize the long-term productivity of the fish stock over short-term national interests (Ravn-Jonsen, 2004; see also Daw and Gray, 2005 and Grainger and Parker, 2013).
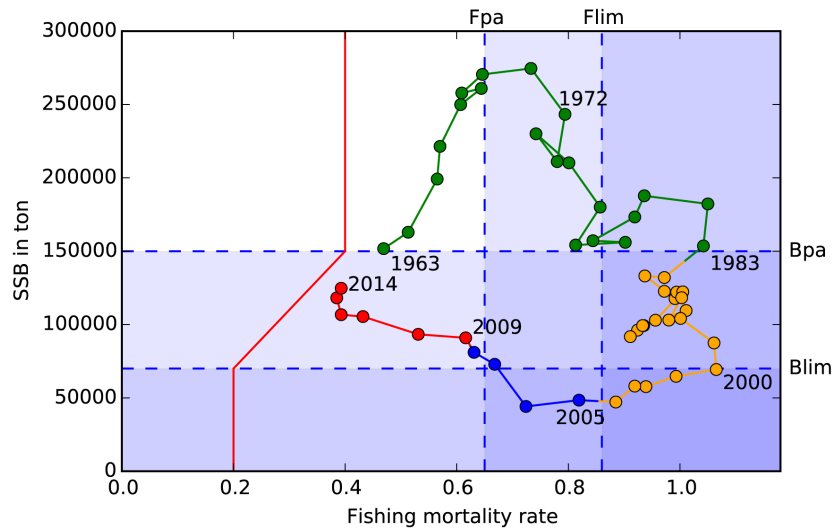
The development in recent years holds promise, however, that ultimately, when the situation becomes severe enough, the Council is able to commit to long-term recovery plans. A key feature of the plans is that they combine stock conservation with predictability in quotas. The plans signal expected developments in TACs to the national political stage, and the mere existence of plans reduces political pressure for action to secure national interests. Thus, the system has to some degree been liberated from prioritizing short-term demands, and the most recent plans are regarded as successful. For the first time in twenty years, the spawning stock biomass of the North Sea cod has, according to the latest reports (ICES, 2015a), reached B_{pa}. Although the HCR regime has not been successful in reaching maximum sustainable yield, which will take many more years, it has been successful in controlling the political management process.

Norwegian spring spawning herring

The Norwegian spring spawning herring is a widely distributed and highly migratory pelagic fish stock that covers large parts of the Northeast Atlantic during its lifespan. The migration pattern is flexible and varying, and depends on ever-changing hydrography and variable feeding opportunities (Holst et al., 2004). The stock moves through a number of exclusive economic zones during its migrations, and the fishery is regulated by the coastal states (that is, the European Union, Faroe Islands, Iceland, Norway, and the Russian Federation). TACs are derived from an agreed long-term management plan, and the coastal states agree on the national allocation of the TACs.

Following heavy exploitation including targeting juveniles, the Norwegian spring spawning herring collapsed during the 1960s (Fiksen and Slotte, 2002). A fishing moratorium was put in place, and for many years, the collapsed stock did not migrate outside of the Norwegian economic zone. After several years, the fishery was again opened. A minimum landing size of 25 cm was introduced, in addition to strict management measures to control the total catch (Tjelmeland and Røttingen, 2009). The available stock-recruitment information suggested that the spawning stock biomass should be at least 2.5 million tonnes to secure sufficient recruitment, and the management objective at the time was to rebuild the stock to this level. Notwithstanding and against all predictions, the 1983 year class – spawned from a spawning stock biomass of a mere 500 thousand tonnes – was exceptionally large. The 1983 event nicely illustrates the uncertainty that permeates fisheries management and underlines the need for adequate management practices. The 1983 year class helped rebuild the stock, and by the mid-1990s, the spawning stock biomass again exceeded 2.5 million tonnes.

A rebuilt herring stock made the rebuilding scheme (with a fishing mortality of 0.05) obsolete, and a new strategy was called for. The 1996 report from the advisory process, when medium-term simulations came into use for policy advice (Bogstad et al., 2000), mentions 2.5 million tonnes as a minimum biologically acceptable spawning stock biomass level (ICES, 1997). Medium-term simulations indicated a low probability of the spawning stock biomass falling below 2.5 million tonnes if the fishing mortality is set at 0.15 and the upper catch limit is set at 1.5 million tonnes (ICES, 1997). Later, the fishing mortality of 0.15 was associated with too high a risk of the spawning stock biomass falling below 2.5 million tonnes. In 2001, the coastal states agreed on a management plan that included a recovery plan. The fishing mortality was to decline linearly from 0.125 at 5 million tonnes SSB (B_{pa}) to 0.05 at 2.5 million tonnes (B_{lim}). The plan was reevaluated in 2013 (ICES, 2013) and has remained in use until present. The stated objectives are to keep the spawning stock biomass above B_{lim} while securing relatively high and stable yields in the long run.
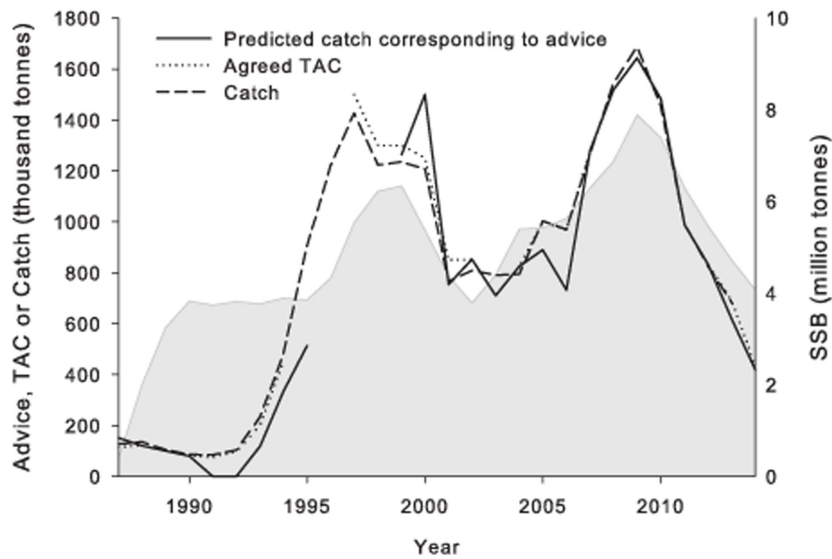
As the larger stock again migrated outside of the Norwegian economic zone, international agreements also became necessary. The story of fishing agreements and conflicts in the Northeast Atlantic is long and beyond our scope to delve into fully here. The coastal states had a working agreement about the allocation of the fishery from 1995 to 2002. Another agreement was signed in 2007. The new agreement partially broke down in 2013. In 2013, the coastal states agreed to carry out a biological investigation into the distribution of the herring throughout its life cycle, as zonal attachment can be used as basis for sharing quotas between fishing nations – longer residence in national waters would translate into higher shares of the total quota. However, the report - rather than furnishing agreement – has seemingly led to further disagreement, among other things over how to interpret the report. But, agreement or not, compliance to the biological advice has seemingly been good (Figure 3). This conclusion presupposes that catch statistics are reliable, but there are no clear indications of substantial misreporting or illegal fishing. The HCR has thus at least seemingly improved the management of Norwegian spring spawning herring in two ways: first by setting clear targets for management (F not to be exceeded while keeping biomass over a certain level), and secondly by making the international negotiations less intense because the TAC value has been defined by the HCR. However, it is difficult to judge how large an impact the serious collapse of the stock has had on the willingness of managers to find agreements and to comply with the biological advice.

Icelandic capelin

Capelin is a small short-lived pelagic fish with subpopulations throughout the Arctic region, with significant stocks in the Barents Sea and near Iceland. It has been fished in Icelandic waters since the mid-1960s, first as a winter fishery only. By the late 1970's, the fishery had expanded both geographically and seasonally, and had become multinational as well. The fishery is mostly based on the maturing part of the stock. Currently, the fishing season extends from late June to the end of March, but in recent years there has been practically no fishery during the summer months. The stock is managed jointly by Iceland, Norway, and Greenland.

During the 1970s, it became clear that regulation was necessary, both to secure a sustainable fishery and because of the role of capelin as a forage fish. The need for management is due to the highly variable recruitment, short life span and the fact that high catch rates may be maintained even at low stock levels. Therefore the primary objective of the management is to prevent the stock from being fished down to a level of reduced recruitment.

In 1979 – until which time the fishery was essentially open access – regulations with a target of leaving 400 thousand tonnes to spawn was established (Gudmundsdottir and Vilhjálmsson, 2002; Vilhjálmsson and Carscadden, 2002). This level was on par with the management of the Barents Sea capelin at the time, which required a spawning stock of 500 thousand tonnes for what was considered to be a larger stock subject to higher predation (Gudmundsdottir and Vilhjálmsson, 2002; Vilhjálmsson and Carscadden, 2002). The Icelandic capelin has been acoustically surveyed since 1978. Already in 1979, it was recommended that the fishing season should open with a low, precautionary quota, which would then be revised upwards after an assessment survey of the mature stock within the fishing season (Vilhjálmsson, 1994). This plan was, however,
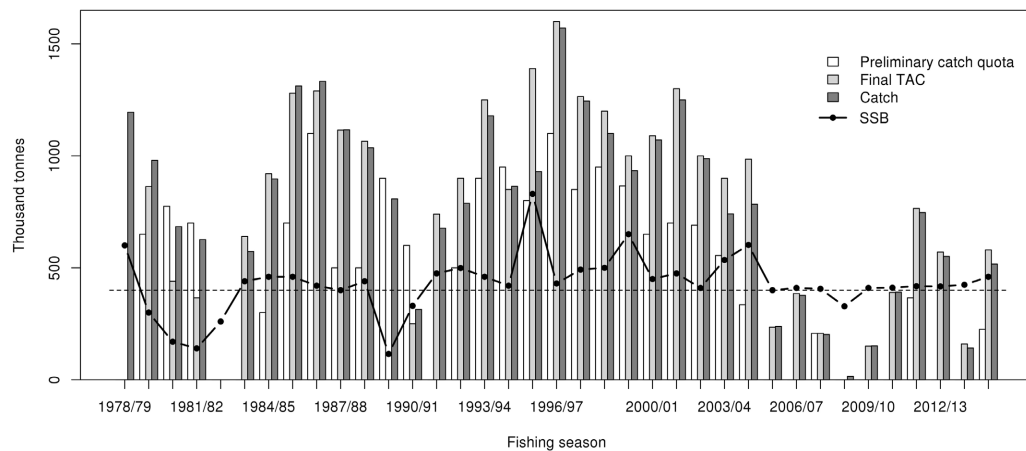
not adopted and preliminary catch quotas were set without adequate knowledge of the condition of the stock. The resulting catches were probably too large; the spawning stock collapsed and a fishing moratorium was put in place in 1982 (Vilhjálmsson and Carscadden, 2002).

After the moratorium, a more cautious management plan was adopted (Gudmundsdottir and Vilhjálmsson, 2002). Preliminary catch quotas were based on acoustic surveys in August of the year before fishing commenced, while a final TAC was determined after acoustic surveys of maturing adults, usually carried out during the spawning migration in January. The principal regulation was still to leave 400 thousand tonnes to spawn under the assumption that the previous problems had to do with how to determine an appropriate preliminary quota and not the harvest control rule as such. Another moratorium was put in place in the early 1990s when the spawning stock collapsed again. When the fishery reopened, the basis for preliminary quotas was changed (existing October surveys were used) while the harvest control rule itself was kept. As earlier, the focus was on how to improve the information on which the preliminary quota was based.

In the early 2000s, the Icelandic capelin fishery was again in dire straits as recruitment was lower than expected and preliminary quotas could not be set on the basis of juvenile indices from the autumn surveys. Recent research suggests that changes in the distribution, which were related to climate changes, affected the results from the autumn surveys (Palsson et al., 2012; Carscadden et al., 2013). Not until 2010, when the survey area was increased and the autumn surveys covered the full distribution of the stock, was it again possible to set a preliminary TAC. The consequences of not having a preliminary quota were that the fishery was not opened until after the adult stock acoustic surveys in late autumn or winter. However, the stock has been at lower levels in the 2000s than during the mid-1990s and the landings have been much smaller (Figure 4).

The principal harvest control rule for the Icelandic capelin, to maintain a spawning stock of 400 thousand tonnes, has not changed since 1979. Thus, harvest control rules have been in practical use before the general institutional framework came into place during the 1990s, as also seen in the Scientific Committee of the International Whaling Commission. The rule has been implemented via a two-step management plan with preliminary quotas that were adjusted when further information arrived. The government has followed the advice based on the HCR. Quotas are allocated through an ITQ system with all of its general control and enforcement provisions. By and large, the management plan has been successful in sustaining a sufficient spawning stock, in particular if the low stock levels of the 2000s are attributed to climate changes that lie outside the scope of the current harvest control rule and management plan. As such, the current harvest control rule failed to realize its full potential in considering environmental changes. It is perhaps pertinent to mention here that the Barents Sea capelin has a history of repeated collapse and recovery, and these collapses are not necessarily tied to fishing but to changes in the ecosystem (Gjøsæter and Bogstad, 1998). It thus seems that capelin is a type of fish that has an extraordinary sensitivity to environmental and ecological conditions, and that appropriate HCRs need to take this into account.

The harvest control rule for the Icelandic capelin has been deterministic in the sense that point estimates of the spawning stock biomass have been taken at face value; that is, no provisions were made for empirical or structural (model) uncertainty. Recently, a new harvest control rule was introduced, again modelled after the harvest control rule for the Barents Sea capelin (Gjøsæter et al., 2015), where empirical uncertainty is taken into account. The new rule also reflects predation on capelin that takes place between the time of the acoustic measurements of the spawning migration and the spawning season. The final TAC will be set such that the probability of a spawning stock below a biomass limit of 150 thousand tonnes is less than 5%. The new HCR is considered to be in agreement with the precautionary approach. This change in formulation is expected to be more conservative in terms of harvest strategy than the earlier method, mostly because of higher

estimated predation. The median of the probability distribution for the spawning stock level is expected to be close to 400 thousand tonnes. Climatic considerations have until now not been taken into account in the harvest control rules despite evidence of impacts on capelin biology and distribution from climatic changes.

Pacific sardine

Pacific sardine are small schooling fish that have at times been the most abundant fish in the California Current ecosystem, which extends from the coast to a thousand kilometers offshore, from Oregon (United States) to Baja California (Mexico). When the Pacific sardine stock is large, it can be abundant from southern Baja California to southeastern Alaska (United States), but when small, sardines may not be prevalent north of Baja California. The fishery for Pacific sardine operates in Mexico, the United States, and Canada. The Pacific sardine is also an important forage fish for larger fish, seabirds, and marine mammals in the California Current ecosystem.

Pacific sardine abundance is cyclical, and boom-bust dynamics have been documented in sedimentary records over hundreds of years (Baumgartner et al., 1992). Periods of high and low abundance of Pacific sardine align with warm and cold periods on a 50-60 year cycle that is associated with the Pacific Decadal Oscillation (Chavez et al., 2003; Zwolinski and Demer, 2012). Water temperatures have long been thought to affect sardine population dynamics, and quantitative analyses confirmed that sardine recruitment was related to not only spawning biomass but also sea surface temperature (Jacobson and MacCall, 1995a, 1995b; Jacobson and McClatchie, 2013).

Since 2000, the US Pacific Fishery Management Council has managed the fishery for the northern sub-population of Pacific sardine under the Coastal Pelagic Species Fishery Management Plan. Management is based on annual catch limits that are set based on an HCR that considers ocean conditions and reduces the catch limit as biomass declines. The HCR is formulated as: $H = (B − C)f\,d$, where H is the (US) harvest target level, B is the estimated biomass of age 1 and older, and the cutoff C is the lowest level of biomass at which directed harvest is allowed; f is the fraction of the biomass above the cutoff that can be taken by the fishery, and d is the average portion of biomass assumed in US waters (PFMC, 2011a). The cutoff C is set to 150 thousand tonnes, and its purpose is to protect the stock when the biomass is low; spawning biomass is the strongest explanatory factor of recruitment of Pacific sardine (Jacobson and MacCall, 1995a). That the harvest level declines with declining biomass accounts for uncertainty in the biomass estimate, which can have a coefficient of variation of as much as 50% (PFMC, 2011a). The cutoff buffer and reduced harvest rate at low biomass levels together protect a portion of the spawning stock from fishing to support rebuilding, but also to provide forage for other species (PFMC, 2014).

The harvest rate is also controlled by the fraction f, which is a proxy for F_{MSY} that incorporates recent ocean temperatures (averaged over the previous three years). Regardless of the relationship with F_{MSY}, however, f is never set higher than 20% and never below 5% (PFMC, 2014; Hill et al., 2015). Ocean temperatures are included in F_{MSY} to scale productivity of the sardine stock, which is higher under warmer conditions and lower under cooler temperatures. The relationship between F_{MSY} and temperature is given as follows: $F_{MSY} = −18.46452+3.25209\ T−0.19723\ T^2+0.0041863\ T^3$, where T is the average sea surface temperature during the three preceding seasons (given in degrees Celsius). If f is approximately equal to F_{MSY}, the HCR will not allow harvesting above F_{MSY}. The distribution parameter d is set to 87%. Ultimately, the US harvest level of Pacific sardine is not allowed to exceed 200 thousand tonnes. This upper limit on harvest reduces risk related to errors in estimated biomass; it also reduces annual variation in catch and deters overcapitalization during short periods of high abundance. The upper limit prevents catch from exceeding assumed maximum sustainable yield, and spreads benefits of strong year classes over time (PCFM, 2014).

The relatively sophisticated HCR for Pacific sardine demonstrates that environmental conditions, stock protection and ecosystem considerations can be accounted for in determining harvest levels in an HCR framework. The HCR has been modified slightly over time (Figure 5). A flawed reanalysis of the recruitment-temperature relationship (McClatchie et al., 2010) resulted in a temporary removal of temperature from the HCR in 2011 (Hill et al., 2011; PFMC, 2011b), but following additional analyses, temperature reentered in 2014 (Jacobson and McClatchie, 2013; Lindegren and Checkley, 2013). When the temperature link was reinstated, it relied on a different sea surface temperature index that provides a better relationship to sardine productivity (PFMC, 2014).

The Pacific sardine HCR has been adhered to in management of the stock (Figure 5) and has proven useful for balancing fishing pressure in the context of varying environmental conditions. In April 2015, the sardine fishery was rapidly closed when biomass estimates dropped below the cutoff value (PFMC, 2015). However, reliance on a similarly complex measure may be difficult to achieve for other fisheries. Myers (1998) recognized Pacific sardine as a unique case of a relationship between recruitment and the environment being incorporated into an assessment that is used to provide tactical management advice. More recently, Szuwalski et al. (2014) recognized Pacific sardine as an example of a species in which recruitment strongly influences spawning stock biomass. The tight link between temperature, recruitment, and biomass observed in Pacific sardine may favor use of a HCR that incorporates ecosystem and population factors, but the approach may not be useful in other stocks where these links are weaker or less consistent over time.
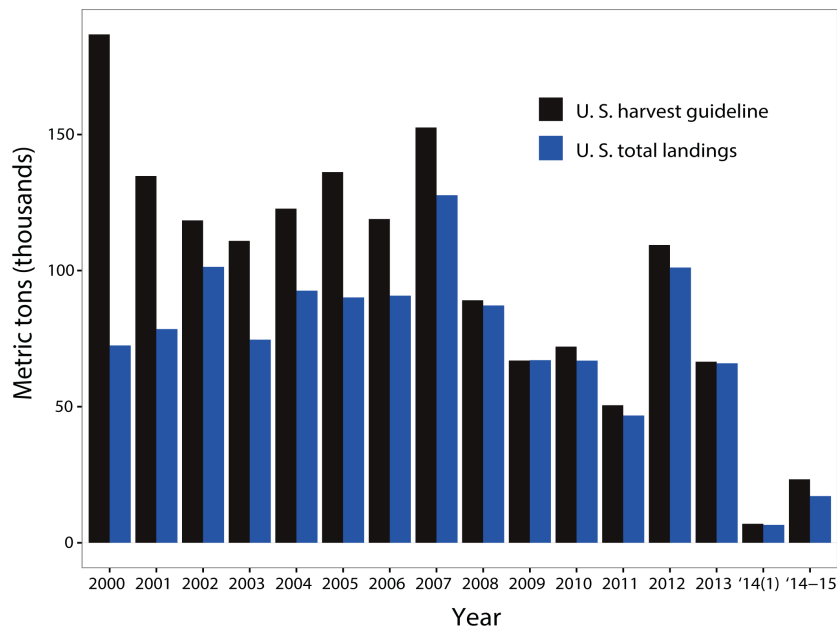
Alaska groundfish

The primary legislation governing management of Alaska's groundfish fisheries is the Magnuson-Stevens Fishery Conservation and Management Act (Magnuson-Stevens Act), originally enacted in 1976. The Act contains 10 national standards to guide fishery management, including the mandate that "[c]onservation and management measures shall prevent overfishing while achieving, on a continuing basis, the optimum yield from each fishery for the United States fishing industry", where optimum yield is defined in terms of providing "the greatest overall benefit to the Nation" and was originally prescribed "on the basis of the MSY from the fishery, as modified by any relevant economic, social, or ecological factor". In 1997, 'as modified by' was changed to 'as reduced by', implying that MSY is to be considered a limit rather than a target reference point, consistent with the precautionary approach (e.g. UN, 1995). For federally managed groundfish fisheries off Alaska, Fishery Management Plans have been developed for the Gulf of Alaska region (NPFMC, 2015a) and for the Bering Sea/Aleutian Islands region (NPFMC, 2015b). Both plans include the same general HCRs, which differ depending on information availability and have been modified for individual species based on ecosystem considerations as illustrated below for walleye pollock (*Gadus chalcogrammus*).

Major commercial fisheries for groundfish off Alaska developed in the 1950s when Japanese fishing resumed after World War II and the Soviet Union developed its distant water fleet. These fisheries were virtually unregulated until 1965 and were only minimally regulated thereafter through bilateral agreements. The first catch limits for pollock and flatfish were implemented in 1973. With the passage of the Magnuson-Stevens Act in 1976 the US established fishery jurisdiction out to 200 miles from shore, which among other things led to a transition from foreign fisheries to a fully domestic fishery by 1991. Thus the domestic groundfish fleet developed at the same time as fishery management plans with HCRs for these groundfish fisheries were developed by the North Pacific Fishery Management Council, one of eight regional councils established by the MSA. Coincident with the passage of the MSA, changes in temperatures and large-scale circulation patterns associated with the 1976/77 climate regime shift increased the productivity of some pelagic and demersal groundfish stocks such as walleye pollock, Pacific cod (*Gadus macrocephalus*), and flatfish. Together, these events created a favorable environment for groundfish that contributed to building trust among the fishing industry, scientists, and managers as the regulatory framework matured and was modified by numerous amendments developed through an open and transparent process.

Under the unified framework, as defined in the groundfish Fishery Management Plans, general harvest control rules are specified but they vary depending on the information available (NPFMC, 2015a, 2015b). Regardless of information availability, an overfishing limit that is based on MSY considerations is established. For species with adequate information, the fishing mortality corresponding to the overfishing limit (F_{OFL}) is set equal to F_{MSY} when biomass is high and decreases linearly with spawning biomass when it is below the biomass corresponding to MSY (B_{MSY}, Figure 6). No directed fishery is allowed below a minimum biomass threshold, currently set to 5% of B_{MSY}. To determine the overfishing limit for the current year, F_{OFL} is applied to the current biomass. To account for uncertainty in overfishing limit determinations, the maximum Acceptable Biological Catch (ABC) is set below the overfishing limit, at a level that is deemed safe to prevent overfishing (defined as catch above the overfishing limit) and to keep the stock from being overfished (defined as biomass
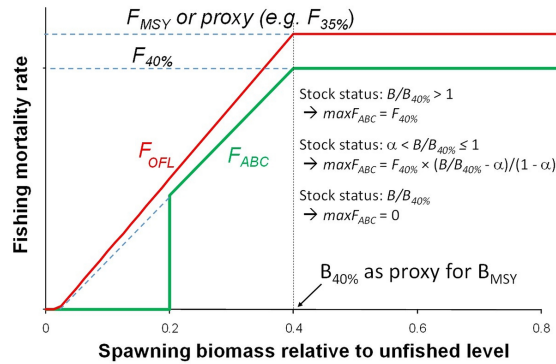
being below a specified threshold). Calculations for data-rich stocks use direct estimates of B_{MSY} and F_{MSY} to specify harvest levels and determine stock status; in these stocks the buffer between F_{ABC} and F_{OFL} is directly linked to uncertainty in F_{OFL}. If MSY-based indicators are not available, proxies based on spawning potential ratio (Mace and Sissenwine, 1993) are used, namely $F_{35\%}$ for F_{MSY} and $F_{40\%}$ for calculating F_{ABC}. The shape of the HCR is designed to maintain spawning biomass near or above the level corresponding to B_{MSY}.

In the US context, each management council's Statistical and Scientific Committee has final authority to specify ABCs based on the best available scientific information. This includes the authority to set ABCs below the maximum permissible ABCs that result from the control rule in cases where conservation concerns, ecosystem considerations, un-accounted for uncertainties, or other considerations warrant a more precautionary approach. Once ABCs have been specified, the Council sets TACs at or below the scientifically established ABCs based on bycatch considerations, management uncertainty, or socioeconomic considerations. Should catches exceed ABCs, accountability measures as specified in Fishery Management Plans are invoked. By assigning the authority to set ABCs to set ABCs to a scientific committee, potentially consequential choices about model structure and parameter values are largely shielded from political influence.

Special provisions that further reduce catches below the maximum permissible ABC have been implemented to account for ecosystem considerations. One of the most consequential provisions was the establishment of ecosystem-wide optimum yield ranges that effectively establish upper limits on groundfish removals because the sum of individual TACs must not exceed the upper optimum yield limit, which is set at 2 million tons in the Bering Sea / Aleutian Islands region and 0.8 million tons in the Gulf of Alaska (Witherell et al., 2000). In the Bering Sea, but not in the Gulf of Alaska, the sum of single-species ABCs generally exceeds the upper optimum yield limit and has been as high as 2.8 million tons (Witherell et al., 2000; Mueter and Megrey, 2006). Consequently, many stocks, particularly flatfish and walleye pollock, have been exploited well below sustainable levels (Witherell, 1995). Other ecosystem considerations have led to a modified harvest control rule for certain groundfish species that are important prey for Steller sea lions (*Eumetopias jubatus*). The modified rule prohibits directed fishing when the spawning biomass of these species, including walleye pollock (Figure 6) is projected to fall below $B_{20\%}$. This modification was enacted to protect Steller sea lions, which were listed as endangered in 1997 under the US Endangered Species Act. The modification remains in effect but has not constrained fishing to date. While broader ecosystem considerations do not formally enter harvest specifications through the HCRs, on at least one occasion such considerations resulted in a substantial reduction in the ABC of walleye pollock from the maximum permissible ABC. This reduction was prompted by concerns over the negative effects of poor prey conditions on walleye pollock, following a series of unusually warm years from 2001–2005 that were associated with changes in zooplankton availability and poor pollock recruitment (Hunt et al., 2011).

While the basic structure of the HCRs has not changed and provides clear guidance for setting harvest limits, the actual outcomes are highly model dependent. Stock assessment models are reviewed and updated annually or biennially and indicators relating to biomass and fishing mortality can be highly sensitive to model structure and to the values of independently estimated parameters. Therefore different model structures can potentially lead to vastly different ABCs. Although there are no formal constraints on year-to-year changes in ABCs, a multi-layer review process and an implicit preference for incremental changes has typically resulted in modest changes to ABCs and TACs, sometimes through an ad-hoc adjustment such as a stair-step approach. However, in some cases ABCs have been reduced drastically when conditions indicated a conservation concern.

Of 35 federally managed groundfish stocks off Alaska whose status can be assessed, none were being overfished or in an overfished condition in 2015. While many factors have contributed to the successful management of Alaska's groundfish stock, the early development of HCRs and strict adherence to these rules, combined with effective accountability measures (NPFMC, 2015a, 2015b), have undoubtedly played a major role in this success.

Discussion

The cases discussed in the previous section demonstrate some of the versatility of the HCR concept. As mentioned, implementations span from rather simple rules connecting spawning stock and allowable catches (Northeast Arctic cod, Norwegian spring spawning herring) to more complex arrangements involving environmental conditions (Pacific sardine) and, potentially, foodweb effects (Icelandic capelin, Pacific sardine, Alaska groundfish). The cases also demonstrate how HCRs can incorporate both recovery plans and long-term plans (North Sea cod, Norwegian spring spawning herring), and that HCRs in practice have been used prior to establishment of the general, globally acknowledged framework. This diversity in part reflects different biological, institutional, and political environments. Yet, the cases we have discussed span only a small subset of the full, potential policy space that HCRs represent. The different cases also feature a number of similarities. For example, all cases except Alaska groundfish involve more than one fishing nation (notably, the Pacific sardine HCR is only used by the US), and all cases rely on modeled indicators. Of particular interest are provisions in some of the HCR schemes for limitations in year-to-year changes in catches; such provisions are present for both Northeast Arctic cod and North Sea cod, notably both demersal cod stocks, but not for the remaining stocks.

HCRs are implemented in a number of fisheries besides those discussed above and are, as our preceding discussion should make clear, viewed as a step forward in fisheries management. Furthermore, HCRs have been heralded as vehicles to introduce environmental and ecosystem effects into operational fisheries management. But among the cases we discuss above, only the Pacific sardine HCR considers environmental measures, despite evidence of abundance-temperature relationships for most of the stocks (Hannesson, 2007); the tendency of not observing environmental measures is likely not confined to our cases. These circumstances notwithstanding, model-based management remains at the center of attention and is, for example, presently referred to as 'the standard fisheries management science paradigm' (Dankel et al., 2016, p. 310). While HCRs transpires to play a central role in future fisheries management, a number of issues require consideration.

A dominating feature of fisheries management, past, present, and future, is that of pervasive uncertainties in our knowledge of ecosystems and their structure, and of related economic systems (generally, natural resource systems; see Peterman and Anderson, 1999). For example, Schnute and Richards (2001) classify errors in fisheries models into three categories: Process error (inconsistencies in recruitment, mortality, growth, or distribution), measurement error, and model uncertainty. While appropriate application of statistics can in theory deal rigorously with the first two types of errors, model uncertainty is not subject to quantification. As discussed above, coping with process and measurement errors via more data collection, further sophistication of models, or ad hoc assumptions have only to a limited degree reduced the overall uncertainty, while the resulting framework for fisheries management advice has become complex and incomprehensible to policy makers and stakeholders. The advent of HCRs has not reduced or made uncertainties in fisheries models less important, but has shifted the focus from model-based catch recommendations to validity and relevance of modelled indicators that serves as basis for rule-based management; that is, the focus has shifted one step away from the policy advice.

Uncertainty in fisheries management goes beyond uncertainty in fisheries models and is particularly relevant for frictions in the science-policy interface (see Dankel et al., 2012, and references therein). A salient feature is that of understanding and communicating scientific uncertainty (Lane and Stephenson, 1995; Kelly and Codling, 2006); that is, problems with understanding and communicating uncertainty add layers of uncertainty to the necessary interchange between the scientific and the political domain. Rowe (1994), in his general classification of uncertainty, calls this translational uncertainty and highlights the importance of recognizing differences in perspectives on uncertainty when communicating scientific uncertainty. At the same time, policy makers and stakeholders should strive to understand the complex analyses that are inherent to fisheries management and as such take part in the scientific perspective (Stringer et al., 2009). A related feature is the tendency of industry to broadly interpret uncertainty to mean that conservative or intervening policy measures are unwarranted (see, for example, Clark, 2006). As discussed by Dankel et al. (2012), this may lead scientists to underemphasize uncertainty, something that may undermine legitimacy and credibility of science in the long run.

Further challenges with HCR schemes that connect to uncertainty, differing perspectives, and communication, are that of vagueness in indicators and objectives. What a healthy stock level is, what the MSY stock level and MSY itself is, what the fishing mortality for a given target is, what the sufficient spawning stock level is; these questions may all have vague answers. These vague answers, or vagueness originating in their communication, may preclude efficient management (Peterman and Anderson, 1999). That HCRs have become a part of the political domain (Figure 1) may make such vagueness and complexity with regard to uncertainty explicit and tangible for decision makers, or may make it necessary to better communicate complexity and uncertainty to decision makers (Stirling, 2010). Another source of uncertainty and vagueness that relates to indicators and objectives is arrival of new knowledge that require changes to management, for example with regard to endangered species, climate change, renewed understanding of ecosystem function, or technical progress in fishing operations. Finally, concern over validity and relevance of modelled indicators has led to suggestions

of schemes based on measured indicators. In particular, Kelly and Codling (2006) discuss a scheme based on empirical indicators and process management that may provide a better and more transparent basis for rule-based management. Ultimately, indicator values will remain subject to uncertainty, and future fisheries management should strive to make HCRs robust to this uncertainty (Schrank and Pontecorvo, 2007).

As discussed above, HCRs reflect "explicit or perceived management objectives" (Punt, 2010, p. 583), but how objectives should be modeled, or what constitutes pertinent objectives for fisheries management, is not clear. Objectives modelling thus pose a crucial challenge in HCR schemes. The precautionary approach, as formulated in the UN Fish Stocks Agreement, and more generally the adoption of sustainable development as principles for fisheries management requires a holistic approach, considering natural, economic, and social systems (Pascoe et al., 2014). Traditionally, fisheries management relied exclusively on biologically based objectives (Daw and Gray, 2005), and such objectives are thus usually well understood and reflected in management. But it is presently viewed as critical to incorporate user preferences and societal goals into management objectives (Deroba and Bence, 2008). "[…] fisheries objectives must be stated as value-laden and measureable in terms of benefits derived from the social and economic activities of the fishery sector" (Lane and Stephenson, 1995, p. 219). However, social benefits may be difficult to measure, and perceived importance of various social benefits may vary; both are challenges that may impede adoption of explicit social objectives (Pascoe et al., 2014). Further, overfishing or ecological crisis may overshadow social issues, or lack of social data may explain the lack of awareness among fisheries managers (Symes and Phillipson, 2009). A related, paradoxical issue is that a focus on sustainable development has "weakened the status of social objectives" (Symes and Phillipson, 2009, p. 1) because securing the needs of future generations has diverted the attention from current social issues. The paradox relies in part on sustainable development depending on environmental, economic, and social stability. Finally, incorporated social objectives are often poorly defined and can thus have limited effect on management outcomes. Failing to develop clear and transparent social objectives may lead to these issues being brought up late in the process, in the heat of political debate, and resulting decisions may take the form of compromise "informed less by scientific evidence and reasoned logic than by sentiment and a quest for political advantage" (Symes and Phillipson, 2009, p. 4; see also Pasco et al., 2014). An approach that has been suggested to promote social objectives and stakeholder interests in fisheries management, and which is compatible with rule-based management, is multi-criteria decision analysis (Estévez and Gelcich, 2015). (See Peterman and Anderson, 1999, for a review of decision analysis with fisheries relevant examples.)

HCRs imply rule-based management that, as discussed above, reduces illusions of a "tidy science-politics boundary" (Hauge et al., 2007, p. 742), and instills in us an awareness of the complex science-policy interaction and interdependence in fisheries management. Garcia and Charles (2007) points out that as science and policy become more interconnected, and as science deals with value-laden societal issues, political agents will try to influence science in various ways. The collapse of Northern cod is a case in point, where political pressure likely influenced abundance estimates (Daw and Gray, 2005; Schrank and Pontecrovo, 2007). The current dispute over ocean calamities – widespread and disruptive changes to ocean ecosystems – is another example where science and policy questions get entangled (see Burgess and McDermott, 2015). Further, when scientific advice is established, stakeholders and user groups like fishers often have strong incentives to deviate from the advice (Daw and Gray, 2005). These incentives, combined with the inherent uncertainty, may have unfortunate but foreseeable consequences. "Caught between an environmental maybe and a negative economic certainty, [fishers] will use whatever economic and political tools are available to pressure the fishery managers to ignore, or mitigate, the application of the scientific recommendation" (Schrank and Pontecorvo, 2007, pp. 272–273).

Maintaining integrity and humility as science and policy integrate is and will likely remain a challenge, not only in fisheries, but across the socio-ecological system (Garcia and Charles, 2007). For HCR schemes, the explicit influence of policy on rules that dictate TACs, for example, could relieve TAC decisions from short-term political pressure. This feature of HCRs is only to a limited degree successfully implemented in many fisheries. Current international negotiations over large pelagic stocks in the Norwegian Sea demonstrate that there is little use of the HCR without a sharing agreement. The current system for negotiations allow for disputes, for example over opportunistic interpretations of scientific evidence, and non-compliance (Hallwood, 2016). The HCR plays little role in these conflicts. It seems HCRs are limited to times and situations where international sharing is settled. Perhaps the more transparent process management schemes, based on empirical indicators, could be valuable in such situations (Kelly and Codling, 2006).

Implemented HCR schemes deviate from the ideal, theoretical setup, and one has to distinguish between the HCR concept and its manifestation in various cases. Deviations may occur in any number of dimensions of the management scheme, including how rules and indicators are defined, and depend on the institutional and legal situation, and the political reality surrounding the fishery. Related to our discussion of science-policy frictions above, deviations also occur in how output from an HCR is used in negotiations between countries over quota distribution, for example, in shared fisheries where fishing rights are disputed, unclear, or not defined. Implementation further includes how the flexibility of an HCR scheme is exploited, for example

towards adaptive governance principles (Schultz et al., 2015). The various cases discussed above provide examples for several of these deviations. Deviations often take the form of provisions built into the HCR scheme, a tendency likely inherited from custom in international public law (Hallwood, 2016).

Provisions of various kinds are commonplace rather than exceptions, and typically make HCRs less efficient in reaching stated objectives. Such provisions are typically made to protect stakeholders' interests, often in terms of income or employment, and sometimes as ad hoc measures to address uncertainty and unexpected events (shocks). An example is the North Sea cod, which has a number of provisions in its HCR scheme. One problem with such provisions is that they may be mutually exclusive, and resolving conflicting provisions may be costly and certainly subject to political pressure. But these provisions may benefit both the establishment and implementation (for example, with regard to compliance) of HCR schemes and may, as such, be rational. One can also view HCRs as providing decision makers and stakeholders with a handle on scientific advice while provisions erode scientific influence on the policy outcome (Daw and Gray, 2005). However, provisions can also help achieve stated objectives if they reduce allowable catches. Alaska groundfish is a case in point.

While HCRs can have economic optimization or ecosystem resilience as objectives (Punt, 2010; Froese et al., 2011; Eikeset et al., 2013), the typical target is MSY or, if the fish stock is overexploited, recovery as fast as possible. A focus on both conservation and economic development contrasts the traditional approach to fisheries management (Guerry et al., 2015), and incorporation of ecosystem effects in management schemes, for example to maintain ecosystem services, has no or little application (Barbier, 2011; Skern-Mauritzen et al., 2015). The future development of HCRs has to embrace the "central challenge of the 21st century" (Guerry et al., 2015, p. 7348) and contribute towards governance systems that achieve sustainable resource use (Peck et al., 2014). Relevant questions that require continued attention, then, are how to formulate objectives, what are relevant targets and indicators, and how to establish these in the political reality of fisheries management (Grainger and Parker, 2013).

Finally, climate change poses a range of challenges for future HCRs (Hoel, 2008) and fisheries management more generally (Brander, 2010). Some of these challenges are similar to challenges already mentioned. For example, climate change may require adjustments to an HCR scheme if, for example, fish distributions change and disrupt existing fishing agreements between nations. If climate change leads to more and larger fluctuations in environmental conditions, predictions about stock development may become more uncertain, and challenges related to uncertainty will become more pertinent. Scientific and fisheries management institutions will in general have to adapt to a changing environment, and the HCR concept with them. While fish population dynamics obviously depend on environmental conditions, environmental measures are seldom taken explicitly into account when giving management advice (Skagen et al., 2013; Pershing et al., 2015). On another note, climate change take place on a different time scale than does fisheries management. Even long-term management plans and strategies are often reevaluated on a yearly basis, sometimes even more often. Thus, while climate change will impact fisheries and their management, the relevant time scales do not match. Thus, overfishing is seen as a more pressing concern than climate change (Beddington et al., 2007). The time scale mismatch does not mean, however, that fisheries scientists and managers can afford to be ignorant of climate change and its effects. Slow changes may lead to fast critical transitions that may be largely unpredictable (Scheffer, 2009). Critical transitions may lead to major changes in the functioning and services of ecosystems, for example by triggering trophic cascades (Heath et al., 2014), and if such transitions are latent, precautionary measures are called for. One particular measure may be to conduct research into early warning signals (Scheffer, 2009) that could be incorporated into an HCR scheme.

In the end, we should admit that HCRs are no panacea and no final solution to the fisheries management problem, something our discussion above should make quite clear. For example, challenges such as costly data collection and complex models remain as major challenges in fisheries science. And situations may occur where the most pertinent management is to deviate from an established HCR. Moreover, some fisheries may not even be suited for HCRs, but policy may nevertheless dictate their use. The complexity and uncertainty inherent in the governance of human-environment interactions may not always be amenable to general solutions. Instead, we need case-specific approaches based on broadly applicable principles and we need to remain critical and open to new ideas, perspectives, and solutions (Crépin et al., 2011; Ostrom et al., 2007).

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Contributions

- All authors but Mueter contributed to conception and design.
- Kvamsdal drafted the manuscript, with assistance from Hoel (institutional background), Eide (the structure of fisheries management), Ekerhovd (Northeast Arctic cod), Ravn-Jonsen (North Sea cod), Enberg (Norwegians spring spawning herring), Gudmundsdottir (Icelandic capelin), Mills (Pacific sardine), Mueter (Alaska groundfish).
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The authors have no competing interests.

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